Ecological and Policy Implications of Introducing Exotic Trees for Adaptation to Climate Change in the Western Boreal Forest

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EXECUTIVE SUMMARY

The purpose of this project was to evaluate the ecological and policy issues related to introducing exotic tree species in the western boreal forest of the Prairie Provinces.

Review of global experience

- Global experience with exotic introductions was reviewed as the basis for this analysis.
- Rationales for introducing exotic tree species include higher productivity, easier management, and suitability for reclaiming disturbed land.
- A new rationale is the role of exotic tree species in adaptation to climate change, which may make the environment increasingly unsuitable for native trees.
- Threats from introducing exotic species include economic losses, spread of diseases, genetic impacts on native species, site degradation, loss of aesthetic values, and invasion of adjacent ecosystems.
- Globally, invasion has been the largest threat associated with exotic introductions. Species used in exotic forestry tend to be fast-growing and seed heavily, and are therefore likely to be invasive. The greatest damage results from invasive species that alter ecosystem function (e.g. forming a dense canopy that excludes other species).
- There are many examples from around the world of introduced forest trees causing serious invasion problems (e.g. lodgepole pine in New Zealand).
- Research has shown that one of the best predictors of which species will become invasive is invasive behaviour elsewhere. Invasiveness is also more likely for species with wide native ranges, and with high reproductive capacity.
- Several systems have been developed for screening proposed introductions to prevent invasion problems. In the American system, exotic tree species from other continents are considered to pose a greater threat than those that are native to other parts of North America.

Potential species for introduction in the western boreal forest

- A simple climatic envelope model was used to assess suitability of the western boreal environment for a large number of tree species, both under the current climate and under three GCM scenarios for the 2050s.
- Native boreal species are expected to shift northward in distribution, probably declining in viability in the southern parts of their current range.
- Hardwoods of the southern prairies such as Manitoba maple and green ash may be suitable for a larger and more northerly range in the future.
- Species of the Great Lakes region may be limited in suitability for our region by climatic dryness, which is expected to increase with climate change.
- Western conifers such as Douglas-fir and ponderosa pine may increase in suitability for our region with the shift to warmer, milder-winter climates.
- Eurasian boreal species such as Scots pine and Siberian larch may show similar trends to our native boreal species, declining in viability in the southern part of the region with climate change.

- The biology of selected species, including assessment of invasion problems, was reviewed in more detail.

Policy on introduction of tree species

- Conservation organizations advocate the use of native species and recommend biological assessment, benefit/risk analysis, and controlled field trials prior to widespread introduction.
- Most governments do not have strong policies against the introduction of exotic species.
- Recent policies in South Africa and New Zealand have placed the legal onus on those introducing exotic species to prevent their spread to adjacent land.
- In Canada, legislation is largely aimed at plant diseases, not at plants themselves. Provincial weed acts are aimed at agricultural weeds.
- Policy for provincial forests generally requires regeneration of native trees following timber harvesting.

Role of exotic species in adaptation to climate change

- The new ecosystems that result from climate change can be expected to be different from those we see now, and probably different from those seen previously.
- The idea of protecting representative examples of natural ecosystems may become meaningless, and be replaced by focus on maintaining resilience, diversity and connectivity.
- Climate change may require abandoning the laissez-faire approach and assisting the movement of species to newly suitable habitats.
- The key question is not whether species is exotic, but whether it contributes to biodiversity preservation, or causes problems because of exponential population growth.

Stakeholders' workshop

- A stakeholders' workshop, attended by representatives of various forest management agencies and companies, was used to explore these issues.
- The above information was presented to the stakeholders. A series of discussion questions was then used to obtain input on desired exotic tree policies.

Policy recommendations

- Planting of exotic tree species is acceptable in some situations, but not in all situations.
- Individual exotic species should be subject to a standardized assessment process and evaluation of benefits versus risks.
- Assessments should vary with the type of land proposed for planting.
- Invasiveness should be one of the most important considerations in assessment.
- Value in adapting to climate change should be an important consideration in the assessment.
- Limited planting trials, with appropriate monitoring and evaluation, should precede widespread planting.
- In the case of species and situations where widespread planting has already happened, assessment should still take place.

- Plantation planning guidelines should be developed to reduce risks associated with planting exotic species.
- Governments should review their current policy on exotic trees and develop new policy to address exotic species issues.

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. INTRODUCTION

The purpose of this report is to assess the issue of introducing new tree species into the western boreal forest, and discuss government policies to address this issue. The regional focus is on the Boreal Plain Ecozone of Manitoba, Saskatchewan, and Alberta. This region encompasses the main areas of production forestry in those provinces, but also includes parks and other protected areas, as well as areas along the southern fringe of the region where forest has been cleared for agriculture.

There has always been an interest in finding new plant species that will be useful for agriculture, horticulture, or forestry. A few exotic[1] tree species form the foundation of commercial forestry in many parts of the world (Richardson 1998). Pines (*Pinus*) and eucalypts (*Eucalyptus*) are the most widely introduced genera for conventional forestry in temperate regions, but species of fir (*Abies*), larch (*Larix*), spruce (*Picea*), Douglas-fir (*Pseudotsuga*), beech (*Fagus*), oak (*Quercus*), and poplar (*Populus*) have also been used in temperate regions. Many legumes such as acacias (*Acacia*) are introduced for nonconventional forestry (e.g. erosion protection, fuelwood production) (Richardson 1998).

Haysom and Murphy (2003) pointed out that the increase in exotic plantation forestry is paralleled by a separate global agenda focused on the dangers of invasive exotic species.

> "Although large numbers of tree species have been introduced from one region to another in the past, most of these species do not naturalize[2] and, of those that do, not all become invasive[3]. However, there are now several well-documented studies that show the hazards that can result from an introduced tree or woody shrub becoming invasive…"

Introduction of exotic species is considered one of the most important threats to native biodiversity; it contributes to the decline of almost half of all endangered species in the United States (Wilcove et al. 1998). One of the directions in the Canadian Biodiversity Strategy (Environment Canada 1995) is to: "Take all necessary steps to prevent the introduction of harmful exotic organisms and eliminate or reduce their adverse effects to acceptable levels…" A national blueprint for addressing the threat of invasive exotic species has recently been developed. The Saskatchewan Biodiversity Action Plan (Government of Saskatchewan 2000) states as an objective to "Formulate provincial policy and management strategies on non-native organisms. Areas for action could include…ensuring non-native species are not introduced in the province until ecological implications are understood…" Current thinking on sustainable forest management emphasizes the need to maintain the natural biodiversity of our forests (CCFM 1998), implying use of natural regeneration or planting of native tree species.

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[1] Exotic species are those that are not native to a given area. Synonyms are "alien" and "non-native". For more detailed definitions, see Appendix 1.
[2] Naturalized plants are exotics that reproduce and form self-sustaining populations in their new environment.
[3] Invasive plants are those that not only naturalize where they are introduced, but also reproduce at a distance from the parents, invading adjacent ecosystems.

A new dimension to the issue of plant introductions is related to climate change. There is increasing evidence for global climatic warming caused by anthropogenic emissions of greenhouse gases. The broad patterns of vegetation around the globe are controlled by climate. The tree species found in a given area are those that are adapted to the prevailing climatic conditions. As the climate in the region changes, it will gradually become less suitable for the tree species found there, possibly leading to dieback of mature trees and failure of regeneration. One method of adapting to these changes would be to intentionally introduce tree species that are better adapted to the anticipated conditions. However, this strategy and the introduction of exotic trees in general appear to conflict with current policy directions aimed at preventing the environmental problems associated with exotic species. To evaluate this conflict, it is necessary to review the global experience with introducing trees and other plants to new environments.

2. RATIONALE FOR INTRODUCING EXOTIC TREE SPECIES

The most important reasons for introducing exotic tree species are the following (Zobel et al. 1987, Richardson 1998, Hansen and Kjaer 1999):

- In many regions forestry requires a coniferous tree species where none now exist, or where the native conifers grow poorly or do not respond well to intensive forest management. For example, pines are among the most widely planted exotics due to their broad adaptability and utility.
- In many cases, exotic species will grow faster than native species, so are more useful for forestry.
- In the tropics and subtropics, native species may be more difficult to manage silviculturally than exotic species.
- Foresters frequently do not know the biology of native species as well as that of widely used exotic species.
- Seed may be more available for widely used exotic species than for native species.
- Much of the land area where exotics are grown is grassland or scrub-forestland, and exotic species are biologically more suited than native species for establishment in these habitats.
- In some cases, exotic timber species are preferred for trade purposes.
- Occasionally, exotics are used to replace native species that are susceptible to diseases or insects and cannot be grown profitably.

In many parts of the tropics, there is no option except to use exotic conifers or hardwoods because suitable native species do not exist (Zobel et al. 1987). In the temperate zone, exotics are less widely used, but some are locally important. They provide increased growth rates and broaden the range of species available to the forester, sometimes supplying a useful species where a suitable native is not available (Zobel et al. 1987). Many North American tree species have been tried in western Europe, with three becoming important in forestry because of higher yields than native species: Douglas-fir (*Pseudotsuga menziesii*), sitka spruce (*Picea sitchensis*), and lodgepole pine (*Pinus contorta*) (Hermann 1987). Douglas-fir has been the most successful overall, with large areas of plantation in France, Germany and Great Britain, but sitka spruce is more important in Great Britain and Ireland (Hermann 1987). Lodgepole pine from British Columbia and Yukon has been widely planted in Sweden, now covering 565,000 ha (Engelmark et al. 2001). The principal argument for using lodgepole is that it has similar wood quality to the native Scots pine (*Pinus sylvestris*) and produces 36% more wood volume (Elfving et al. 2001, Engelmark et al. 2001). Similarly Rehfeldt and Gallo (2001) reported the high productivity of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) introduced into Argentina. There is more use of exotic trees in parts of the temperate zone with fewer native species, such as northern Europe, whereas there have been fewer introductions from Europe to the diverse forests of North America (Zobel et al. 1987). Carnus et al. (2003) argued that the introduction of exotics in forestry increases species diversity. For example, in France introduction has added 30 new species to the 70 tree species that are native.

Introduction of exotic trees has frequently been justified by their value in reclamation of disturbed areas. According to Frelich and Puettman (1999), there are cases where severe site conditions will not allow native trees to grow, and reforestation with exotic tree species may be better than no forest at all. Exotics have commonly been planted on grasslands, shrublands, eroded lands, or abandoned farmlands where the natural forests have previously been destroyed, and have not directly replaced native forests (Zobel et al. 1987). Establishment of exotic plantations helps to protect these sites against erosion (Zobel et al. 1987). In Britain, where there has been serious concern about the impacts of exotic tree plantations, Peterken (2001) argued that they have allowed the expansion of forest cover (e.g. on moorlands), and have buffered the effects of agricultural intensification on biodiversity.

It has also been argued that using fast-growing exotic plantations to produce required wood volumes reduces the need for intensive management of the remaining natural forest (Zobel et al. 1987). Sedjo (2001) presented a variety of evidence for the potential of intensively managed plantation forests to supply a large proportion of the world's timber needs, thereby reducing pressure on the remaining natural forest. He argued that the high production on private forest land in the United States has made it possible for the National Forest system to place a higher priority on biodiversity. Similarly, in reviewing the impact of the introduction of lodgepole pine into Sweden, Elfving et al. (2001) and Engelmark et al. (2001) suggested that the additional wood supply from this productive exotic species will allow foresters to implement biodiversity practices that reduce production, such as longer rotations and retention of hardwoods.

Williams (1997) even argued that exotic species can have positive ecological value, defined as contributing to the structure or function of a particular ecosystem. Examples include:

- Playing a similar ecological role to an extirpated native species.
- Providing a keystone food resource or habitat for certain fauna.
- Facilitating the regeneration of key species or the successional dynamics of the system.
- Positive role in the cycling of energy and materials in the system.
- Beneficial influence on disturbance regime (e.g. fire frequency or intensity).

For example, introduction of the European mosqueta rose (*Rosa rubiginosa*) into degraded Argentine forests provided shelter for reestablishment of native woody plants. In Illinois forests degraded by grazing, exotic shrubs such as honeysuckle (*Lonicera* spp.) and common buckthorn (*Rhamnus cathartica*) act as a substitute for the missing native shrubs in providing nesting habitat for certain birds (Williams 1997). Schmidt et al. (2005) found that Japanese barberry (*Berberis thunbergii*) provides nesting habitat for songbirds in forests in New York State.

Williams (1997) suggested that exotic species are more likely to have ecological value in human-altered systems than in natural systems.

Johnson and Mayeux (1992) made the extreme argument that the origin of species (exotic versus native) is irrelevant. This was based on the increasing evidence that the species composition of major native plant communities has not been stable, but rather has changed through the centuries with species migrations and extinctions. In their view, vegetation stability is associated with physiognomy and functional processes rather than with species composition. This argument is based largely on range fluctuations in species that are native to North America (i.e. locally exotic species; see definitions in Appendix 1), and does not address the possibility that species that are exotic to North America may have qualitatively different impacts. However, Pinto et al. (1997) found that replacement of native forests by exotic species in Portugal had little effect on soil fauna.

A new rationale for introduction of exotic trees is its role in adaptation to climate change. Recent climate change assessments in the Prairie Provinces have made the point that the climate may become less suitable for the native trees, leading to reduced growth, regeneration failure, and gradual loss of forest cover. Retention of the economic and environmental values associated with forest may require introduction of exotic varieties or species that are adapted to the warmer and drier climate (Thorpe et al. 2001, Henderson et al. 2002). Williams (1997) argued that exotic plants now considered problematic may turn out to have ecological value in the future because of climate change, by filling the ecological roles played by native species that have been eliminated.

The current practice in forestry is to use planting material from the same "seed zone" as the planting site, to prevent the failures that occurred in the past as a result of planting genotypes that were poorly adapted to the local climatic conditions. However, adaptation to climate change may require that planting programs use non-local seed sources imported from further south or from lower elevations (Ledig and Kitzmiller 1992, Spittlehouse and Stewart 2003). This will require a system for conserving native gene pools in seed banks or clone banks (Ledig and Kitzmiller 1992). Seed from a lower elevation at the same latitude would be preferred, because it is adapted to warmer temperatures but the same photoperiod as the planting site, while seed from further south is adapted to a different photoperiod (Ledig and Kitzmiller 1992).

Rehfeldt et al. (1999) analyzed the genetic variation among populations of lodgepole pine (*Pinus contorta*) in British Columbia. They predicted that forest productivity could increase under global warming (assuming that genotypes and future climates are appropriately matched), because the populations that will be suited to northern areas are faster-growing than the populations that are there now. However, they pointed out that postglacial migration of lodgepole pine across British Columbia occurred at a rate of 1° of latitude per 900 years, suggesting that there will be a substantial lag in migration of the best-adapted populations for the new climates. Therefore, they recommended planting programs to transfer appropriate material between seed zones. Davis and Shaw (2001) argued that this migration should not be viewed simply as movement without adaptive change. While seeds from more southerly climates may be somewhat preadapted to the new environment, there will also be selection for new genetic combinations (e.g. of photoperiod and temperature responses). This suggests that human intervention to match genotypes with climate may require breeding of new varieties.

Ledig and Kitzmiller (1992) acknowledged the uncertainty in climate change projections, but argued that the best hedge against this uncertainty is to plant a diverse array of seed sources or even of species. For example, this approach might include mixing a local seed source with one from a warmer climate. They also argued that tree breeders should aim for generalist varieties, which will perform moderately well in a range of environments, rather than specialist varieties that will quickly become maladapted as climate changes.

3. THREATS FROM INTRODUCING EXOTIC TREE SPECIES

3.1 Economic threats

Some of the concern about introducing exotic tree species has stemmed less from ecological concern than from the possibility that plantations will fail (i.e. the risk of wasting money). For example, widespread use of exotics for timber in the northeastern United States lost favor as a result of planting of unsuitable seed sources that grew poorly, had poor tree form, or were attacked by pests (Zobel et al. 1987). According to Hansen and Kjaer (1999), many of the problems with exotic plantations are related to poor matching of species and site, or poor plantation management in general. Even Zobel et al. (1987), whose book is generally a defence of exotic forestry, recognized that exotic species are often used where native species would be more suitable. Enthusiasm for the potential of a new tree species may overrule biological information. In some cases, exotic forests have been established without planning for the utilization of the wood that will be produced (Zobel et al. 1987).

3.2 Disease threats

One of the risks of introducing an exotic species is that it may be vulnerable to insects or disease. Lodgepole pine (*Pinus contorta*) planted in high elevations in northern Sweden was extensively infected by a fungal disease (*Gremmeniella abietina*). Strains of this fungus occur in both Europe and North America but are not damaging in North America. Better selection of lodgepole pine provenances has reduced this problem, but even the northernmost provenances are damaged in the harshest parts of Sweden, whereas native pine and spruce are resistant (Karlman 2001). The solution has been to stop planting this species on sites where it is vulnerable to this disease (Elfving et al. 2001).

The greatest risk associated with lodgepole pine plantations in Sweden is considered to be the possibly of introduction of North American pathogens of this species, followed by transfer to the native Scots pine (*Pinus sylvestris*) (Ennos 2001). According to Ennos (2001), the global experience is for introduction of an exotic tree to be followed by the later arrival of pathogens from the tree's native range. For example, a needle blight (*Dothistroma pini*), which was a minor pest of radiata pine (*Pinus radiata*) in its California home, arrived in exotic plantations of this species around the world some 30-40 years after their establishment, and became a major problem in some areas (Zobel et al. 1987, Ennos 2001). Ennos (2001) hypothesized that the transfer of a co-evolved host-pathogen system to a novel environment increases selection for more aggressive populations of the pathogen, leading to higher damage levels than in the native environment. When Douglas-fir (*Pseudotsuga menziesii*) was introduced into Europe, needle cast fungi from North America accompanied it and changed into more virulent strains; however better selection of provenances from North America has minimized this problem, allowing widespread planting of Douglas-fir (Karlman 2001).

Exotic introductions can lead to worldwide transmission of diseases. Eastern white pine (*Pinus strobus*) introduced into Europe was infected by white pine blister rust (*Cronartium ribicola*) transmitted from European pine species. This disease was later transferred back to eastern white pine populations in North America, where it became a major problem (Karlman 2001). Other examples of worldwide spread of fungal diseases are Dutch elm disease (*Ophiostoma ulmi*) and chestnut blight (*Cryphonectria parasitica*) (Karlman 2001).

3.3 Genetic threats

Exotics may hybridize with locally well-adapted species, leading to loss of adaptation to specific site requirements (Hanson and Kjaer 1999).

Particular concern has been attached to plantations of genetically modified trees, where there is a risk of introgression with related natural populations, leading to spread of the engineered traits (Carnus et al. 2003). NRCan (2002) suggested that Norway spruce (*Picea abies*) would be a suitable species for genetic modification in Canada, because it will not interbreed with native spruces.

Strauss et al. (2001) discussed the concerns about planting of transgenic poplars (*Populus* spp.) in the United States. The main traits that would be sought in development of this material are herbicide tolerance, wood chemistry or structure modification, insect resistance, and sexual infertility. Development of herbicide tolerance raises the risk of spread of this trait into closely related wild populations. Transgenic sterility will reduce this problem, but containment will not be perfect. This problem may require use of less friendly herbicides to control herbicide-resistant trees, and may promote excessive use of herbicides. Changes in wood characteristics could lead to ecosystem effects (e.g. altered nutrient cycling), but the differences with native poplars would be much less than the differences between poplars and conifers. Development of insect resistance could affect insects that require a poplar food source, but this would affect a narrow range of insects. Insect-resistant trees could also become more competitive in the wild. Strauss et al. (2001) argued that transgenic trees would still be limited by habitat requirements (e.g. cottonwoods would still require moist sites), which would prevent them from becoming “supertrees”. However, suitable habitats for poplars in the western boreal forest extend over vast areas, so this argument is not very convincing.

3.4 Site degradation threats

Fast-growing plantations, whether of exotic or native species, can deplete soil nutrients, requiring addition of fertilizer (Zobel et al. 1987). Other claims of site degradation by exotic plantations have not been supported by research, according to Zobel et al. (1987). However, planting of exotic conifers in Britain has led to accumulation of needle litter, podsolization of soils, and acidification of streamwater (Peterken 2001, FOE 2004).

3.5 Biodiversity threats

Forest trees help to create the habitat for other plants and animals, so changing the tree species can have a variety of impacts on biodiversity. Zobel et al. (1987) dismissed this concern, citing several cases in which "wildlife" (broadly defined) became more abundant following exotic plantations. However, modern concepts of biodiversity have become more sophisticated, recognizing that there is a wide range of plant and animal species that may be affected in different ways.

In Sweden, Engelmark et al. (2001) argued that most of the effects of lodgepole pine (*Pinus contorta*) plantations on biodiversity are really the effects of intensive forest management. Increasing biodiversity requires more areas managed on long rotations and more retention of hardwoods in conifer-dominated stands, and this is as true for native conifer plantations as it is for exotics. Sjoberg and Danell (2001) found that vertebrates use lodgepole pine similarly to native conifers, and argued that a greater admixture of broad-leaved species in conifer plantations would have a bigger effect on vertebrate diversity than changing the conifer species. However, Engelmark et al. (2001) reported that lodgepole pine stands are shadier than the native Scots pine (*Pinus sylvestris*) stands, reducing the diversity of the understory plant community. The denser shade of lodgepole stands presumably reflects higher leaf area, and so is related to the higher productivity of these stands.

Britain has seen a large increase in plantation of exotic tree species (Peterken 2001). In some cases, these plantations have caused complete changes in plant community, as in the conversion of open moorland to conifer forest. The shady conifer stands have low plant diversity beneath them, by contrast with plantations of exotic broad-leaved species that have similar ground vegetation to native broad-leaved forest. Many invertebrates are found on exotic trees, but generally not as many as on native trees. Because of the increased area of conifer forest resulting from exotic plantations, some birds that use conifers have increased in population.

3.6 Aesthetic threats

In Sweden, Engelmark et al. (2001) pointed out the increasing value being attached to rural areas as natural counterpoints to urbanization. They argued that any intensive forestry reduces this value, but that planting an exotic species has a greater impact because it affects the unique character of the Swedish landscape. They considered this factor to provide an argument for restraint in exotic planting programs. In Britain, extensive conifer plantations are widely criticized for their alteration of the traditional character of the landscape, and foresters have been forced to include more broad-leaved species in plantations, even though their economic return is less (Wallace 1981). On the other hand, in New Zealand, where spread of exotic conifers from plantations has been documented, many people find pastures invaded by scattered conifers to be visually attractive (Ledgard 2003). This shows that, while there may be aesthetic threats from exotic trees, there may also be aesthetic benefits.

3.7 Invasion threats

Globally the most important threat attached to introduction of exotic plants is the potential for invasion of adjacent ecosystems. According to Despain (2001), the impacts of exotic trees would be of little concern if they remained restricted to plantations, because their location and size is easily controlled. However, many exotic species become invasive. Species used in exotic forestry tend to be fast-growing and seed heavily, so are more likely to be invasive (Hansen and Kjaer 1999).

According to Richardson (1998), *Pinus* and *Acacia* are the most prominent exotic forestry trees on weed lists (i.e. introduced trees that have become invasive). Richardson et al. (2000) estimated that 50% to 80% of invasive plants are weeds, meaning that they grow where they are not wanted because of economic or environmental effects. According to Vitousek (1990), while most successful invasions do not alter large-scale ecosystem processes, some invasions do have major impacts. This happens in situations in which invaders:

1. Differ substantially from natives in resource acquisition or utilization
2. Alter the trophic structure of the invaded area
3. Alter disturbance frequency and/or intensity

For example, *Myrica faya*, an invasive tree in Hawaiian forests, alters ecosystem-level properties by fixing atmospheric nitrogen and thus expanding the resource base of the ecosystem (Vitousek and Walker 1989). Native to the Azores and the Canary Islands, it is a successful invader due to its prolific seed production, rapid growth rate, and widespread seed dispersal by exotic birds (Vitousek and Walker 1989). Research has shown that growth of native vegetation is nitrogen-limited, and that *Myrica* provides more nitrogen to the system than any other biotic or abiotic source (Vitousek and Walker 1989). While this additional nitrogen is available to other plants, *Myrica* uses it to form dense stands that exclude the native trees, and its litter inhibits germination of *Metrosideros polymorpha,* a dominant native tree (Walker and Vitousek 1991). A similar eutrophication effect can occur where deep-rooted invading plants bring soil nutrients to the surface, where they may be available to a range of organisms. For example, Hodgkins (1984) described a conservation area on sand dunes in Wales that has been invaded by hawthorn (*Crataegus monogyna*) since the elimination of rabbit-browsing. The soil beneath the hawthorn is becoming enriched in nitrogen and phosphorus, a change which will favour nutrient-demanding weedy species over the characteristic sand dune vegetation.

Invasion may also change ecosystems by resource depletion. An example is the invasion of salt-cedar (*Tamarix spp.*) into riparian areas of the semi-arid southwestern United States. Salt-cedar is a phreatophyte (rooted into groundwater) that does not actively regulate its transpiration, and thus can significantly reduce water levels in streams and marshes (Neill 1983). The ratio between two resources can also be altered by exotic invasion. On floodplains in New Zealand, the dominant native shrub (Coriaria arborea) is a nitrogen-fixer, whereas an invasive exotic shrub (Buddleja davidii) accumulates soil phosporus, altering the N:P ratio of the system (Bellingham et al. 2005).

An invasive plant that has changed the characteristic disturbance regime of an ecosystem is European cheatgrass (*Bromus tectorum*). Invasion of shrub-steppe in the Great Basin of the United States by this annual grass has increased fire frequency sufficiently to eliminate shrubs and create a cheatgrass monoculture (Pimentel et al. 2000).

According to Woods (1997), community alterations are particularly marked when an invading species is of a previously absent or scarce growth form. In most cases, community changes result from competition for light, with invading species reducing the light available to shorter plants. Invasion of shrubs into forests that normally have low shrub cover can cause reduced cover and diversity of the herbs that are shaded out. In other cases, invaders are of life forms already present in the community, but they become more dominant than native species, probably because of release from population controls present in their native habitat. For example, exotic honeysuckles (*Lonicera* spp.) invading forests in the eastern United States are more aggressive than the native honeysuckles already present. Invaders with novel phenological patterns may be particularly likely to change community patterns by altering both community structure and dynamic community properties

Of the large number of shrubs and trees that have been introduced to Britain, Peterken (2001) considered rhododendron (*Rhododendron ponticum*) to be the most damaging because it spreads vigorously into native woods and forms a dense shrub layer that excludes other species and prevents tree regeneration. Introduced forestry trees in Britain have been less invasive, but planted conifers of North American origin, including Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and Sitka spruce (*Picea sitchensis*), are regenerating and invading native woodlands on acid soils (Peterken 2001, Birnie et al. 2004).

Starfinger et al. (2003) reviewed the introduction of black cherry (*Prunus serotina*), a North American tree, into Germany starting in the 1700s. It was originally introduced for timber production, but its growth under German conditions was disappointing. It is now considered to be a noxious weed, because it forms a dense shrub layer that excludes other species, although its rate of spread within forests is slow. It has proven difficult to control, and is now accepted as a naturalized component of the central European flora. Rejmanek (1996) reported that eastern white pine (*Pinus strobus*), a North American conifer introduced into central Europe about 250 years ago, has only recently been recognized as an important invader.

Engelmark et al. (2001) assessed the potential for invasion by the North American lodgepole pine (*Pinus contorta*) in Sweden, but found that there has been little natural regeneration from plantations there, and that seed dispersal is usually limited to within 100 m of the parent tree. Nevertheless, they recommended guarding against dispersal by avoiding planting on ridges or upwind of sensitive areas, and by planting rows of native conifers around lodgepole plantations. Knight et al. (2001) argued that the extensive planting of lodgepole in Sweden, and its known invasive behaviour in other countries, mean that it will eventually spread into native forests, while its similarity in growth-form to Scots pine will make it difficult to eradicate.

Richardson and Higgins (1998) documented cases in which various pine species were introduced in the Southern Hemisphere and became invaders. These included:

- Jack pine (*Pinus banksiana*) in New Zealand
- Lodgepole pine (*Pinus contorta*) in New Zealand
- Corsican pine (*Pinus nigra*) in Australia and New Zealand
- Ponderosa pine (*Pinus ponderosa*) in Argentina, Chile, Australia, and New Zealand
- Scots pine (*Pinus sylvestris*) in New Zealand

Richardson et al. (1994) reported that pines introduced for forestry in South Africa have invaded "fynbos" (the natural shrubby vegetation of the region), causing major ecosystem alterations. According to Richardson et al. (2000), nine pine species are invasive in South Africa, and five of them appear on a list of the 84 most important environmental weeds in the region.

Ledgard (2001a) reported the concern over introduction of conifers in New Zealand because of "wilding", i.e. spread from plantations into adjacent areas by natural regeneration. This invasion is considered to threaten landscape character, native vegetation, grazing potential, and water yields. Wilding has been documented in several pine species as well as European larch (*Larix decidua*) and Douglas-fir (*Pseudotsuga menziesii*) (Ledgard 1988). The most invasive exotic tree species in New Zealand is lodgepole pine (*Pinus contorta*), because of early reproductive maturity, high cone number, high number of seeds per cone, high seed viability, and small seed. Because of wind dispersal of seeds, long-distance spread occurs most often from trees on ridges, hilltops, and north or west-facing slopes (facing prevailing winds). Sheep grazing helps to control lodgepole pine, so wilding is most likely on ungrazed land. Lodgepole pine has been declared a noxious weed in parts of the North Island, and planting of this species has been halted and most of the natural regeneration eliminated by control programs (Ledgard 2001a).

Zalba and Villamil (2002) listed numerous exotic woody species that are invasive in the remnant native grasslands of the Argentine pampas. Most of these were introduced through intentional afforestation for timber, shade, shelter-belts, or orchards. Areas invaded by exotic trees showed significant decreases in plant diversity.

Martin (1999) found that Norway maple (*Acer platanoides*), a commonly planted street tree of European origin, invades native hardwood forests near urban areas in New York state, reducing plant diversity and preventing regeneration of native tree species. Merriam and Feil (2002) documented invasion of hardwood forest in North Carolina by Chinese privet (*Ligustrum sinense*), an ornamental shrub. It has penetrated 30 m into stands, created 100% cover, and reduced herb diversity and density of small trees.

Lesica and Miles (1999) described the invasion of Russian-olive (*Elaeagnus angustifolia*), a tall shrub that is exotic to North America, in riparian cottonwood (*Populus deltoides*) forests in Montana. Factors that discriminate against the native cottonwood trees, including beaver damage and elimination of natural flooding, help Russian-olive to spread.

White and Haber (1993) surveyed botanists across Canada as to perceptions of which species of exotic invaders are problematic. Most of the species listed are herbaceous, but the following shrubs and trees are listed:

Principal invasive exotics:

- Common buckthorn (*Rhamnus cathartica*)
- Glossy buckthorn (*Rhamnus frangula*)

Moderate invasive exotics:

- Tatarian honeysuckle (*Lonicera tatarica*)

Minor invasive exotics:

- Black alder (*Alnus glutinosa*)
- Black locust (*Robinia pseudoacacia*)
- Caragana (*Carana arborescens*)
- European birch (*Betula pendula*)
- Himalayan blackberry (*Rubus discolor*)
- Lilac (*Syringa vulgaris*)
- Manitoba maple (*Acer negundo*)
- Multiflora rose (*Rosa multiflora*)
- Norway maple (*Acer platanoides*)
- Scotch broom (*Cytisus scoparius*)
- Scots pine (*Pinus sylvestris*)
- White mulberry (*Morus alba*)
- White poplar (*Populus alba*)

The species listed by White and Haber (1993) also appear on other lists for Canada or neighbouring areas. Mosquin (1997) listed the following shrubs and trees (all exotic to North America) as invasive in Canada's parks and other protected areas: European birch, Scots pine, common buckthorn, glossy buckthorn, white mulberry, and Scotch broom. In Alberta, ANPC (2000) listed the following shrubs and trees as exotic invaders: caragana, sea-buckthorn (*Hippophae rhamnoides*), elderberry (*Sambucus* spp.), Russian-olive, common buckthorn, Tatarian honeysuckle, Manitoba maple (minor), and lilac (minor). In Minnesota, caragana is described as an invasive species in savanna, woodland edge, and grassland habitats (MN DNR 2004). In Montana, the only woody plants on the noxious weeds list are Russian-olive and Scotch broom (MNPS no date).

While almost all of the species on these lists are of Eurasian origin, black locust and Manitoba maple are native to North America, but are considered to be invaders in some habitats such as disturbed and urban areas. According to survey results reported by White and Haber (1993), most botanists do not consider invasives that are native to North America to be a problem, because any increases are probably part of the natural fluctuation in community composition. Such a view is supported by the large changes over the centuries in the species composition of natural communities (Johnson and Mayeux 1992). For example, the northward expansion of forest trees following the retreat of the last continental glacier was faster for some species and slower for others, implying that "natural" community composition was continually changing (Davis and Shaw 2001). Creosotebush (*Larrea tridentata*), a dominant species in arid ecosystems of the southwestern United States, is known to have gradually spread across this region over the past 11,000 years (Johnson and Mayeux 1992). At each stage of range expansion, it would have been a locally exotic invasive plant in the new habitat. This shows that the perception of a species as an invasive exotic may depend on the time-scale: the movement of a species over a long time may be perceived as a change in range, whereas movement over a short time may be perceived as invasion. Accelerated climate change because of anthropogenic greenhouse gases may mean that changes in species ranges will be compressed into shorter time-frames than occurred in the past.

Mosquin (1997), in considering invasive species that are native to North America, differentiated between natural range extensions and those that have been assisted by human activity, but acknowledged that this distinction is not always obvious. His list of species for which human-assisted range expansion is historically documented includes mostly animals, such as brown-headed cowbird. The only plant on the list is black locust (*Robinia pseudoacacia*), a tree that is native in the eastern United States, and has invaded southern Ontario on disturbed sites. He considered monitoring of such range expansions to be a low priority.

From a survey of botanists in Canada, White and Haber (1993) reported a general opinion that invasions occur mostly in disturbed areas, and are not generally a problem for natural communities. For example, most survey respondents did not consider Scots pine a problem, even though it frequently escapes to disturbed sites such as old field and roadsides, or to open bogs or woods. However, in spite of the opinions reported by White and Haber (1993), in the Prairie Provinces exotic invasion of natural communities is a matter of great concern to ecologists and land managers. The invaders in most of these cases are herbaceous, either accidentally introduced weeds such as leafy spurge (*Euphorbia esula*) (Belcher and Wilson 1989) and purple loosestrife (*Lythrum salicaria*), or intentionally introduced forages such as smooth bromegrass (*Bromus inermis*) (Romo et al. 1990), crested wheatgrass (*Agropyron cristatum*), and sweet-clover (*Melilotus* spp.). Trees and shrubs are less conspicuous as invasive species in this region, but three species intentionally introduced for horticultural purposes, Siberian elm (*Ulmus pumila*), caragana (*Caragana arborescens*) and common buckthorn (*Rhamnus cathartica*) (Archibold et al. 1997), are extremely invasive in local areas. Common buckthorn has only been planted in a few places. Caragana, on the other hand, is one of the most widely distributed species for shelterbelt plantings. In the aspen parkland and forest fringe, where agricultural settlement is in contact with boreal forest, there are many areas where caragana has aggressively spread into aspen stands, virtually eliminating the native understory plants. Siberian elm is also widely planted, and is well-known for long-distance dispersal and germination of its wind-borne seeds. The observed invasive behaviour of some intentionally introduced species, even if they are only a small fraction of the total number of introductions, provides the main reason for concern about future introductions.

4. ASSESSING THE INVASION PROBLEM

4.1 Predicting which species will be invasive

Williamson and Fitter (1996a) reviewed the stages of a plant invasion, from introduction to “casual” to “naturalized” to “weed” status, and generalized that about one species out of ten makes each transition. Research has attempted to predict which exotic species will become invasive, given the ecological reality that weedy species can only be prevented, they probably cannot be eradicated once they are naturalized (Coblentz et al. 1991). Mack (1996) detailed the main methods that have been used to predict invasiveness:

- Identification of species that have been weeds in their home range or elsewhere.
- Compilation of the biological traits of species that are known to be invaders (e.g. reproductive traits).
- Assessment of invasive potential based on similarity between the climates of the home range and the area being invaded.
- Mathematical models (e.g. predicting invasiveness from calculated rate of spread).
- Controlled environment experiments to directly measure climatic tolerances of species.
- Comparison of traits of species of the same genus that differ in invasiveness.
- Deliberate planting beyond the current range (mostly used in agricultural crops).
- Deliberate planting beyond the current range, with experimental manipulation of environment to determine limiting factors.

Williamson and Fitter (1996b) noted the variation in results among different studies, and concluded that predictions of invasiveness must be specific to particular groups of species. Noble (1989) argued that generalized classifications or lists of characteristics are of little help in recognizing potentially invasive organisms, but that groups of species with correlated sets of ecological and physiological characteristics related to invasiveness do seem to occur. Pysek et al. (1995) argued that, because of the difficulty of defining unequivocally predictive plant traits, the performance of invading species should be assessed with respect to particular ecological situations. Invasion depends on the biological characteristics of the species, the characteristics of the receiving habitat, and the interactions between species and habitat (Heger and Trepl 2003). Because the invasion process consists of a series of steps, each of which may have different barriers that must be overcome, the invasiveness of a species is difficult to predict (Heger and Trepl 2003).

In spite of these limitations, research has identified a number of traits that are related to invasiveness. Much of the scientific literature relates to herbaceous weeds, but the material related to trees and shrubs is emphasized below:

Invasiveness elsewhere

Reichard and Hamilton (1997) compared exotic woody species that have become invasive in North America with those that have been on this continent for many years but have never maintained populations outside of cultivation. The best variable for predicting invasiveness in North America was invasive behaviour elsewhere. Similarly, Lockwood et al. (2001) found that in the United States, species that invade natural areas in one state are statistically likely to become invaders in another state. Scott and Panetta (1993) found that for plants introduced from southern Africa into Australia, the best predictor of “weedy” behaviour was identification as a weed at home. Williamson and Fitter (1996b) agreed that invasiveness elsewhere is a good predictor at the species level, but cautioned that these predictions should not be extended to other species within the same genus, because small genetic differences can cause large differences in invasiveness. Mack (1996) cited published lists of global weed species that can be used for predicting invasiveness, but noted that this method may miss species that would be invasive but have not yet migrated. According to Haysom and Murphy (2003), there are many cases in which a species is invasive in one country but not another, but this may be because the species has not been present long enough to become invasive.

Phylogenetic affinity

Among conifers, the pines stand out as having a high proportion of invasive species, probably related to their ecological role as post-disturbance colonizers (Richardson and Rejmanek 2004).

Geographic range

The idea that similarity of climates between the native range of a species and the area of introduction increases the probability of invasion (e.g. MNPS, no date) is called the homoclime approach. Richardson et al. (1994) observed that pines are more invasive in temperate than tropical zones of the Southern Hemisphere. Mack (1996) cited some cases where the homoclime approach works and others where it does not. For example, radiata pine (*Pinus radiata*) is naturalized in a much wider range of climates than its native range. The native range may not represent the true range of climates that can be tolerated, but rather the area to which the species is restricted by competition, pathogens, predators, or unsuitable soils (Mack 1996).

Williamson and Fitter (1996b) generalized that wide-ranging species (either habitat generalists or specialists with widely abundant habitats) are more likely to be invasive than scarce species with small ranges. This has been documented for many herbaceous species (Roy et al. 1991, Rejmanek 1995, Goodwin et al. 1999). Radiata pine is again an exception, because it has a very restricted native range but is invasive where it has been introduced elsewhere (Williamson and Fitter 1996b).

Life form

Williamson and Fitter (1996b) compared invasive exotic plants with native plants in the British flora and found that phanerophytes (i.e. shrubs and trees) are more likely to be invasive than other growth-forms. However, Pysek et al. (1995) found that the most successful invaders in seminatural habitats in the Czech Republic tend to be geophytes (i.e. perennial herbs).

Reproductive capacity

Research on mostly herbaceous weeds has related invasiveness to reproductive capacity (Baker 1974, Forcella et al. 1986, Perrins et al. 1993, Mack 1996, Williamson and Fitter 1996b). The importance of reproductive potential has also been recognized in invasiveness of forest trees. Hansen and Kjaer (1999) generalized that the exotic tree species used in forestry tend to be heavy seed-producers, so are more likely to be invasive. Ledgard (2001a) attributed the invasiveness of lodgepole pine (*Pinus contorta*) in New Zealand to prolific seed production: early reproductive maturity (5 to 12 years in New Zealand), high number of cones, high number of seeds per cone, high seed viability, and small seed compared to other pines. Spread of some subspecies of lodgepole is limited (in the absence of fire) by cone serotiny, so the nonserotinous coastal subspecies (*P. contorta* ssp. *contorta*) accounts for most invasions. Richardson et al.

(1990) analyzed 60 pine species and identified a functional group with short juvenile period, poor fire tolerance, serotinous cones, and small seeds that includes the most invasive species in South Africa. This group also includes jack pine (*Pinus banksiana*) and lodgepole pine (*Pinus contorta*), which Richardson et al. (1990) noted to act as pioneer species in North America. In a subsequent analysis, Rejmanek and Richardson (1996) used discriminant analysis to separate invasive from non-invasive pine species, and identified three predictive variables. The first two predictors, short juvenile period and short interval between large seed crops, suggest early and consistent reproduction which leads to rapid population growth. The third predictor, small seed mass, could be important because it is related to larger number of seeds produced, better dispersal, high initial germinability, shorter chilling period needed to overcome dormancy, and higher relative growth rate of seedlings. They also used other information to generalize that invasiveness is higher for plants with dry fruits than those with fleshy fruits. Richardson and Rejmanek (2004) applied this pine-based model to conifers other than pines, and found that predictions from the model are correlated with observed invasive behaviour. However, Simberloff et al. (2002) questioned the usefulness of this model. They studied an area in Argentina in which many exotic tree species had been planted. The species found to be most invasive were all rated as "invasive" by Rejmanek and Richardson (1996), but there were was also a long list of species rated as "invasive" that had failed to invade.

Reichard and Hamilton (1997) compared exotic woody species that have become invasive in North America with those that have been on this continent for many years but have never maintained populations outside of cultivation. While the best variable for predicting invasiveness was invasive behaviour elsewhere, it was also possible to predict invasiveness from reproductive characters: vegetative reproduction, perfect flowers (this variable emerged mainly because Gymnosperms were less invasive than Angiosperms), and no seed pretreatment needed. Pysek et al. (1995) also found that vegetative reproduction is related to invasion success in the Czech Republic.

Rapid growth

Research on mostly herbaceous weeds has related invasiveness to rapid growth rate (Baker 1974, Forcella et al. 1986, Perrins et al. 1993, Pysek et al. 1995, Williamson and Fitter 1996b). Similar generalizations have been made about trees. Hansen and Kjaer (1999) noted that trees used in exotic plantations are fast-growing, so are more likely to be invasive. Similarly, Richardson (1998) recognized that it will be difficult to replace invasive forestry trees with less invasive species, because the most productive species tend to be invasive.

4.2 Predicting which communities will be invaded

The nature of the recipient habitat is another critical determinant of the likely success of any invader (Noble 1989). Rejmanek (1989) made the following generalizations about the invasibility of plant communities:

- Invasibility declines with succession. Some type of disturbance is usually needed for invasion of natural communities, with few exotic species invading late-succesional plant communities.

- Communities in mesic environments are more invasible than those in extreme environments. Dry environments are not favourable for germination and seedling survival of many exotic species, and wet habitats do not provide open space for invaders because of fast growth and high competitiveness of native species.
- Riverbanks and floodplains are often high in invasibility, because they are transitional between mesic and hydric environments, they undergo frequent natural disturbance, and they often have human activity that causes disturbances and introduces exotic propagules.

The role of resource supply in invasibility has been supported in other studies. Low vegetation cover, implying reduced competiition for light, has frequently been found to favour invasion. Ledgard (2001a) found that invasion by lodgepole pine (*Pinus contorta*) in New Zealand is greatest where cover is lowest. Simberloff et al. (2002) found that invasion by exotic plants in Argentina was greatest in somewhat open areas such as road verges, small remnant pastures, and deer trails, and least in forest. In South Africa, Richardson et al. (2000) found that susceptibility to invasion by exotic pines is greatest on bare soil and dunes, and declines from grassland to shrubland to forest. Richardson and Rejmanek (2004) generalized that invasion by pines is usually into grassland or shrubland, with invasion of forest requiring substantial disturbance. In surveys in the western United States, invasion of undisturbed vegetation (away from roadsides) is greatest in grasslands and dry forests, and less in moist forests where dense vegetation creates a shadier environment (Forcella and Harvey 1983, Weaver et al. 2001). Environmental extremes also reduce invasion, with less occurring in alpine environments where it is limited by cold (Weaver et al. 2001), and in the driest grasslands where it is limited by aridity (Forcella and Harvey 1983). Williamson and Fitter (1996b) found that in Great Britain, invasive species are most common at low altitudes and on fertile soils. However, Richardson et al. (1994), in reviewing invasion by exotic pines in the Southern Hemisphere, observed that nutrient-rich sites with vigorous herbs are resistant to invasion.

The role of disturbance has also been supported in other studies. Lozon and MacIsaac (1997) reviewed literature on invasibility and agreed that exotic plants usually depend on disturbance for both establishment and range expansion. Richardson et al. (1994) said that disturbances that temporarily reduce competition from plants already present are needed to initiate invasions by exotic pines in the Southern Hemisphere. Richardson et al. (2000) found that in South Africa, susceptibility to invasion by exotic pines increases with moderate increases in the frequency of natural or human-induced disturbance. However, Lesica and Miles (1999) reported a contrary example. In riparian areas of the Great Plains, natural disturbance by flooding promotes regeneration of the native cottonwood (*Populus deltoides*) forest. Elimination of this natural disturbance by dam construction may actually make these forests more prone to invasion by the exotic Russian-olive (*Elaeagnus angustifolia*).

Davis et al. (2000) proposed a theory that encompasses both site conditions and disturbance: in general, a community becomes more invasible whenever there is an increase in the amount of unused resources. This could result from reduced resource use (e.g. removal of vegetation by disturbance) or increased resource supply (e.g. higher water supply, eutrophication). This theory is consistent with the various studies relating invasion to disturbance, eutrophication, etc., but is more general.

Specific biological factors may also play a role in invasibility. Ledgard (2001a) reported that because sheep grazing tends to control spread of lodgepole pine (*Pinus contorta*) in New Zealand, ungrazed areas are more susceptible to invasion. Richardson et al. (2000) found that in South Africa, susceptibility to invasion by exotic pines increases with presence and abundance of organisms required for mutualistic interactions (e.g. seed-dispersing vertebrates and mycorrhizal symbionts). Richardson et al. (1994), in reviewing invasion by pines in the Southern Hemisphere, noted that lack of mycorrhizal fungi originally prevented invasion, but that species are now widespread throughout the southern hemisphere.

Location in relation to sources of propagules also appears to be important for invasibility. Williamson and Fitter (1996b) generalized that the invasion risk increases with the size of the propagule source in the local area. For example, if an exotic plant has been widely introduced, it is more likely to be invasive. Richardson et al. (1994, 2000) found that in South Africa, susceptibility to invasion by exotic pines increases with proximity to large parent stands that have been established for a long time. Richardson et al. (1994) concluded that most barriers to invasion could be overcome by swamping with propagules. Ledgard (2001a) found that long-distance spread of lodgepole pine (*Pinus contorta*) in New Zealand seeds occurs most often from trees on windy sites (ridges, hilltops, and north or west-facing slopes). Richardson et al. (1994) made the following recommendations to reduce the likelihood of exotic pine spread beyond the boundaries of a plantation:

- Do not plant high-risk species near take-off sites for long-distance seed dispersal, especially upwind of important natural vegetation zones.
- Minimize boundary areas by planning fewer, larger plantations rather than many smaller ones.

A new aspect of the invasibility question is the potential impact of climate change. As with any species, exotic plants are adapted to a given range of climates, and climate change could either expand or shrink the range that is a suitable for a given exotic species (Dukes and Mooney 1999). However, Sutherst (2000) argued that exotic species are unique in that global change can affect them on a global scale, by affecting their sources, pathways, and destinations. Moreover, climate change may discriminate against late-successional species with long generation times that cannot quickly extend their ranges into new regions. The result may be poorly adapted communities that are susceptible to to invasion, especially by fast-dispersing species that can shift ranges rapidly (Dukes and Mooney 1999). Also, species that tolerate a wide range of climates in their native range are most likely to be successful invaders, and their climatic tolerance could give them an advantage as native species are stressed by climate change (Dukes and Mooney 1999).

Richardson et al. (2000) developed climatic envelopes for invasive species in South Africa, and predicted changes in their distribution with climate change. Sykes (2001) did the same for lodgepole pine (*Pinus contorta*) in Europe; his simulations suggested that the probability of invasion by this exotic species will increase with climatic warming.

4.3 Screening systems for preventing invasion

For the assessment of a potential introduction of a new species, the International Union for the Conservation of Nature (1987) approved the following as key pre-introduction questions (should the introduction be accepted, experimental controlled trials are recommended):

- What is the probability of the exotic species increasing in numbers so that it causes damage to the environment, especially to the biotic community into which it will be introduced?
- What is the probability that the exotic species will spread and invade habitats besides those into which the introduction is planned? Special attention should be paid to the exotic species' mode of dispersal.
- How will the introduction of the exotic proceed during all phases of the biological and climatic cycles of the area where the introduction is planned? It has been found that fire, drought and flood can greatly alter the rate of propagation and spread of plants.
- What is the capacity of the species to eradicate or reduce native species by interbreeding with them?
- Will an exotic plant interbreed with a native species to produce new species of aggressive polyploid invader? Polyploid plants often have the capacity to produce varied offspring, some of which quickly adapt to and dominate native floras and cultivars alike.
- Is the exotic species the host to diseases or parasites communicable to other flora and fauna, man, their crops or domestic animals, in the area of introduction?
- What is the probability that the species to be introduced will threaten the continued existence or stability of populations of native species, whether as a predator, competitor for food, cover, breeding sites or in any other way? If the introduced species is a carnivore, parasite or specialised herbivore, it should not be introduced if its food includes rare native species that could be adversely affected.

A number of standardized systems have been proposed for screening proposed introductions for potential invasive behaviour (Ledgard 1994, Tucker and Richardson 1995, Reichard and Hamilton 1997, Pheloung et al. 1999). Daehler et al. (2000) tested several of these systems on species introduced into Hawaii. They found that both the American system (Reichard and Hamilton 1997) and the Australian system (Pheloung et al. 1999) were successful in flagging species that have in fact proven to be invaders in Hawaii. They considered their results to show that universal screening tools can be developed, and recommended the Australian system as the model for a universal system.

In New Zealand, Ledgard (1994) found that "…conifer spread is generally very predictable…", and created a simple and reasonably reliable form for assessing the risk of spread of conifers from new plantings. The form asks simple questions, such as the species under consideration, the location, the siting, and the surrounding land use. It can be completed in a few minutes. A numerical score then rates the likelihood of spread. If the risk is high, the user can choose not to plant on that site, or consider a different species.

The South African system (Tucker and Richardson 1995) is specific to invasion of fynbos, the native shrubland there, and is more complex. The development of this model was based on previous experience with invasive species in this environment. The model was fine-tuned by running known invaders and failed introductions through it and making changes where necessary. The model depends both on properties of the species (including its home environment) and properties of the receiving habitat (e.g. precipitation, soil nutrients, fire regime, presence of vertebrate seed dispersers). Use of the model requires a knowledgeable ecologist with detailed life history information on the species and its home environment. Answers are required for 24 questions, grouped as follows:

- Broad scale environmental conditions
- Population characteristics and habitat specialization
- Dispersal
- Seed production
- Seed predation
- Life history adaptations to fynbos fire

Responses are chained together in a series of IF…THEN statements to give ratings (Low Risk 1, Low Risk 2, High Risk 1, etc.).

The American system (Reichard and Hamilton 1997) was based on an analysis of woody species intentionally introduced into North America. Discriminant analysis and classification/regression trees were used to compare 235 species that have naturalized with 114 species that have been here for a long time and have not naturalized (i.e. not found outside of cultivation). The results, in combination with other knowledge, were used to develop a "decision tree" which follows a series of questions with yes/no answers, related to place of origin, invasive behaviour elsewhere, hybrid status, vegetative reproduction, length of juvenile period, and germination requirements. Branches of the tree lead to recommendations, including "Accept", "Reject", or "Further Analysis/Monitoring Needed". It is relevant to the western boreal forest that they found that conifers tend to be non-invasive in North America, and that species native to North America do not tend to be invasive when introduced to new North American habitats.

The Australian system (Pheloung et al. 1999) was developed for regulatory authorities to evaluate plant introductions for potential for economic or environmental damage. The tool consists of 49 questions, most with yes/no answers, in the following categories:

- Domestication/cultivation
- Climate and distribution
- Weed elsewhere
- Undesirable traits
- Plant type
- Reproduction
- Dispersal mechanisms
- Persistence attributes

The system assigns a numerical score to each yes/no answer, and the scores are summed. Some questions apply to agricultural weeds, some to environmental weeds, and some to both. Therefore, separate agricultural and environmental scores can be determined. Ranges of the final score prompt a recommendation to "Accept", "Evaluate", or "Reject". The system was tested by comparing the results with subjective assessment by local experts, and gave reasonable agreement.

4.4 Priority ranking of existing invasions

Once invasions have happened, there is often a need to prioritize them for management response. Control programs are expensive and limited resources dictate that control be directed towards the biggest problems first. Zalba and Villamil (2002) cautioned that such rankings may change, because species presently classified as non-invasive could become invaders in the future due to increases in propagule pressure (e.g. from enlargement of plantation area), changes in disturbance regime, or introduction of other species that facilitate their expansion.

Zalba and Villamil (2002) constructed a ranking system for woody plant invasions in Argentina, which emphasized the characteristics of the plant community after invasion. Their system considered area of impact, population structure of the exotic species, proportions of native and exotic species in the invaded community, changes in species abundances from natural communities, and overlap with habitat of endemic species.

Mosquin (1997) provided a subjective ranking of invasive exotics in Canada's parks. His highest priorities for control were assigned to species with negative impacts that were considered to be very severe (e.g. common buckthorn, leafy spurge, smooth brome, crested wheat grass) or significant (e.g. reed canary grass, Canada thistle, glossy buckthorn). However, he placed some species with large impacts into a separate class because controlling them is unfeasible, except in small areas (e.g. purple loosestrife, downy brome). His recommendation for these species: "Live with them."

Probably the best known ranking system in North America is that of Hiebert and Stubbendieck (1993), which was developed for the U.S. National Park Service. The system uses responses to a series of questions to calculate two scores. The first score is for significance of impact, and combines current level of impact (based on distribution, abundance, effect on natural processes, threat to park resources, and visual impact) and innate ability to become a pest (based on reproductive traits, competitive ability, and known impact in other natural areas). The second score is for feasibility of control, and combines abundance within the park (based on number and areal extent of populations) and ease of control (based on seed banks, vegetative regeneration, level of control effort required, proximity of propagules, side effects of control, effectiveness of techniques such as burning and biological control). The scores are presented on a graph with level of impact on one axis and ease of control on the other. For example, in the Indiana Dunes National Lakeshore, purple loosestrife (*Lythrum salicaria*) appears in the quadrant of the graph representing "serious threat" and "hard to control", whereas Scots pine (*Pinus sylvestris*) is in the quadrant representing "lesser threat" and "easy to control". Application of this system results in many exotic species being rated as low in impact. The highest priority for control is given to species that present serious threats and are easy to control, but the authors note that few species fall into this combination. The lowest priority is given to species that present lesser threats but are difficult to control. While the system provides a logical basis for categorizing exotic problems, the manager must still make difficult decisions, such as whether to allocate resources to species that present serious threats but are difficult to control.

5. POTENTIAL SPECIES FOR INTRODUCTION IN THE WESTERN BOREAL FOREST

5.1 Climatic suitability for introduced tree species

5.1.1 Introduction

One of the issues related to introducing tree species is their suitability for the climate in the region. The native tree species currently found here are obviously suited to the current climate. If the climate changes significantly over the coming century, its suitability for the current native species may decrease, while species from other regions may be better suited.

Many species will survive under cultivation in a region where they are not part of natural communities. For example, silver maple (*Acer saccharinum*) can be grown in horticultural settings in the Canadian prairies, but the closest native populations are in northwestern Ontario. One of the main goals of cultivation is to reduce competition with the planted species, whereas in natural communities the composition is determined by competition among species. The species that are best adapted to the regional environment are the most successful in competition, while others which are physiologically capable of surviving in that environment lose out in the competitive race. Native species are also adapted to survive the climatic extremes that occur within their range, and not just to the average climatic conditions. An introduced species may survive in most years, but not during occasional extreme events such as severe droughts. Cultivation may also entail added water from irrigation, eliminating the need for adaptation to the local moisture regime

For these reasons, the climatic suitability for a given species is judged not by horticultural standards (e.g. plant hardiness ratings), but rather by climatic relationships in the area in which the species is part of the natural vegetation. One common approach is to determine the minimum and maximum values of particular climatic variables over the geographic range of a species (Tuhkanen 1980). Lenihan and Neilson (1993) argued that such models should be based not on raw climatic data (e.g. mean temperatures, annual precipitation), but rather on derived variables that are more physiologically meaningful. Their “Canadian Climate Vegetation Model” used growing degree-days, absolute minimum yearly temperature, and water deficit (calculated as potential evapotranspiration minus actual evapotranspiration). Similarly, the STASH model developed by Sykes et al. (1996) to predict ranges of European tree species used growing degree-days, mean temperature of the coldest month, and the ratio of actual to potential evapotranspiration. Mean temperature of the coldest month was used by Sykes et al. (1996) as an easily obtainable substitute for the absolute yearly minimum.

Growing degree-days express the amount of heat available during the growing season, and are calculated by summing the daily positive differences between the mean temperature and a 5°C base. Tuhkanen (1980) showed that the vegetation zonation from temperate to boreal to arctic zones is closely related to growing degree-days.

Winter minimum temperatures are included because some tree species are excluded from an area because they are insufficiently cold-hardy. Pielou (1991) noted that there are two entirely different mechanisms for cold hardiness in trees, with -40°C representing a critical boundary. Black spruce, white spruce, tamarack, jack pine, balsam fir, aspen and balsam poplar can survive temperatures below -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 -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 into the current western boreal.

The moisture index (the ratio of actual evapotranspiration to potential evapotranspiration) expresses the degree to which plant water use is limited by insufficient moisture supply. Stephenson (1990) and Frank and Inouye (1994) related the distribution of vegetation formations to the difference version of this index (actual evapotranspiration minus potential evapotranspiration). Hogg (1994) showed that the transition from forest to grassland in western Canada is closely related to a similar index (precipitation minus potential evapotranspiration).

Thompson et al. (2000a, 2000b) developed statistical distributions for the same bioclimatic variables as used by Sykes et al. (1996), over the ranges of North American tree and shrub species. For the current analysis, the 10^{th} and 90^{th} percentiles from these distributions were used to represent the climatic envelope of each species. A number of coniferous and broad-leaved species occurring within the study area and in adjacent regions were considered. The approach was to compare the present and future climates in the Canadian Prairies with the climatic envelopes of these species, to judge their climatic suitability.

5.1.2 Methods

The study area was defined as including the Prairie, Boreal Plain, and Boreal Shield Ecozones between 49° and 57° north latitude, and 95° and 120° west longitude. The current climate was represented by 1961-90 normals for monthly temperature and precipitation. D. McKenney of the Canadian Forest Service has developed continuous climate surfaces for Canada based on the 1961-90 normals, represented by a grid of points every 0.13 degrees of latitude and longitude (available from www.cics.uvic.ca). Gridpoints falling within the study area were extracted.

Future climates were represented by three scenarios for the 2041-2070 period (referred to as the 2050s). These scenarios were originally selected by Henderson et al. (2002) to represent the range of variation among the available GCM models:

- CGCM2 A21
- HadCM3 B21
- CSIROMk2b B11

Outputs from these scenarios (available from www.cics.uvic.ca) show change values between current (1961-90) and future (2050s) climates, on a coarse grid of several degrees of latitude and longitude. Change values from the scenario gridpoints were applied to the finer McKenney gridpoints for 1961-90 normals by an inverse distance-weighted interpolation, and used to calculate future values for monthly temperature and precipitation.

For both the current climate and the three scenarios for the future climate, temperature and precipitation data were used to calculate the three bioclimatic variables used by Thompson et al. (2000a, 2000b). Growing degree-days were calculated by using the Brooks (1943) sine-wave interpolation to generate mean daily temperatures from monthly temperatures, then summing the daily deviations above 5°C. The lowest monthly mean temperature (usually for January) was determined. For the moisture index, Thompson et al. (2000a, 2000b) calculated potential evapotranspiration by the Thornthwaite (1948) method, and actual transpiration from a water-balance model presented by Wilmott et al. (1985). However, when these methods were used with the Canadian data, the resulting moisture index values were significantly higher than shown in the maps in Thompson et al. (2000a, 2000b). The reason for this discrepancy was not found. However, calculating potential evapotranspiration by the Baier-Robertson method (Baier and Robertson 1965, Baier 1971) gave moisture index values more comparable to those shown by Thompson et al. (2000a, 2000b), so this method was substituted. Actual evapotranspiration was determined using the WATBAL water balance model (www.metla.fi/hanke/3098/ewat_bal.htm) (which gave similar results to the Wilmott model), assuming a loam soil with water holding capacity of 150 mm.

For each gridpoint, the climatic suitability for the species was determined by comparing the three bioclimatic variables at the point with the 10%/90% thresholds for the species given by Thompson et al. (2000a, 2000b). If all three bioclimatic variables fell within the 10%/90% range, the point was considered to be suitable. This was done for the 1961-90 normals and for each of the three 2050s scenarios. Results were mapped by comparing the present map with one of the future maps, and determining the following categories:

- Continued unsuitability – unsuitable in both 1961-90 and 2041-70
- Declining suitability – suitable in 1961-90, unsuitable in 2041-70
- Continued suitability – suitable in both 1961-90 and 2041-70
- Increasing suitability – unsuitable in 1961-90, suitable in 2041-70

These categories were mapped over the study area using MapMaker GIS software. The datum was NAD 1983, and the map projection was Lambert Conformal with standard parallels at 50.8333° North and 58.16666° North. Maps were clipped to 49° and 57° North and 95° and 120° West, and to the boundaries of the Prairie, Boreal Plain, and Boreal Shield ecozones (Ecological Stratification Working Group 1995) based on a digital map obtained from http://sis.agr.gc.ca/cansis/nsdb/ecostrat/index.html.

We were also interested in climatic suitability for Eurasian species, which were not addressed by Thompson et al. (2000a, 2000b). Sykes et al. (1996) used a similar approach to develop climatic envelopes for several north European species, including Scots pine (*Pinus sylvestris*). However, their growing degree-days variable was adjusted by subtracting the portion of the season before budburst, making it difficult to relate to the variable used in the current analysis. Tchebakova et al. (1994) gave climatic envelopes for Siberian vegetation types, which included growing degree-days and the ratio of potential to actual evapotranspiration. As a rough approximation of the climatic envelope for Scots pine, we selected the Siberian vegetation types in which it was listed by Tchebakova et al. (1994) as a major species, and determined the overall limits enclosing all of these types. We followed the same approach for Siberian larch (*Larix sibirica*). Actual climatic envelopes for these species may be somewhat larger, because they may be present in other vegetation types where they are not major species. This approach also does not consider the European range of Scots pine, where it occurs in a wide range of climates. Tchebakova et al. (1994) did not give limits for the temperature of the coldest month. However, given that Scots pine and Siberian larch are distributed widely across a region with the coldest winters outside of Antarctica, it can be assumed that they will not be limited by winter temperatures found in the study area.

5.1.3 Results

Means and ranges over the gridpoints in the study area for the three bioclimatic variables are shown in Figures 1 to 3. Each of these variables shows a substantial range because of climatic variability within the study area, but shifts in means and ranges from present to future climates are an indication of the amount of climatic change predicted.

Growing degree-days are predicted to increase substantially in all three scenarios (Figure 1). Mean temperature of the coldest month is predicted to increase in two scenarios, but not change much under the HadCM3 B21 scenario (Figure 2). The moisture index is predicted to shift downward somewhat under all three scenarios, indicating somewhat drier conditions (Figure 3).

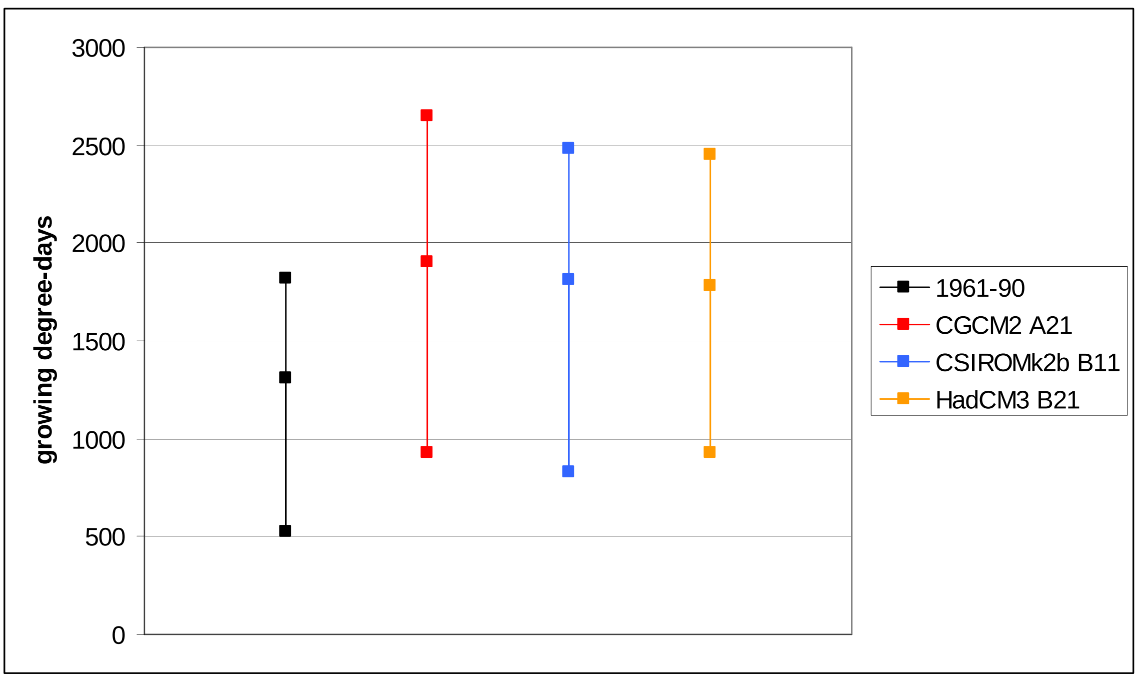

Figure 1 Means and ranges of growing degree-days (5 °C base) over the study area, under the current climate (1961-90 normals) and under three scenarios for the 2050s.

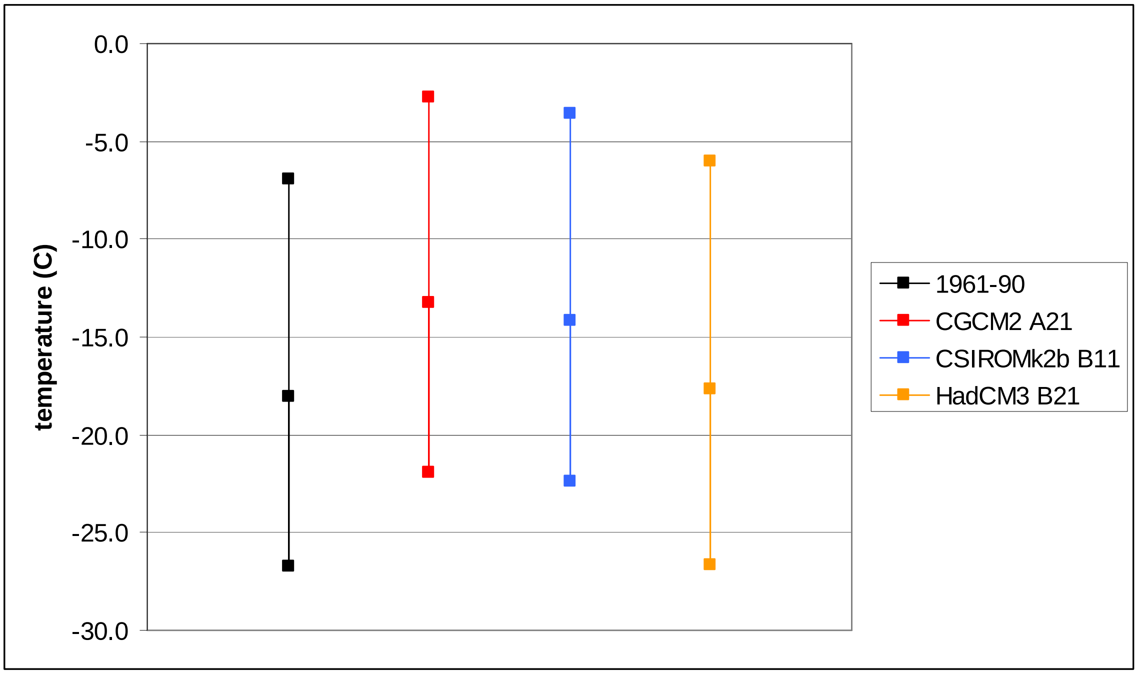

Figure 2 Means and ranges of the mean temperature of the coldest month (° C) over the study area, under the current climate (1961-90 normals) and under three scenarios for the 2050s.

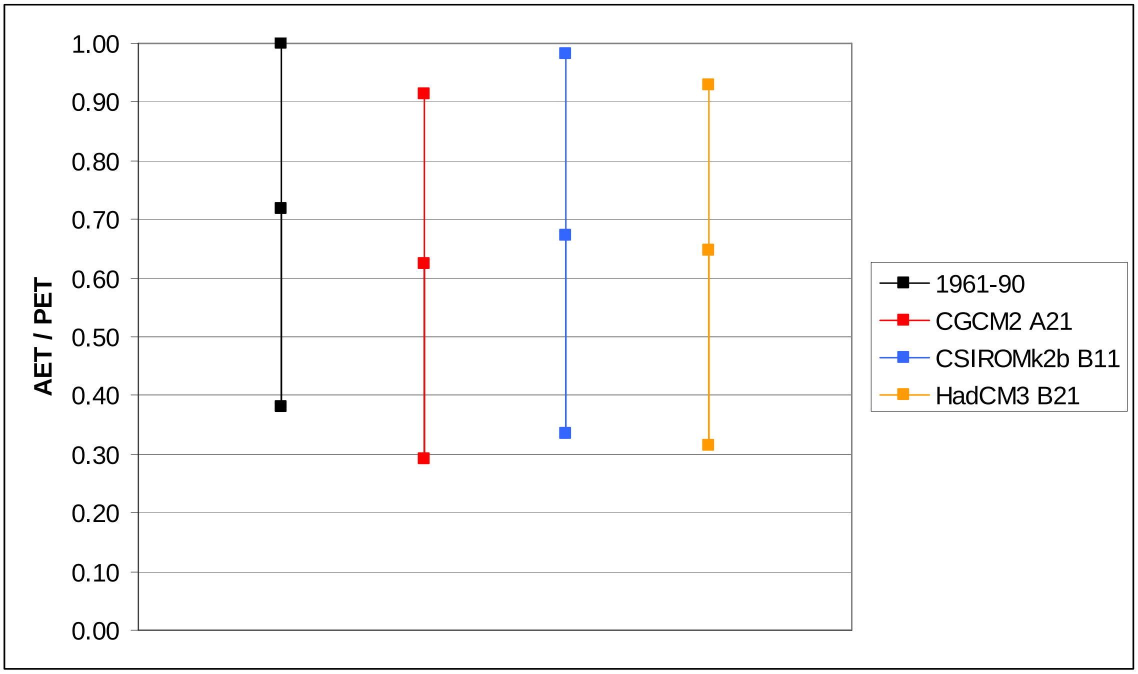

Figure 3 Means and ranges of the moisture index (actual evapotranspiration divided by potential evapotranspiration) over the study area, under the current climate (1961-90 normals) and under three scenarios for the 2050s.

The bioclimatic variables were mapped for the current climate and one of the scenarios (CSIRO Mk2 B11) (Figures 4 to 6). These maps show the same trends as the above graphs, but also show the spatial distribution. The growing degree-days variable shows a substantial change, with all parts of the study area predicted to have much warmer growing seasons (Figure 4). Mean temperature of the coldest month shows more moderate shifts towards milder winters in all regions (Figure 5). The moisture index shows slight shifts towards somewhat drier conditions in most regions (Figure 6).

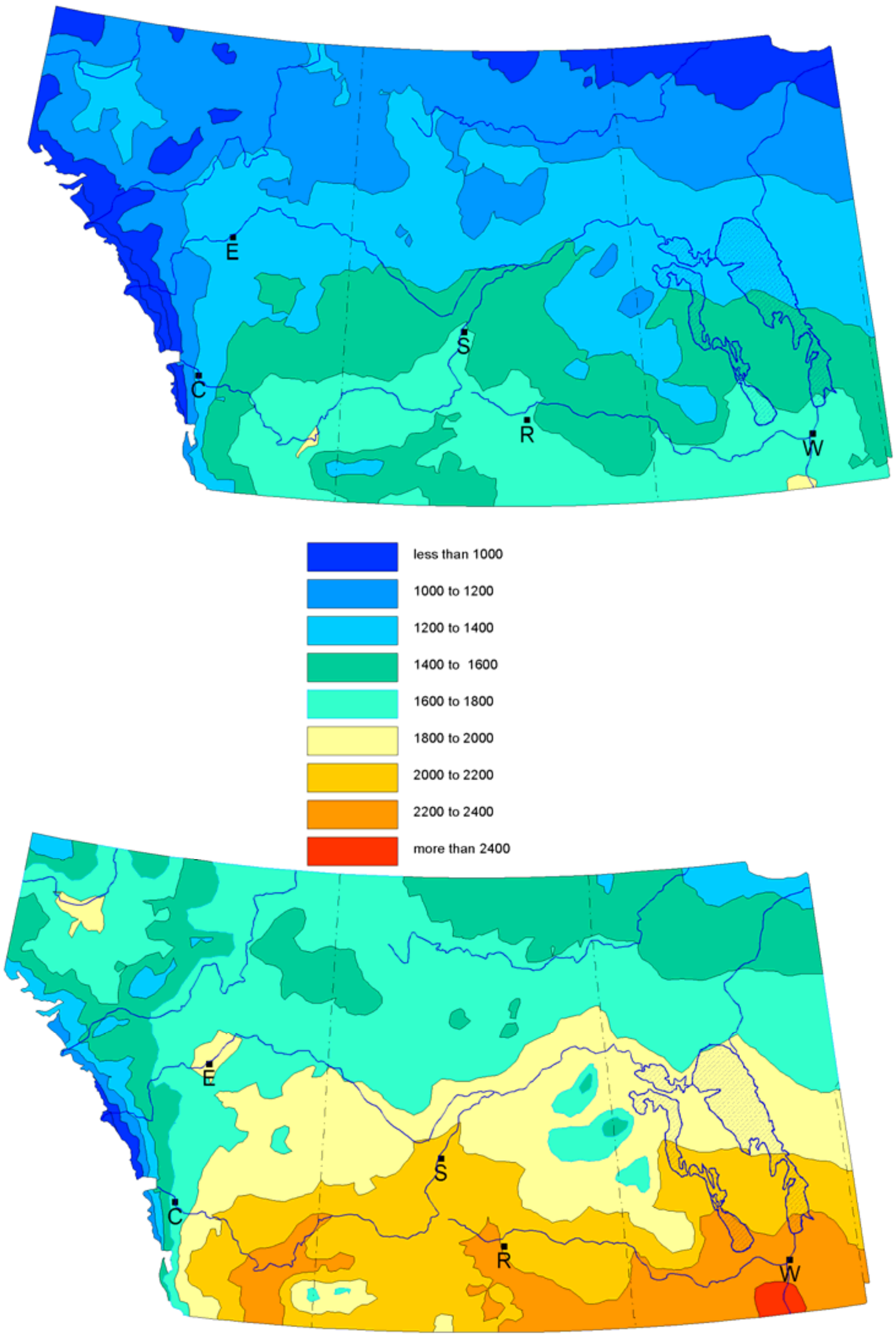

Figure 4 Growing degree-days (5° C base) for the study area, in the current climate (top) and in the CSIRO Mk2b B11 scenario for the 2050s (bottom). Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C – Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

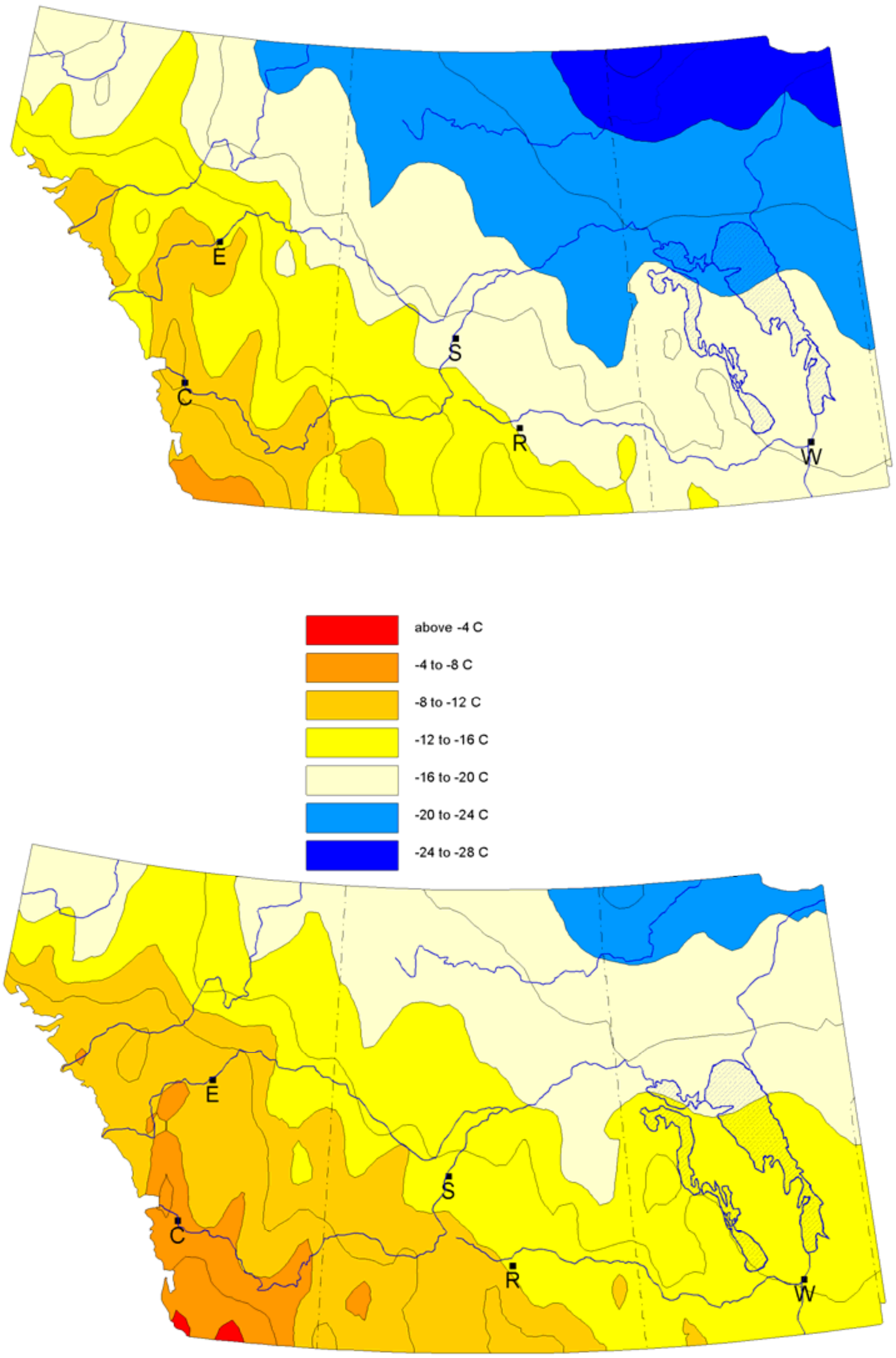

Figure 5 Mean temperature of the coldest month (° C) for the study area, in the current climate (top) and in the CSIRO Mk2b B11 scenario for the 2050s (bottom). Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C – Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

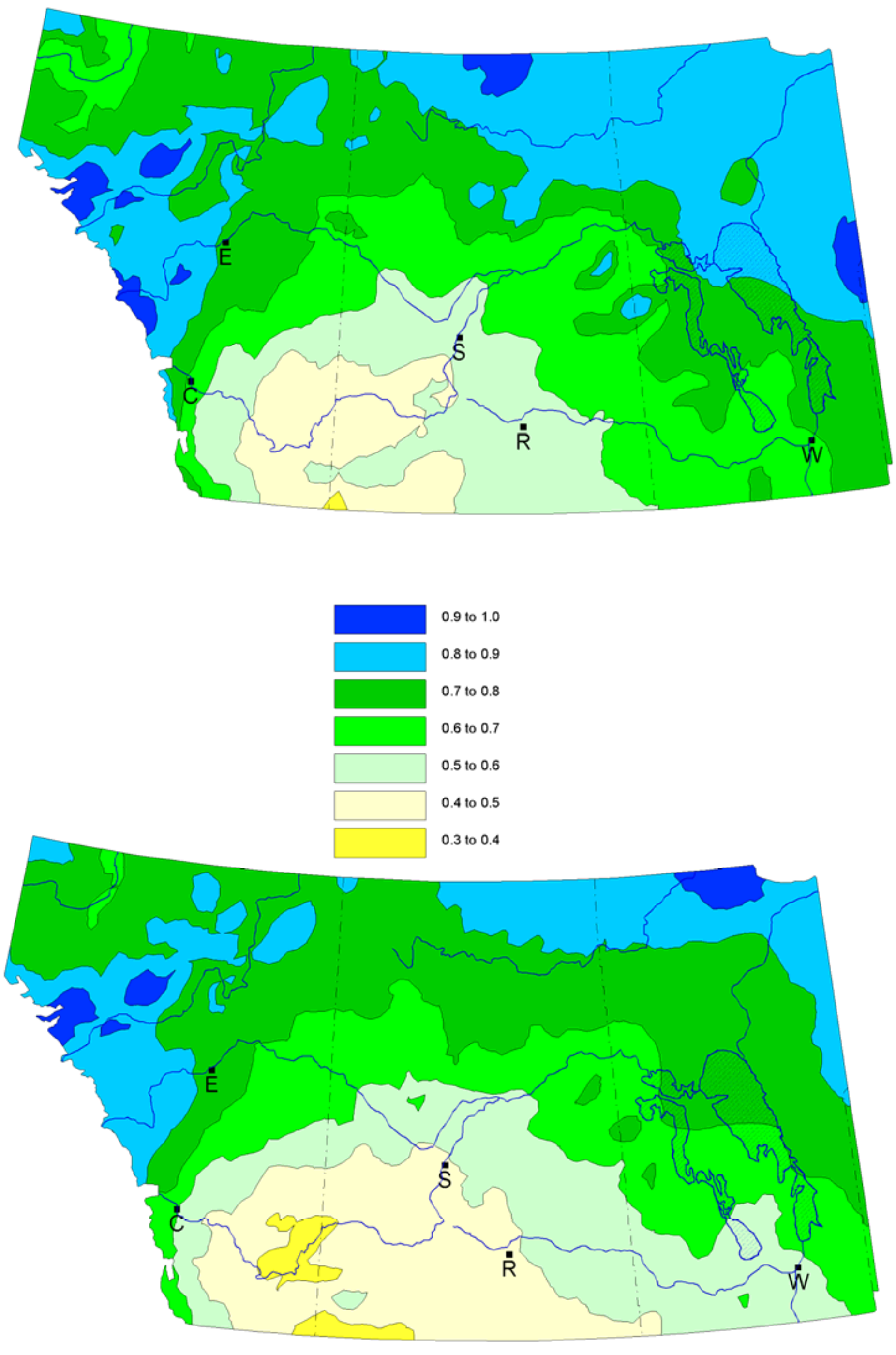

Figure 6 Moisture index (actual evapotranspiration divided by potential evapotranspiration) for the study area, in the current climate (top) and in the CSIRO Mk2b B11 scenario for the 2050s (bottom). Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C – Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

Climatic thresholds for individual species according to the bioclimatic model are shown in Table 1:

Table 1 Ranges of boreal and temperate tree species in relation to upper and lower limits of bioclimatic variables. Values for North American species are from Thompson et al. (2000a, 2000b). Values for *Pinus sylvestris* and *Larix sibirica* are interpreted from Tchebakova et al. (1994).

	Mean temperature of coldest month (°C)		Growing degree-days (5°C base)		Ratio of actual to potential evapotranspiration	
thresholds	**lower**	**upper**	**lower**	**upper**	**lower**	**upper**
Acer negundo	-17.0	7.8	1500	4800	0.52	0.98
Acer rubrum	-13.4	9.3	1400	5000	0.91	0.99
Acer saccharum	-14.7	-0.1	1300	3300	0.94	0.99
Betula alleghaniensis	-15.8	-3.3	1200	2400	0.94	0.99
Betula papyrifera	-27.7	-9.1	500	1600	0.52	1.00
Fraxinus pensylvanica	-16.6	7.5	1500	4800	0.58	0.99
Populus balsamifera	-27.5	-10.6	600	1600	0.51	1.00
Populus tremuloides	-27.8	-6.3	600	2100	0.50	0.99
Quercus macrocarpa	-16.6	2.4	1600	4000	0.66	0.98
Quercus rubra	-13.3	3.8	1500	3900	0.93	0.99
Tilia americana	-14.5	-0.7	1500	3200	0.92	0.99
Ulmus americana	-17.4	7.7	1300	4800	0.68	0.99
Abies balsamea	-23.7	-9.1	600	1700	0.73	1.00
Juniperus scopulorum	-11.6	-1.9	500	2100	0.42	0.93
Larix laricina	-27.9	-9.5	500	1700	0.56	1.00
Picea glauca	-28.5	-11.9	500	1400	0.49	1.00
Picea mariana	-29.3	-11.6	500	1500	0.47	1.00
Picea pungens	-11.3	-4.9	400	1700	0.45	0.90
Pinus banksiana	-28.1	-13.8	700	1500	0.52	0.97
Pinus contorta	-22.3	-2.2	500	1300	0.55	0.99
Pinus ponderosa	-8.8	7.0	800	3900	0.44	0.88
Pinus resinosa	-18.1	-6.9	1200	2000	0.94	0.99
Pinus strobus	-16.9	-1.0	1200	2700	0.94	0.99
Pseudotsuga menziesii	-11.5	4.5	500	2500	0.51	0.96
Pinus sylvestris			800	1650	0.50	1.00
Larix sibirica			300	1650	0.50	1.00

Application of the bioclimatic model to the current climate gave reasonable approximations to actual distributions for the boreal species currently found in the study area: white spruce (*Picea glauca*), black spruce (*Picea mariana*), balsam fir (*Abies balsamea*), jack pine (*Pinus banksiana*), tamarach (*Larix laricina*), trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), and white birch (*Betula papyrifera*). The distributions of these native species are of interest mainly as a test of the performance of the model. Under the three scenarios for the 2050s, all of these species showed a northward range shift, consistent with other analyses in the region (e.g. Carr et al. 2004). This shift was mainly driven by the increase in growing degree-days to levels beyond the upper limit for these species. Model outputs for trembling aspen under one scenario are shown in Figure 7 as an example. Model outputs for all species and scenarios are shown in Appendix 2.

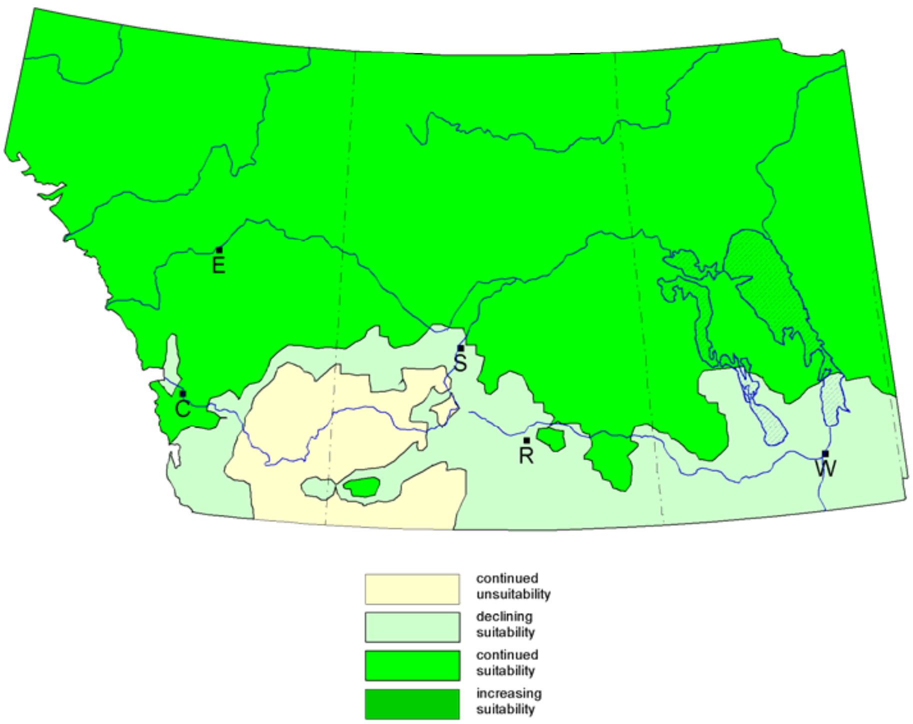

Figure 7 Changes in climatic suitability for trembling aspen from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

Suitable ranges modeled for hardwood trees of the southern Prairie Provinces, including Manitoba maple (*Acer negundo*), green ash (*Fraxinus pensylvanica*), American elm (*Ulmus americana*), and bur oak (*Quercus macrocarpa*), were generally smaller than their actual ranges. This may be appropriate, given the restriction of these species to particular habitats within their range. The area suitable for these southern species was predicted to shift northward and expand under all three climate change scenarios, driven by increasing growing degree-days and mean temperature of the coldest month. Results for Manitoba maple under one scenario are shown as an example in Figure 8.

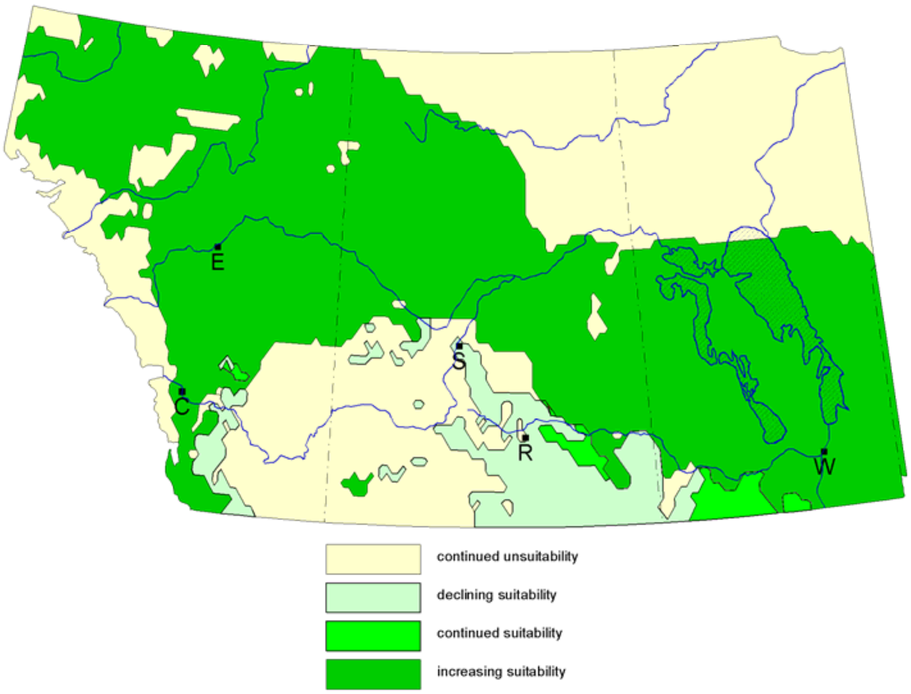

Figure 8 Changes in climatic suitability for Manitoba maple from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

Bur oak is of particular interest because it appears to be capable of growing under warmer/drier conditions than aspen. For example, bur oak is common on south-facing slopes of the eastern Qu'Appelle Valley (in Saskatchewan and Manitoba), whereas 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).

The bioclimatic model indicated that both the present and the future climates of the study area are unsuitable for tree species currently associated with the Great Lakes region and further east. There was no suitable area under any scenario for sugar maple (*Acer saccharum*), northern red oak (*Quercus rubra*), or basswood (*Tilia americana*), while there were a few suitable gridpoints under one of the 2050s scenarios for red maple (*Acer rubrum*), yellow birch (*Betula lutea*), red pine (*Pinus resinosa*), and eastern white pine (*Pinus strobus*). Generally, these species were modeled as requiring higher moisture index values than found at present or in future predictions for this region. Model outputs for red pine under one scenario are shown in Figure 9 as an example.

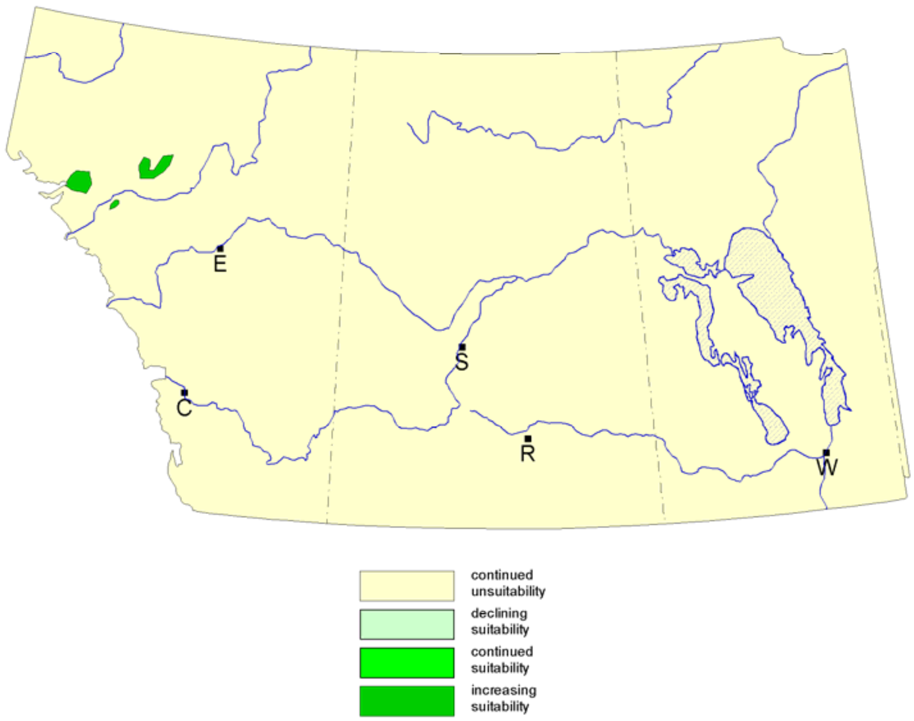

Figure 9 Changes in climatic suitability for red pine from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

Western montane conifers, including Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), Rocky Mountain juniper (*Juniperus scopulorum*), and blue spruce (*Picea pungens*), were all modeled as being suitable for climates in the southwestern corner of the study area. In all cases, the model exaggerated the suitable area compared to the actual ranges of the species. The actual distributions of Douglas-fir and Rocky Mountain juniper extend slightly into the southwest corner of the study area, while ponderosa pine occurs just to the west across the continental divide. Blue spruce's natural range is much further south in Wyoming and southward, but it is widely planted as an ornamental tree in the Canadian prairies. For all of these species, the bioclimatic model indicated expansion of suitable range under two of the climatic change scenarios, but slight contraction under the Hadley scenario. The driving factor for range expansion appears to be milder winters. These results suggest that the western species are more likely than the Great Lakes species to be useful in adaptation to climate change in our region. Their natural ranges extend to relatively warm and dry environments, such as the Okanagan Valley in British Columbia, so they may be suited to the much warmer and somewhat drier climate predicted for the study area. Results for Douglas-fir and ponderosa pine under one scenario are shown as examples (Figures 10 and 11).

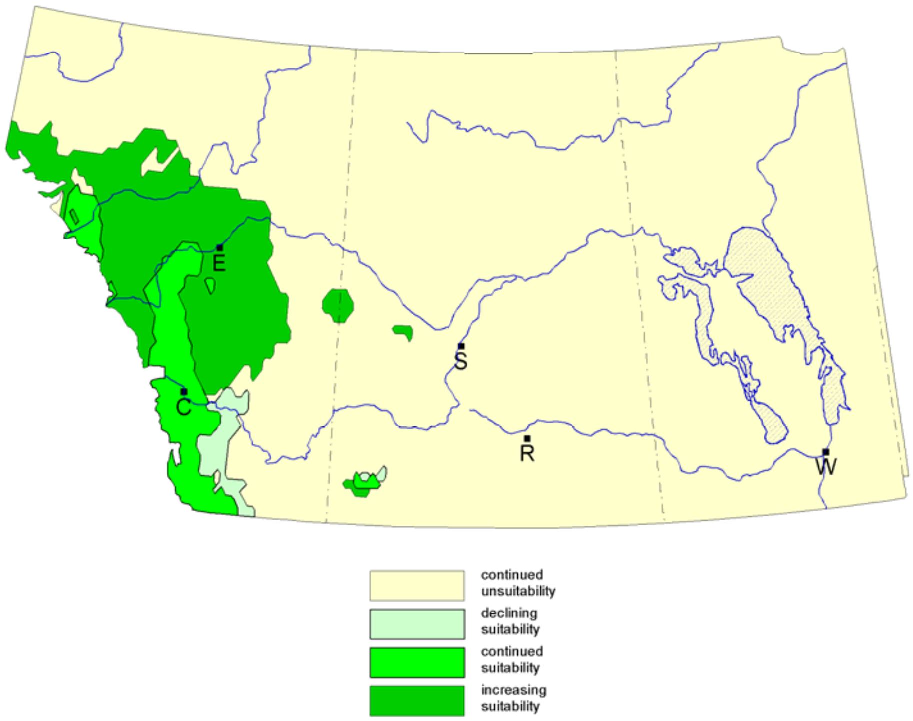

Figure 10 Changes in climatic suitability for Douglas-fir from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

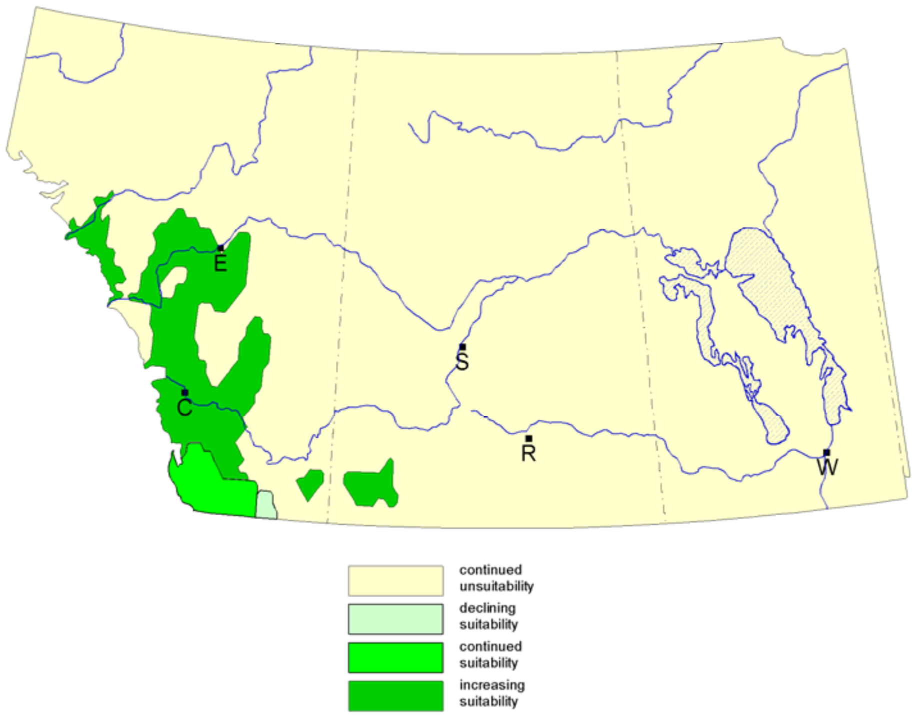

Figure 11 Changes in climatic suitability for ponderosa pine from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

Lodgepole pine (*Pinus contorta*) is distinctive in that it is a western conifer, but with a substantial range in the study area, in the Boreal Forest of northwestern Alberta. The area modeled as suitable extended further east than the actual range (Figure 12). However the absence of lodgepole further east may be complicated by biogeographic interaction with jack pine, which is closely related and occupies a similar ecological niche. The ranges of these two fire-adapted pines meet in central Alberta, lodgepole having spread from the west and jack pine from the east following deglaciation. The area suitable for lodgepole pine is predicted to decrease almost to the point of elimination under all three climate change scenarios, driven by rise in growing degree-days beyond the modeled threshold for this species. While we do not know how accurate these range limits are, the fact that lodgepole pine has a similar climatic envelope to other boreal species suggests that it may not be the best candidate for adaptation to climatic warming.

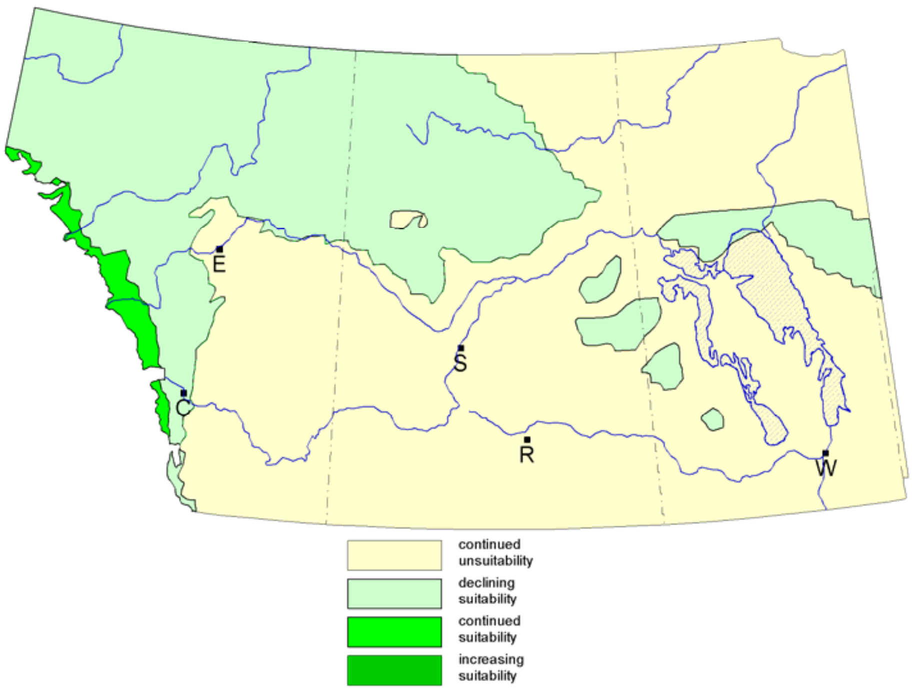

Figure 12 Changes in climatic suitability for lodgepole pine from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

Suitable climates for the two Eurasian species considered, Scots pine (*Pinus sylvestris*) and Siberian larch (*Larix sibirica*), were modeled to be similar to the native boreal trees (Figures 13 and 14), although extending further south. However, the data for climatic thresholds for the Eurasian species were less exact (see Methods), so not too much should be read into small discrepancies. As with the natives, the model predicts a substantial northward shift in the suitable climates for the Eurasian species under all three climate change scenarios, driven by increases in growing degree-days beyond the upper limit for these species (Figures 13 and 14).

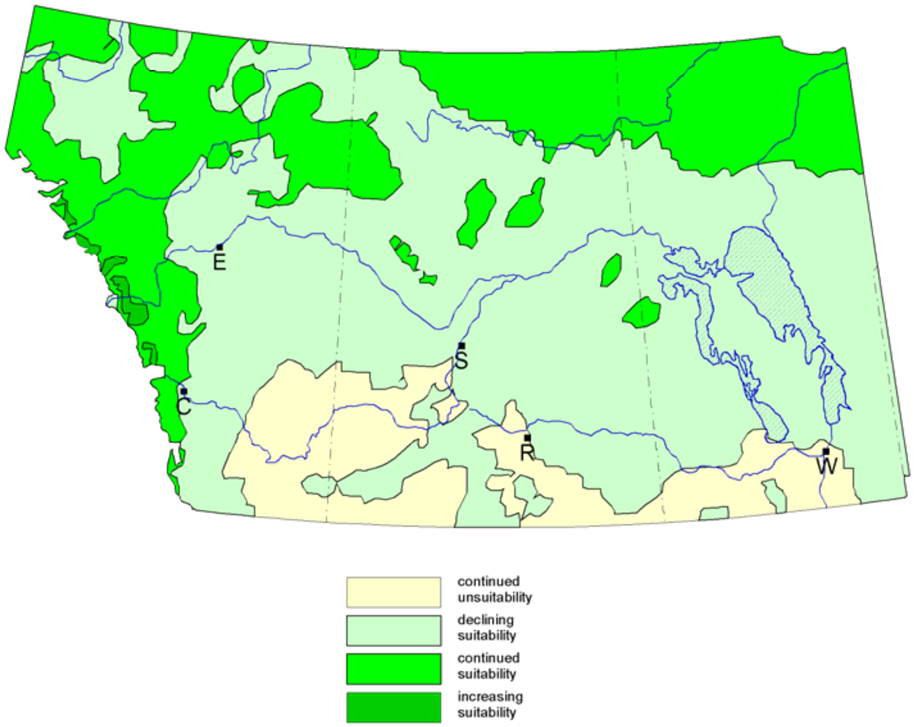

Figure 13 Changes in climatic suitability for Scots pine from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

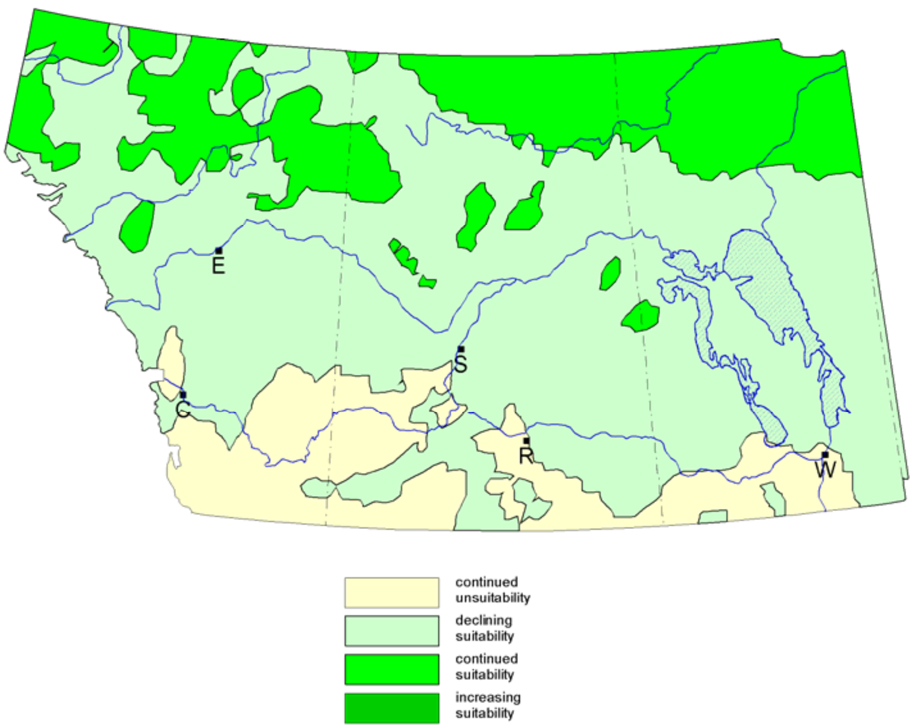

Figure 14 Changes in climatic suitability for Siberian larch from the current climate (1961-90 normals) to the CSIROMk2b B11 scenario for the 2050s. Provincial boundaries, major rivers and lakes, and major cities (E – Edmonton, C - Calgary, S – Saskatoon, R – Regina, W – Winnipeg) are shown for reference.

5.1.4 Conclusions

No model is an exact representation of reality. There are a number of limitations to the model used in the above analysis:

- The model thresholds are based on analysis of the overall range of each species, and may not reflect the details of distributions in our region.
- The model is based on correlations between range limits and climatic variables, and does not identify the actual mechanisms underlying range limits.
- Variations in methods used to calculate potential and actual evapotranspiration can lead to different results for moisture balance among literature sources.
- There may be discrepancies in standard climatic variables depending on the data collection methods used in different countries (e.g. Henderson et al. 2002).

Because of these limitations, the model should not be expected to give exact results. As noted above, there were errors in modeling the current distributions of species native to the region, although the general patterns were represented. In spite of these problems, the analysis did suggest a number of conclusions:

- Native boreal species are expected to shift northward in distribution, probably declining in viability in the southern parts of their current range.
- Hardwoods of the southern prairies such as Manitoba maple and green ash may be suitable for a larger and more northerly range in the future.
- Species of the Great Lakes region may be limited in suitability for our region by climatic dryness, which is expected to increase with climate change.
- Western conifers such as Douglas-fir and ponderosa pine may increase in suitability for our region with the shift to warmer, milder-winter climates.
- Eurasian boreal species such as Scots pine and Siberian larch may show similar trends to our native boreal species, declining in viability in the southern part of the region with climate change.

5.2 Case studies: selected species

5.2.1 Introduction

To look more closely at the issues related to tree introductions, information was gathered on several species that might be considered for planting for forestry purposes in our region. These included three western North American conifers (lodgepole pine, ponderosa pine, and Douglas-fir), one eastern North American conifer (red pine), two Eurasian conifers (Scots pine and Siberian larch), and hybrid poplar, which is currently being widely planted in the region.

5.2.2 Lodgepole pine (Pinus contorta)

Lodgepole pine is native to western North America, including British Columbia and western Alberta (Rocky Mountain foothills, boreal forest in northwestern Alberta). It is also native to the Cypress Hills in southeastern Alberta and southwestern Saskatchewan. This species has several varieties: trees in coastal British Columbia are var. *contorta* whereas those in interior British Columbia and eastward are var. *latifolia* (Hosie 1969). Lodgepole pine and the more eastern species jack pine (*Pinus banksiana*) are closely related and occupy similar ecological niches. Their ranges overlap slightly in central Alberta, lodgepole having arrived from the west and jack pine from the east following glaciation (Yeatman 1967). Lodgepole pine differs from the other species considered in that it is native in a significant part of the study area, but would be exotic if planted in the boreal region of Saskatchewan and Manitoba.

The ecoclimatic model in Section 5.1 showed that much of the boreal forest region, especially in the western part of the study area, is suitable for lodgepole pine. The model implied that it is limited by winter cold from growing in the colder eastern part of the study area, limited by excessive warmth from growing in the southern boreal forest as well as the grassland region, and limited by aridity from growing in the grassland region. The range in northwestern Alberta has relatively cool summers (low growing degree-days) but milder winters (higher mean temperature of the coldest month) than further east in the boreal forest. The suitable range was modeled to shift northward under future climates, to the point of elimination from most of the study area, apparently limited by excessive warmth as well as aridity.

Lodgepole pine begins producing seed early, at 5 to 15 years. Cones are usually serotinous (i.e. remaining closed at maturity) in *P. contorta* var. *latifolia,* but the variety does include non-serotinous populations. Serotinous cones require fire to open, so represent an adaptation to heavy seed dispersal following fire. By contrast, non-serotinous populations disperse seeds more continuously through the life of the tree. The seeds are relatively small and disperse easily with the wind, but most seeds fall within 60 m of the parent tree. Seedlings establish best on exposed mineral soil (www.fs.fed.us/database/feis/plants/tree).

Lodgepole pine is an aggressive pioneer species following disturbances, especially fire. It quickly establishes dominance on disturbed sites because of high seedling survival and rapid early growth. Populations with serotinous cones produce even-aged stands following crown fire. These are often succeeded by more shade-tolerant species (Douglas-fir, Engelmann spruce, subalpine fir) 50 to 100 years after fire, but lodgepole pine may remain the dominant species in areas with frequent stand-replacing fires. Populations with non-serotinous cones may produce uneven-aged stands. On certain sites, no other species can grow, and lodgepole pine produces open self-regenerating stands (www.fs.fed.us/database/feis/plants/tree).

The mountain pine beetle attacks mature lodgepole pine stands, killing most mature trees and creating open, multi-aged stands. Mountain pine beetle does not occur north of the -40°C isotherm of annual minimum temperature, excluding it from lodgepole pine stands in Yukon, Northwest Territories, and Alberta. Climate change is expected to increase the area vulnerable to mountain pine beetle (http://wlapwww.gov.bc.ca/air/climate/indicat/beetle_id1.html).

Lodgepole pine is invasive in other countries where it has been introduced. In a global survey, Binggeli (1996) listed lodgepole pine as highly invasive. Richardson and Rejmanek (2004) listed lodgepole pine as naturalized in Argentina and Russia, and invasive in Australia, Chile, Great Britain, Ireland, New Zealand, and Sweden. Their model of invasiveness based on biological characteristics showed the coastal variety (*Pinus contorta* var. *contorta*) as the most invasive of the pines examined (Rejmanek and Richardson 1996).

In New Zealand, lodgepole pine is the most invasive introduced species (Ledgard 2001a). Ledgard (2001a) attributed its invasiveness to early reproductive maturity, high number of cones, high number of seeds per cone, high seed viability, and small seed compared to other pines. Most invasion is into open grassland, with less in shrubland and none in closed forest. On level ground, most spread is limited to 50 m from the parent trees, but trees on "take-off sites" (e.g. ridges, slopes facing the prevailing winds) can cause distant spread. Spread is limited (in the absence of fire) by cone serotiny, so the nonserotinous coastal variety (*P. contorta* var. *contorta*) accounts for most invasions. However, in Australia, plantations of the interior variety with serotinous cones (*P. contorta* var. *latifolia*) release massive amounts of seeds following fire, forming dense regeneration and preventing recruitment of native plants (Weber 2003).

Engelmark et al. (2001) assessed the potential for invasion by lodgepole pine in Sweden, but found that there has been little natural regeneration from plantations there, and that seed dispersal is usually limited to within 100 m of the parent tree. Nevertheless, they recommended guarding against dispersal by avoiding planting on ridges or upwind of sensitive areas, and by planting rows of native conifers around lodgepole plantations. Knight et al. (2001) argued that the extensive planting of lodgepole in Sweden, and its known invasive behaviour in other countries, mean that it will eventually spread into native forests, while its similarity in growth-form to Scots pine will make it difficult to eradicate.

Reichard and Hamilton (1997) presented a decision tree for predicting invasions of woody plants introduced into North America. Application to lodgepole pine gives the following results:

- Does the species invade elsewhere, outside of North America? **Yes.**
    - Is it in a family or genus with species that are already strongly invasive in North America? **No.**
        - Is it native to parts of North America other than the region of the proposed introduction? **Yes.**
            - **Further analysis/monitoring needed.**

5.2.3 Ponderosa pine (Pinus ponderosa)

Ponderosa pine is native to dry parts of western North America, including the interior of British Columbia. The ecoclimatic model in Section 5.1 showed that suitable climates for ponderosa pine are found in the extreme southwest corner of the study area . The model implies that it is limited by winter cold, as large parts of the study area are within the suitable range of growing degree-days and moisture balance for this species. The suitable range is predicted to expand under two of the three climate change scenarios considered, mainly in western Alberta.

Ponderosa pine produces seed at 10 to 20 years of age. Seeds are not usually dispersed more than 37 m from the parent tree. Seedling establishment requires exposed mineral soil. Ponderosa pine is an early-seral species, intolerant of shade and requiring disturbances such as fire or logging for regeneration. It may be replaced by more shade-tolerant species such as Douglas-fir or spruce after several decades without disturbance. On sites too dry for other species, ponderosa pine may persist in open, uneven-aged stands. In its native range, ponderosa pine tends to invade adjacent grasslands in the absence of fire. Reduced grazing and high precipitation also favour tree encroachment on grasslands (www.fs.fed.us/database/feis/plants/tree).

In a global survey, Binggeli (1996) listed ponderosa pine as possibly/potentially invasive. Richardson and Rejmanek (2004) listed ponderosa pine as naturalized in Russia, and invasive in Argentina, Australia, Chile, and New Zealand. Their model of invasiveness based on biological characteristics showed ponderosa pine in the middle of the range of pine species examined (i.e. neither highly invasive nor highly non-invasive) (Rejmanek and Richardson 1996).

Reichard and Hamilton (1997) presented a decision tree for predicting invasions of woody plants introduced into North America. Application of this decision tree to ponderosa pine gave the following results:

- Does the species invade elsewhere, outside of North America? **Yes.**

- Is it in a family or genus with species that are already strongly invasive in North America? **No.**
  - Is it native to parts of North America other than the region of the proposed introduction? **Yes.**
    - **Further analysis/monitoring needed.**

5.2.4 Douglas-fir (Pseudotsuga menziesii)

Douglas-fir is native to western North America, including coastal and interior British Columbia, and extending to low-altitude valleys in the Alberta Rockies. Coastal Douglas-fir differs from the variety found in the interior (blue or Rocky Mountain Douglas-fir, *Pseudotsuga menziesii* var. *glauca*) (Hosie 1969).

The ecoclimatic model in Section 5.1 showed suitable climates for Douglas-fir on the southwest edge of the study area. The model implies that it is limited by winter cold, as large parts of the study area are within the suitable range of growing degree-days and moisture balance for this species. The suitable range is predicted to expand under two of the three climate change scenarios considered, mainly in western Alberta.

Douglas-fir produces seed at 12 to 15 years. Most seeds fall within 100 m of their source, but some seeds disperse much farther, and stands have been produced from seed sources 1 to 2 km away. Seedlings establish in mineral soil and thin organic seedbeds. The successional role of Douglas-fir varies among regions. It is an early-seral species in wet forests, but a late-seral species in dry to moist forests, where it tends to replace ponderosa pine, lodgepole pine, western larch, and trembling aspen. It invades grassland, sagebrush-grassland, and meadows in response to reduced fire frequency, climate change, and grazing pressure. It also invades trembling aspen stands with prolonged absence of fire. (www.fs.fed.us/database/feis/plants/tree)

In a global survey, Binggeli (1996) listed Douglas-fir as possibly/potentially invasive. Richardson and Rejmanek (2004) listed Douglas-fir as naturalized in Czech Republic, Germany, Ireland, New Zealand, and New York state, and invasive in Argentina, Austria, Bulgaria, Chile, and Great Britain. Their model of invasiveness based on biological characteristics showed Douglas-fir as one of the most invasive non-pine conifers (Richardson and Rejmanek 2004).

Hermann (1987) reported that Douglas-fir is the most extensively introduced North American tree in Europe, particularly in France, Germany, and Great Britain. Peterken (2001) reported that Douglas-fir planted in Britain regenerates successfully, and implied that there has been some invasion of native pine, birch, and oak woods on acid soils. In Spain, Broncano et al. (2005) found invasion by Douglas-fir up to 100 m into heathland, originating from plantations only 16 to 18 years old. In New Zealand, while lodgepole pine is the most aggressive invader in open areas, the more shade-tolerant Douglas-fir has more ability to invade shrubland and canopy caps in beech forest (Ledgard 2001b). In Argentina, Simberloff et al. (2002) found Douglas-fir to be the most invasive of a large number of introduced tree species, mainly appearing in openings within 1 km of plantations.

Reichard and Hamilton (1997) presented a decision tree for predicting invasions of woody plants introduced into North America. Application of this decision tree to Douglas-fir gave the following results:

- Does the species invade elsewhere, outside of North America? **Yes.**
  - Is it in a family or genus with species that are already strongly invasive in North America? **No.**
    - Is it native to parts of North America other than the region of the proposed introduction? **Yes.**
      - **Further analysis/monitoring needed.**

5.2.5 Red pine (Pinus resinosa)

Red pine is native to eastern Canada and northeastern United States, including the Great Lakes Forest in Ontario, and extends into southeastern Manitoba. The ecoclimatic model in Section 5.1 showed no suitable climates for red pine in the study area. The model implied that it is limited by aridity, as large areas in the southern part of the study area are within the suitable range of growing degree-days and mean temperature of the coldest month for this species. According to the model, the study area will continue to be unsuitable for red pine under future climates. Other eastern species such as eastern white pine, yellow birch, maples, etc. show similar ecoclimatic patterns.

This analysis receives support from research by Flannigan and Woodward (1994), who related the natural range of red pine to growing degree-days and annual precipitation. They concluded that red pine is probably limited on the north by insufficient warmth, but on the southwest by insufficient precipitation. Red pine does not occur naturally where the annual precipitation is less than 580 mm. However, the authors cautioned that the real limiting factor in the west could be increasing frequency of fire or drought. Under a climate change scenario, the range was predicted to expand northward but shrink from the west, becoming even less suitable for western regions. Similarly, Brown et al. (2000) predicted that red pine's natural range would retreat from its westernmost limit in Minnesota. Flannigan and Bergeron (1998) found that red pine at its northern limit in Quebec is probably limited by the crown-fire regime of the boreal forest, rather than by climatic limitations on reproduction. Similarly, Sutton et al. (2002) found that red pine at the northwestern limit of its range in Manitoba is probably not limited by ability to reproduce, because trees at the range limit produced numerous cones and viable seeds.

Red pine plantations have been attempted in Saskatchewan, with some success and some failure. This species is more likely to succeed on sites with good moisture supply (John Thompson, Saskatchewan Environment, personal communication). This is by contrast with the dry sands where red pine often occurs in its native range.

Red pine in open stands produces seed at 15 to 25 years. Large seed crops occur once every 3 to 7 years. Seed dispersal averages only 12 m, but some seeds may be carried up to 275 m. Seedlings establish on mineral soil exposed by fire. Seedlings will establish in partial shade. Red pine is considered to be intolerant of shade, but less so than jack pine, aspen, or white birch. It may be successional to these extreme intolerant species, while it may in turn be replaced by more tolerant species such as eastern white pine, white spruce, or balsam fir. On coarsetextured, infertile sites, red pine may self-regenerate in open, park-like stands. (www.fs.fed.us/database/feis/plants/tree).

Richardson and Rejmanek (2004) did not list red pine in a survey of global occurrence of invasive conifers. Their model of invasiveness based on biological characteristics showed red pine in the middle of the range of pines analyzed (i.e. neither highly invasive nor highly non-invasive (Rejmanek and Richardson 1996).

Reichard and Hamilton (1997) presented a decision tree for predicting invasions of woody plants introduced into North America. Application to red pine gave the following results:

- Does the species invade elsewhere, outside of North America? **No.**
    - Is it an interspecific hybrid with known seed sterility? **No.**
        - Is it native to parts of North America other than the region of the proposed introduction? **Yes.**
            - Does is spread quickly by vegetative means? **No.**
                - **Accept.**

5.2.6 Scots pine (Pinus sylvestris)

Scots pine is native to a wide range of habitats in Eurasia, from western Europe to eastern Siberia. It is common in the boreal forest of Eurasia, and also found in mountains further south (e.g. Spain, Balkans, Turkey).

The ecoclimatic model in Section 5.1 showed the boreal region in the study area as suitable for Scots pine, with a potential distribution similar to jack pine or tamarack. Suitable climates are predicted to shift northward with climate change. Therefore, Scots pine may be no more suited for adaptation to a warmer climate than our native boreal species. However, Scots pine is found in such a wide range of native habitats that there may be provenances that are better suited to dry climates than our native boreal species.

Scots pine produces seed at 10 to 15 years. Good seed crops occur every 3 to 6 years. The cones require alternating wet and dry weather to open. Dispersal is usually within 50 to 100 m of the parent tree. Scots pine is an early-seral species, intolerant of shade, and with low survival under suppression. It usually regenerates in canopy gaps or after stand-replacing disturbance. Scots pine in Sweden is maintained by fire, and is usually replaced by Norway spruce (*Picea abies*) in the absence of fire. However, on some sites, Scots pine maintains its population in uneven-aged stands (www.fs.fed.us/database/feis/plants/tree).

In a global survey, Binggeli (1996) listed Scots pine as highly invasive. Richardson and Rejmanek (2004) listed Scots pine as naturalized in Argentina, Ireland, and eastern United States, and invasive in Ontario, Chile, and New Zealand. Their model of invasiveness based on biological characteristics showed Scots pine as relatively invasive (Rejmanek and Richardson 1996). Scots pine plantations in France (in a region where this species is not native) lead to pine invasion of abandoned grasslands and heathlands (Prevosto et al. 2003). White and Haber (1993) listed Scots pine as an exotic invader in Canada. Mosquin (1997) listed Scots pine as invasive in Canada's parks and protected areas.

In Scots pine plantations in Saskatchewan, abundant seedlings have established in adjacent open areas (e.g. roadsides, cutovers) (J. Thorpe, personal observation). This suggests that spread from the original location is possible in this environment.

Reichard and Hamilton (1997) presented a decision tree for predicting invasions of woody plants introduced into North America. Application to Scots pine gave the following results:

- Does the species invade elsewhere, outside of North America? **Yes.**
    - Is it in a family or genus with species that are already strongly invasive in North America? **No.**
        - Is it native to parts of North America other than the region of the proposed introduction? **No.**
            - **Reject.**

5.2.7 Siberian larch (Larix sibirica)

Siberian larch is native to a huge range of boreal and subarctic forest extending over most of Siberia (Tchebakova et al. 1994).

The ecoclimatic model in Section 5.1 showed the boreal region in the study area as suitable for Siberian larch, with a distribution similar to jack pine or tamarack. Suitable climates are predicted to shift northward with climate change. In our region, Siberian larch is promoted as a shelterbelt tree because it has been found to have high drought tolerance and moderate salinity tolerance, but less tolerance for wet soils than the native tamarack (www.agr.gc.ca/pfra/shelterbelt/shbpub27.htm). It is possible that it may be better suited to the drier future climate than the native species.

Siberian larch is an early-successional, shade-intolerant species. However, no information has been found on invasive behaviour. In Siberian larch plantations in Saskatchewan, seedlings have established in adjacent open areas, so spread from the original location appears to be possible (J. Thorpe, personal observation).

5.2.8 Hybrid poplar (Populus spp.)

Currently in our region, much of the planting of introduced trees has focused on selected clones of hybrid poplar. Most of the poplar clones in use are cottonwoods rather than aspens because they are easily propagated from stem cuttings (Dickmann 2001). Some hybrids are derived from native species. For example, Northwest poplar is a male clone produced by hybridizing balsam poplar (*Populus balsamifera*) and eastern cottonwood (*Populus deltoides*), and is widely used in the northern prairies (Eckenwalder 2001). Others are derived in part from exotic species, such as Walker poplar, a female clone hybridized from eastern cottonwood and European black poplar (*P. nigra*) (Eckenwalder 2001).

Most of the extensive literature on hybrid poplar cultivation focuses on selection of clones and silvicultural methods (e.g. Dickmann et al. 2001), with much less attention to possible negative impacts of plantations. Planting of hybrid poplars in the vicinity of native poplar stands raises the possibility of hybridization with the natives. According to Strauss (1999), the hybrid clones in use will cross readily with native cottonwoods, because they are only one or two generations removed from them. This is essentially invasion by genes rather than by whole species. Braatne (1999) documented several cases where this has happened, but concluded that the influence on native populations has been limited by low viability of seeds from crosses. According to Strauss (1999),

> "Because of the limited size of plantations compared to wild stands in most areas, plantation-derived propagules are usually greatly diluted with propagules from wild stands, including those located a short distance from plantations…Hybrid breakdown and maladaptation are expected to limit the ability of hybrid progeny to invade established areas of wild poplar stands. However, when wild stands are small compared to hybrid plantations, introgression may be observed after long periods of time, as has been detected at low levels among wild stands of *P. nigra* in Europe…"

Braatne (1999) recommended that if hybrids are planted near native stands, preference be given to use of sterile triploids or genetically engineered sterile clones, particularly in rows on the edges of plantations. He also recommended planting of female clones to limit pollen release to native females.

The particular issue of transgenic poplars was discussed by Strauss et al. (2001). For example, planting of herbicide-resistant poplars raises the risk of spread of this trait into native populations. Genetically engineered sterility will reduce this problem, but containment will not be perfect (Strauss et al. 2001).

5.2.9 Conclusions

- Adaptation to climate change: Limited modeling suggests that the drier future climate of the study area is unlikely to be suitable for red pine. Much of the study area except the northernmost parts is likely to be too warm for lodgepole pine. This is probably also true for Scots pine and Siberian larch, but they may be more tolerant of climatic dryness than the native boreal species. Ponderosa pine and Douglas-fir may be suited for the milder western parts of the study area, provided that moisture is adequate (i.e. excluding the grassland region where moisture will be limiting).
- Relative invasiveness based on biological characteristics and behaviour elsewhere (Rejmanek and Richardson 1996, Richardson and Rejmanek 2004) is approximately as follows: lodgepole pine>Scots pine>Douglas-fir>ponderosa pine>red pine. However, the lodgepole planted in the prairies is the interior variety with predominantly serotinous cones, so invasiveness will depend on the frequency of non-serotionous trees in the population, or on the occurrence of fire. There is insufficient information to place Siberian larch in this list.
- Application of the Reichard and Hamilton (1997) decision tree for predicting invasions in North American tree introductions gave the following results:
    - Further analysis/monitoring needed: lodgepole pine, ponderosa pine, and Douglas-fir
    - Accept: red pine
    - Reject: Scots pine

- Most of these conifers are shade-intolerant, early-seral species. Therefore they are unlikely to invade intact forests. However, Douglas-fir is more shade-tolerant than the others (e.g. functioning as a late-seral species in montane pine forests), and has shown some evidence of invading woodlands elsewhere, so is more likely to be capable of invading forests, especially open forests.
- All species could invade open habitats adjacent to plantations (e.g. pastures, roadsides).
- All species could regenerate following disturbance in adjacent forests (e.g. fire, cutting).
- Most seedfall of these conifers occurs in close proximity to parent trees. Long-distance spread is possible in some circumstances, but less likely. This implies that further spread will not occur until the initial generation of invaders reaches reproductive maturity. The age of reproductive maturity ranges from 5-15 years for lodgepole pine to 15-25 years for red pine.
- Therefore invasion is likely to be a slow process for these species, depending on proximity of open habitats and/or disturbance of adjacent forests.
- Another issue that has received less consideration is gene transfer from introduced trees to adjacent native populations. This risk has been considered in the case of widespread planting of hybrid poplars which could cross with native cottonwoods, but it is not clear whether the risk is significant.

6. POLICY ON INTRODUCTION OF TREE SPECIES

6.1 World policy experience

The International Union for Conservation of Nature and Natural Resources (IUCN 1987) set out a position statement that exotic species should only be considered for introduction "...if clear and well defined benefits to man or natural communities can be foreseen..." and "...no native species is considered suitable for the purpose for which the introduction is being made." The IUCN also recommended that no "...exotic species should be deliberately introduced into any natural habitat..." and should be introduced into a semi-natural habitat only when "...there are exceptional reasons for doing so". If the proposed introduction meets these criteria, the next step would be a detailed assessment of benefits and risks, including probability of increase in numbers, probability of invading other habitats, potential for interbreeding with native species, and disease problems. If benefits are found to out-weigh risks, then small, closely monitored field trials should be done. If the species behaves as predicted, then it can be introduced more extensively, but with monitoring and control measures if necessary. The organization introducing the species should bear the cost of control (IUCN 1987).

United States legislation to control the introduction and spread of exotic species is fragmented amongst many federal and state agencies (Ruesink et al. 1995). Plant species are controlled under the federal *Noxious Weed Act 1974* which prohibits the import of 94 listed plants. This legislation tends to be reactive, prohibiting known problem species and allowing other organisms entry. Although in 1978 President Carter signed an executive order directing government agencies "...to prevent the introduction of exotic species into natural ecosystems of the United States...", the US Fish and Wildlife Service at the same time rejected a proposal to switch from prohibiting named species to listing allowable introductions and prohibiting unlisted species (Ruesink et al. 1995). Simberloff et al. (2005) stated that US federal policy "...is far from providing a coherent approach to the intentional introduction of species…Nor does it lay out a conception of ecological place (e.g. by expressing a preference for indigenous species in wildlands)…introductions are assumed 'innocent until proven guilty'". Reichard and Hamilton (1997) also deemed the *Act* "...less than successful in preventing introductions of plant species capable of invading natural ecosystems".

Some US state governments have a stricter approach. Minnesota maintains a list of prohibited or regulated exotic species, a "clean" list of unregulated exotics thought to be of low risk, and an intermediate list. Organisms on this latter list may be owned, sold and transported, but if anyone wishes to release the organism into the wild they must apply for permission and the species must then be evaluated to see if it qualifies for inclusion on the "clean" list (Simberloff et al. 2005).

The California Native Plant Society (1993) advocated that exotic tree species be planted in natural environments only where "...community alteration is being attempted in order to preserve a rare and endangered species that cannot be protected by other means." This means that exotics should be allowed if they are a tool for protection of a favoured native species. The Montana Native Plant Society (2003) in its voluntary guidelines for selecting horticultural material advocated the use of plants native to Montana and surrounding states, and strongly warned against extra-continental exotics, especially those from similar climatic regions (i.e. those likely to be successful).

European Union legislation (Birnie et al. 2004) contains provisions to ensure that exotic introductions do not prejudice local flora and fauna. Nonetheless, European nations have a long history of importing and domesticating many new plant species, including exotic conifers. Britain, for example, has plantations of non-native conifers covering about 6% of its landmass. These conifer plantations have had various justifications which have evolved over time. Originally they were planted to stabilize eroding soils, later as a source of timber to reduce dependency on imports, and later still for aesthetic and diversity reasons.

Karlman (2001) noted that in the early 1980s the Swedish National Board of Forestry specifically recommended planting lodgepole pine "...in areas of northern Sweden where satisfactory regeneration with native conifers was hard to achieve". However, when extreme weather conditions between 1984 and 1987 caused lodgepole plantations to be extensively affected by the fungal pathogen *Gremmeniella abietina*, the Board subsequently restricted the planting program, as Scots pine was less affected than lodgepole. Up to this point, the introduction seemed to have been judged largely on production criteria. However, Elfving et al. (2001) noted that the Board has also restricted the planting of lodgepole in other ways. Planting on harsh sites thought unsuitable for lodgepole in northern Sweden has indeed been banned, but planting south of 60° north latitude is now only allowed on an experimental basis, and lodgepole cannot be planted within one kilometre of a national park or nature reserve boundary (to prevent in-migration). Elving et al. (2001) concluded that the future of lodgepole plantations in Sweden is unclear, and that scenarios from deliberate expansion of intensively bred lodgepole, to the cutting of existing plantations after a short rotation with no renewal of lodgepole allowed, are all possible future management directions.

Rouget et al. (2002) noted that in South Africa recent legislation deals with the risks of invasion via deliberate introductions of commercially valuable exotic tree species (including *Pinus* spp.) by requiring a government permit to plant, by the demarcation of areas open and closed to these exotics, and by putting the legal onus on the planter of the exotic species to prevent its spread onto adjacent land. In effect, the exotic is considered a weed outside the allowed area. This is a similar outcome to lodgepole management in Sweden. Rouget el al. (2002) also suggested that uninvaded areas be classed as either low or high risk for invasion. A high risk area might be a priority conservation area, for example, or an important watershed. If an exotic planting was approved in a high risk area, "...permits should stipulate the need for intensive management to prevent invasions."

Ledgard (2001a) discussed the spread of introduced lodgepole pine in New Zealand, where the *Resource Management Act 1991* aimed to promote sustainable land management. Proponents of land use change must complete what could be called an ecological impact analysis. One of the principal outcomes is that managers of plantations of exotic species are responsible for controlling spread outside the bounds of the permitted planted area. Government has declared lodgepole a noxious weed in some regions of the country. Simberloff et al. (2005) noted that New Zealand's *Biosecurity Act 1993* in effect established a principle of "...guilty until proven innocent..." with regard to proposed introductions.

6.2 Canadian policy

Maxwell et al. (2002) were very critical of the Canadian federal government's lack of effective oversight, knowledge and control of potential and extant exotic invaders, concluding that Canada does not meet its international or national commitments. The federal *Plant Protection Act 1995* proscribes the import to Canada or transport within Canada of various species, in the interests of phytosanitation. The transport of some tree species (such as various apples, cherries, elm, Douglas-fir, firs, hawthorns, junipers, larches, peaches, pears, pines, spruces) from any area designated as infectious can be interdicted, and in practice regulations under the *Act* proscribe many specific translocations. To move named species between named regions, an applicant must apply for a "movement certificate". The general intent is to limit the spread of pathogens. However, one is free to transport uninterdicted species. The *Act* could block the import of an exotic tree species, or, in theory, the movement of a species within Canada, if it was thought that the movement of that species might result in spread of a problem pathogen. In theory, the "New Substances Notification Regulations" under the *Canadian Environmental Protection Act 1999* could also require an extensive risk assessment of a new non-Canadian tree species imported into Canada, but the regulations seem largely directed at inorganics, micro-organisms, and potential impacts on human health, although invasiveness is also a concern. The *Act* would not have effect for trees already in Canada (and most potentially useful exotic introductions into the western boreal will already be present somewhere in Canada). The federal *Seeds Act 1985*, administered by Agriculture Canada, exempts tree seeds from the requirement of variety registration in Canada (Keddy 1993). In summary, there is no known example of an exotic tree whose transport and planting is forbidden by law in Canada. In fact, the planting of exotic tree species is widespread (and often encouraged on freehold land).

All three Prairie Provinces have similar Acts to control the spread of undesired species. Concern originally centred on agricultural objectives, i.e. protecting farm production from weeds, but now also extends to an interest in protecting the natural environment from invasive exotics. In Alberta the relevant Acts are the *Agricultural Pests Act* and the *Weed Control Act*. In Saskatchewan there are the *Noxious Weeds Act* and the *Pest Control Act*. In Manitoba there exist the *Noxious Weeds Act* and the *Plant Pests and Diseases Act*. None of these Acts is directed at trees. The closest to legislated concern about trees is Manitoba's *Dutch Elm Disease Act,* which attempts to control the spread of Dutch Elm Disease, and which is an attempt to preserve a native tree species from an exotic pathogen.

Keddy (1993) specifically considered the applicability of federal and provincial weeds Acts to protect natural habitats from invasive species. She noted that "...provincial Weed Acts were established as agricultural aids for the control of plant species that may detrimentally affect the agricultural use of land or reduce crop values. Typically they address controlling the spread of these weeds from other land to agricultural land and their control on agricultural land…In the Acts, no distinction is made between native and exotic weed species." In general, Keddy (1993) concluded that provincial Acts would need to be modified to have much impact on protecting natural habitats.

Saskatchewan tries to protect native elm trees by means of the *Forest Resource Management Act 1996,* which generally applies only to Crown land, but which has implications for freehold land when a particular species is designated infectious. Elmwood is not allowed to be transported because of the potential for the spread of Dutch Elm Disease. More broadly, under section 24 of the *Act*, "No person, without the written authority of the minister, shall import any thing into Saskatchewan that, in the minister's opinion, could cause the spread of insects or diseases harmful to Saskatchewan's forests, trees or other arboraceous vegetation". The import of exotic trees proper would not appear to be affected by this Act, nor is the risk of hybridization addressed, but certainly an exotic tree species with a known risk of harbouring pathogens dangerous to a new ecosystem could be denied entry under this wording. However, section 63 of the *Act* states that it is the operator or controller's (i.e. the forestry company's) responsibility in general to control insects and diseases that impair timber values, and to destroy infectious materials, which in most cases would be native woods. In practice, it is not clear that section 63 has real effect when damage is widespread (such as from mistletoe).

In terms of Saskatchewan stocking policy, unless specifically allowed within a licencing agreement between the provincial government and the forestry company, reseeding must be done with seed collected in the ecoregion, i.e. from cones collected from a harvested site and then started at a nursery. Trials are underway with red pine, as it is resistant to mistletoe disease. Hybrid poplar is being studied, but must be approved by a scientific advisory board (Darwin Janke, Saskatchewan Environment, pers. comm. 2005).

Manitoba's Forest Renewal Program has as its stated objective for untenured Crown land to ensure that all harvested forests are satisfactorily regenerated to maintain the existing mosaic of forest ecosystem stand types (Manitoba Conservation Department 2005a). This would seem to preclude the introduction of exotic tree species. As well, the Manitoba government is actively involved, in partnership with forestry companies, in tree improvement programs for the three main reforestation species: black spruce, white spruce and jack pine. The objectives are "...to increase the productivity of forest plantations by providing a source of improved planting stock that will result in increased growth, better form and wood quality, and improved insect and disease resistance.... The genetic resource is managed to ensure that genetic diversity is maintained so that the forests are able to adapt to changing conditions in the future." (Manitoba Conservation Department 2005b). The assumptions here are that maintenance of the existing mosaic is possible and that forests will be able to adapt. If this is not possible for native species, it is not clear what the policy options would be.

In Alberta, reforestation is mandatory and legislated under the *Forests Act* and *Timber Management Regulation*, and detailed in the relevant Forest Management Agreements. Forestry companies are obligated to follow the *Alberta Regeneration Survey Manual*, *2000* which contains the standards by which various forest types are to be reforested and the required timing of regeneration surveys and survey methods. Reforestation has been mandatory for over 30 years (Alberta Sustainable Resource Management Department 2005).

Alberta established The Alberta Reforestation Standards Science Council to recommend improvements to reforestation policy from a scientific basis. In 2001, the Council issued a report to government: *Linking Regeneration Standards to Growth and Yield and Forest Management Objectives*. An underlying assumption of the Council was that restocking was to include only native tree species – one of the nine guiding principles the Council adopted for its work was: "Maintaining biodiversity and ecosystem function should be a priority in development of the future forest. In the absence of specific plans, the general goal should be to reproduce the composition and structure of the current forest." A few species (birch, tamarack, Douglas-fir and whitebark pine) were noted as being particularly worthy of retention in future forests simply because they are "less common" (Alberta Reforestation Standards Science Council 2001), but introduction of exotic species as a potential tool to support biodiversity or ecosystem function is nowhere discussed. Nonetheless, within a Detailed Forest Management Plan it is possible for the government and a forest company to agree on significantly different regeneration standards, a process termed "Management by Objectives". The Council noted this offers "...possibilities for enhanced forest level planning, and greater opportunities for adaptive management..." but also poses "...a potential risk to the forest..." if not implemented responsibly (Alberta Reforestation Standards Science Council 2001).

Most recently, Alberta Sustainable Resource Development (2005) issued *Standards for Tree Improvement in Alberta* with respect to the forested provincial Crown lands of the province. This is the most detailed document to deal with reforestation "tree improvement" issues yet written amongst the Prairie Provinces. The document's emphasis is clearly on the preservation of existing biodiversity. For example, the government and industry are to "...ensure the adaptability, diversity and health of wild populations of trees on the landscape". The document also states that use of GMOs "...is not approved for reforestation of Provincial Crown land at this time. Federal legislation controls the testing of GMOs, though Alberta may refuse testing on Crown Land if risks are deemed unacceptable." Yet at the same time, government and industry "...recognize the value of tree improvement in enhancing the productivity of the forest landbase and generating economic benefit."

Potential document conflicts are easily imaginable when Alberta's goals of forest management and reforestation are examined. These goals include conserving "...the genetic integrity, adaptability, diversity and health of wild and managed populations while recognizing that genetic change will occur through evolutionary pressure, breeding and deployment...", but also maintaining or enhancing forest productivity, and being "...consistent with sustainable forest management principles (economic, social and environmental sustainability)". If one was faced with a scenario where native tree species were unable to regenerate to support a sustainable forest, it is not clear where these guidelines would lead. The possible introduction of exotic species is not specifically addressed.

6.3 A different perspective: exotic species and adaptation to climate change

The review of ecological threats in Sections 3 and 4, and the policy review in Sections 6.1 and 6.2, have emphasized the negative aspects of introducing exotic species. In many well-documented cases around the world, introduced exotic plants have been identified as serious threats to biodiversity and other values, usually because of invasive behaviour leading to alteration of neighbouring ecosystems. Recent developments in policy have focused on reducing these threats in order to protect existing ecosystems. However, the increasing evidence for global climate change challenges some of the assumptions on which this discussion has been based. The following presents a different perspective on the issue of introducing exotic tree species.

When considering management of today's western boreal forest, we need perspective. Comparing Holocene environments in the northern Great Plains across millennia, or even across only centuries, indicates a highly variable landscape. It required only relatively small shifts in temperature or precipitation to effect significant landscape and ecosystem change. While anthropogenic climate change and human management are now changing the boreal forest rapidly, the region has a pre-history of frequent change.

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.

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 a laissez-faire wilderness preservation model and adopt increasingly intensive management policies. These could include deliberately assisting the movement of species to newly suitable habitats. This issue is particularly acute for forest systems, where natural migration (for example, of new tree species or of climatically more suitable genetic varieties of locally extant tree species) may not be possible without human intervention.

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) believed that 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[4] into the ecosystem, and keeping our ecological options open, may be more appropriate. Pernetta (1994) believed 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) noted 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."

A failure to incorporate climate change impacts within strategic planning is typical of conservation management throughout North America, well beyond the issue of the western boreal. For example, Montana's forestry policy management strategy for state lands (MDNRC 1996) postulated 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 refer to 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

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[4] 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 western boreal possesses relatively few tree species, there is relatively little redundancy, and it would fall to management to introduce new potential keystone species.

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, preservation of biodiversity 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.

Simply trying to maintain existing ecosystem components may not prove an adequate management response. The United States National Synthesis Team (Joyce et al. 2001) recommended 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). "Connectivity" for trees may 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.

With reference to the region south of the western boreal, Wells (1965) made 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 argued that the key restrictions on tree expansion on the Plains have been its flatness and lack of barriers to frequent fire. He accurately pointed out that some xerophytic junipers can be found in the most arid environments of the southern Plains. Since climate scenarios suggest that the western boreal will trend more Plains-like, i.e. warmer and drier, Wells' findings suggest that we can still have trees in future in the western boreal, assuming reasonable fire control. The question is really one about which species are viable and desirable.

A key concern must be the positive or negative impacts that management could have on the aspen component of the (southern) western boreal. The importance of aspen for the persistence of many other plant and animal species can hardly be over-emphasized. It is at least conceivable that we could, if necessary, substitute a more drought-tolerant pine for jack pine, or possibly a more drought-tolerant spruce for white spruce, and maintain a recognizably familiar ecosystem at some western boreal 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 flora and fauna is directly or indirectly dependent on aspen's presence.

The Canadian National Forest Strategy Coalition (1998) stated that forest "...resilience needs to be maintained so that forest ecosystems can adapt to global disruptions such as extended cycles of climatic change...", but offered no views on what would be acceptable management in the face of climate change. Their current strategy (National Forest Strategy Coalition 2003) went no further, simply noting in an action item that it is necessary to "...develop a better understanding of the effects of climate change and the Kyoto Protocol commitments on the forest ecosystem and incorporate these into forest policy and forest management planning."

Mosquin (1997) noted that human activities can lead to a numbers and density expansion of a native organism, with the result that other native species may disappear as the natural habitat is changed. Human activities can have much greater system impacts than the introduction of an exotic. An abnormal or "exotic" ecosystem can be said to have resulted. White-tailed deer, for example, have greatly increased in numbers and ecological impact in many semi-natural areas because of human alterations to the landscape. One can rationally conclude that range expansion of a species (which would traditionally result in the species being stigmatized as "exotic") may have fewer ecological consequences than a numbers expansion within a species' traditional range (which might not even be recognized as an unnatural event). Climate change will increasingly contribute to both range and density changes.

Mosquin (1997) also noted that one should not simply assume that ecosystem adjustments to an exotic species are only negative, and that "...geographical and ecological circumstances can exist where an invasive exotic organism appears to make a positive contribution to ecosystem function and integrity". Categorization as "exotic" should not mean a species is automatically considered "*organisma non grata*" in a given ecosystem: "It is pointless to continue to be seriously concerned with the exotic species (or populations) that are relatively benign or possibly even advantageous in their new homes."

Mosquin (1997) asserted that, while the problem of exotic invasiveness is sometimes serious in parts of southern Canada, it is "...essentially non-existent in the North." The impact of invasive exotics seems to be "...greatest in subtropical and warm temperate regions of the world." This generalization would seem to be in line with the Swedish experience with lodgepole pine, where the management and policy conclusion at the moment is to continue to allow the planting of this exotic, but only in regions north of 60° north latitude. The difficulties of invasive tree control in South Africa also support this generalization.

When considering the desirability of an exotic within a given ecosystem, Mosquin (1997) would consider "...the potential for causing harm to native species or ecosystems through processes such as hybridization, predation, parasitism, interference with communication among other species, [and] competition". He would also consider whether the species fills an unused habitat niche or provides added food for obviously important native species and functions in the ecosystem. Such a standpoint would allow for the consideration of the introduction of other trees when native species are threatened by climate change and the entire ecosystem is endangered.

Andersen et al. (2004) noted that "...One of the few accepted generalizations about the ecological effects of biological invasions is that the greatest impacts occur when a nonindigenous species performs an entirely novel function in the recipient community." The introduction of tree species into the western boreal could therefore be on the low risk end of introductions, as we would be seeking to find species that replicate as closely as possible the ecological function of extant species in the forest. This is a different rationale than normally used for exotic tree plantation introductions, which typically have a commercial purpose in mind. In the case of the western boreal the main objective would be to minimise ecosystem change, not maximise wood production.

Schwartz (1997) noted that as humans alter environments, whether through anthropogenic climate change or other modifications, it becomes difficult to define whether plant responses and species range changes constitute exotic invasions or not. In fact the whole concept of what is “natural” breaks down. Schwartz (1997) considers that key management criteria are not “exoticness”, but whether a given species contributes to large-scale biodiversity preservation within a complex ecosystem, and whether it causes management problems because of exponential population growth.

Systems which have been described above attempt to measure the invasiveness of potential tree introductions. The assumption underlying these systems is that invasiveness is a bad thing, and that where one has a choice, one should favour an uninvasive species over an invasive one. If we are thinking in plantation terms, where we do not want the introduced species to be successful outside of narrow bounds of control, then this assumption makes sense. But if we are in effect trying to assist the evolution of a tree-dominated “natural” ecosystem from an unsustainable state into a new ecological equilibrium that can once again survive as a forested landscape without constant human manipulation, we would want the introduced tree species to be successful, and in particular, we would want it to regenerate successfully unaided. To do otherwise would doom us to continual replanting (which would be anyways impractical given the scale of the study area) or failure of the introduced species and the entire tree-dependent ecosystem.

Englemark et al. (2001), in their discussion of exotic lodgepole pine plantation management in Sweden, noted that emotional and ethical factors must be taken into account when considering exotic introductions. Writing in terms of plantations (rather than in terms of ecological transformation, as immediately above), they noted a case for restraint, based on the observation that we already interfere to such a large degree in natural processes and environments. To a degree, landscape also impacts national identity, and large-scale landscape modifications may impair that identity. To limit the impact of lodgepole pine plantations, they therefore recommended the following management techniques that could also be relevant for exotic introductions in the western boreal should we not wish introductions to spread: concentrate introductions in more controllable areas (to restrict unwanted spread); define exotic-free zones; define maximum extent of introductions; monitor. The Swedish experience, being also boreal, is particularly relevant for the western boreal.

Value judgements are at play in evaluating the pros and cons of a deliberate introduction and “...the perceived value of an introduction depends on which interest group does the accounting” (Ruesink et al. 1995). Simberloff et al. (2005) noted that “...various stakeholders [will] dispute whether the harm caused by an introduced species will outweigh its benefits”.

The remarkable values divide in species introduction policy in the Prairie Provinces centres on the question of land ownership. The southern edge of the boreal forest in this region has been cleared for agriculture and is largely in private hands. If land is privately held, it is acceptable to plant whatever tree species you like on that land. Indeed, there is a long history of federal and provincial government support for the planting of exotic tree species by farmers, largely in an effort to reduce soil erosion and increase crop yields. Over its roughly century-long existence, the PFRA Shelterbelt Centre near Indian Head, Saskatchewan, has supplied many millions of trees to this end across the prairies. By far the most widely planted tree has been the caragana, an Asian-origin exotic. As discussed above, the only significant limiting factor on a private landholder's planting choices are the obligations under the various noxious weeds acts. Private woodlots exist, and there is no legal constraint on planting exotic species on private lands or on the timing or nature of harvest operations.

It is a different story when we consider an existing forested landscape vested in public hands. Outside of the area of agricultural settlement, the western boreal forest is largely in the hands of the respective provincial governments, with significant portions assigned to forest management agreements (FMAs) with private forest companies. Without exception, these FMAs specify that the harvesting forest company has a responsibility to reforest harvested portions of their management area, and that restocking must be done with tree species previously extant on the harvested land base (it is not usually required, however, that the proportions of preharvest tree species be replicated in restocking efforts).

The explanation for this dichotomy of thought may partly be found in western society's urge to put as few limitations as possible on what may be done with private property, and partly in North American society's tendency to put the burden and responsibility for nature conservation onto public lands so far as possible, thereby relieving the private landholder of any such responsibilities and freeing him or her to pursue economic gain or other personal objectives (Henderson 1992). Land ownership is decisive in shifting both responsibility for, and attitudes toward, conservation.

For example, shelterbelts have been long used and encouraged for agricultural soil and moisture management, as incidental tools to the goal of grain and oilseeds crop production. But most recently governments have begun to get interested in the potential for trees to be a crop in and of themselves. Agroforestry is being heavily promoted in Saskatchewan by the provincial government, which in its most recent Throne Speech (November 2005) set as a goal the conversion of 10% of the province's arable land base to trees within 20 years (Government of Saskatchewan 2005). These trees would be destined either for pulp and paper, or for higher value wood products. The plantings would occur almost entirely on freehold lands. The exoticness of these plantings will vary by degree. Some tree plantings can be expected to be of existing native North American species, but Eurasian plantations are also possible, while the greatest bulk is foreseen to be fast-growing hybrid poplars (of both North American and Eurasian origin), developed at research centres such as the PFRA Shelterbelt Centre. Most land used for agroforestry is expected to be along the forest fringe, i.e. the area that naturally supported boreal forest but has been cleared for agriculture. One can conclude that one Prairie government at least is committed to introducing exotics on a large scale, in a broad sweep along the southern edge of the boreal forest, and primarily on freehold, not Crown, land (although plantations on Crown agricultural leaseland remain a possibility).

It is possible, even likely, that there will be contamination of the southern boreal forest if large numbers of exotic trees are planted nearby (dependent on many factors, such as hybridization potential, fecundity, distance, soils and wind). Certainly, caragana has been known to spread into adjacent woodlands. However, there has been no significant public concern with regard to the Saskatchewan government's agroforestry plan on ecological grounds, even though, if the plan is successful, it implies a major landscape change. This is probably because trees in general are thought of as an "improvement" in predominantly agricultural landscapes, and the trees are understood to be destined for freehold farmland.

While not discussed in any detail in this report, if we accept the logic or need to introduce exotic tree species into the western boreal for ecosystem reasons, we also need to consider the practicality and desirability of introducing midstory and understory exotics together with exotic trees. This would certainly be contentious. Many would reluctantly accept the introduction of an exotic tree to the western boreal if it can be demonstrated that other extant boreal flora and fauna could prosper better with this exotic than without, i.e. if the introduced tree can stand in for a similar tree species that is becoming climatically threatened. To a degree, though, this is wishful thinking. If some tree species are becoming less and less likely to regenerate, it will also be true that some understory and midstory vegetation will be facing similar challenges, and that mid and understory vegetation from other regions may be better suited to the changing climate. There may therefore be a case for introducing a mid- and understory as well. Frelich and Puettmann (1999), writing in the context of restoration ecology, noted that "...it is often assumed that understory vegetation will establish over time ('plant trees and the rest will come'), but natural invasion may not automatically bring back all species desired." However, if we know little about exotic tree introductions in the western boreal, we know next to nothing about mid- and understory introductions. Any past research focus has been on trees (probably because of their ecological keystone status and owing to their economic importance as a source of fibre).

What is under consideration here is not the established science of restoration ecology, but more accurately "creation ecology", as there are no antecedent landscapes that we are targeting. In fact, the future ecosystem that results from rapid climate change in the western boreal is likely to be unprecedented in ecological history, whether exotics are introduced or not (just as certain ecological systems present thousands of years ago in North America have entirely vanished).

As a policy conclusion, we note that there do not seem to be legal prohibitions against the introduction of exotic tree species into the western boreal. Exotics can be and are frequently planted on freehold land. Public policy and regulations are against exotic trees in Crown lands in the western boreal. Yet some experimentation seems to be going on, and breeding for genetically superior varieties of native species is encouraged. There also appears to be room within individual FMAs, without legislative change, for the government and the relevant leaseholder to replant harvested areas with exotic tree species, should both parties agree. However, this would be a radical departure from current practice and would not be practical without extended and open discussion with many interested stakeholders.

6.4 Policy conclusions

The following conclusions about future policy needs were developed by integrating the results of the scientific literature review (Sections 2, 3, and 4), the policy review (Section 6.1, 6.2, and 6.3), and the Stakeholders Workshop (Appendix 3).

Policies on introduction of exotic tree species should not be seen as absolute, but rather as dependent on the species and the situation. The feedback we received from stakeholders supported the view that exotic introduction is acceptable in some situations, but not in all situations. A policy that planting exotic species is acceptable in all situations would ignore the extensive evidence of ecological damage done by invasive exotics, and potential conflicts with other land-use or ecological objectives. A policy that planting exotic species is not acceptable in any situation would ignore the usefulness of many species in horticulture, shelterbelts, and agroforestry, and also ignore the increasing evidence that new species will be needed to adapt to climate change.

One consideration in deciding on policies is the type of land tenure and use. The consensus of our Stakeholders Workshop was that policy should differ among protected areas, crown forest land, and privately owned land. In protected areas that serve as ecological benchmarks, it was thought that exotic plantations should not normally be approved, except in special cases (e.g. reclamation of contaminated sites). On provincial forest land, exotic plantations may be acceptable in some situations, but would require a high level of assessment and planning prior to approval. On private land such as farms, the requirement for assessment and planning is less. However there is still an onus on government to assess the species used for widespread planting and prohibit the distribution of problem species.

Another consideration in deciding on policies is the biological attributes of the exotic species of interest. Each species proposed for widespread planting should be subject to a standardized assessment process. The assessment could make use of tools such as the decision tree for screening exotic tree species developed by Reichard and Hamilton (1997). However, the assessment should not rely totally on such tools, but should be expanded to include a comprehensive review of the biology and management experience for the species. The International Union for the Conservation of Nature has provided a template for such assessments (IUCN 1987) which provides a good starting point, to which additional attributes can be added depending on the local situation. The end point of such an assessment is a comparison of the potential benefits of an introduction with its ecological and economic risks.

World experience has shown that one of the most critical risks from introduction of extoic species is invasion of adjacent ecosystems. Rating the potential invasiveness of the individual species should be an important consideration in assessment. Species that are non-reproductive in the planting environment are obviously safest. However, there may be sufficient reasons to consider species that are capable of naturalizing in the new environment (e.g. allowing silvicultural systems based on natural regeneration; contributing to adaptation to climate change). However, species with the kind of aggressive reproduction that leads to negative impacts on ecosystem resilience and diversity are not acceptable. Some exotic species show a strong tendency to form dense, single-species stands, crowding out most of the native plants. The usefulness of an exotic species, whether in timber production, forage production, erosion control, or adaptation to climate change, should never justify the use of such highly invasive species.

The continental origin of exotic trees should not automatically qualify or disqualify a species, because a noninvasive Eurasian tree may be more acceptable than an invasive North American tree. However, it should be a consideration in the assessment, because most cases of severe invasiveness have involved extracontinental introductions.

Value in adapting to climate change should be an important consideration in the assessment. One component of the assessment is evaluation of the benefits of the proposed introduction. Benefits have traditionally related to immediate utility values, such as timber production, forage production, or erosion control. However, there is increasing evidence that climate change will make the environment unsuitable for our current native trees. If society wishes to maintain forests in some of their current range, it may be useful to intentionally introduce varieties or species of trees that are adapted to the new climate.

For species that pass the initial assessment, IUCN (1987) recommended that the next step be limited planting trials, with appropriate monitoring and evaluation, prior to widespread planting. Such planting trials should allow control measures in the event that invasion or spread of disease become apparent. One of the goals of these trials should be evaluating the usefulness of the species in adaptation to climate change. Priority for such trials should be given to provenances of native species from warmer/drier climates than the planting site, and species that are exotic to the planting site but native to warmer/drier parts of North America.

If use of exotic trees is supported by the assessment process, then guidelines should be developed for planning of plantations. Approval of plantations could be made conditional on adherence to such guidelines. Methods that have been recommended in other regions include:

- defining maximum extent or proportion of land in exotic plantations.
- defining exotic-free zones (e.g. protected areas).
- avoiding large concentrations of a single exotic species within a given region, which could lead to swamping of native vegetation by exotic propagules.
- concentrating exotic plantations in more controllable areas (e.g. farmland versus provincial forest; or island forests).
- isolation of plantations by cultivated fields, etc. to reduce risk of invading native vegetation.
- minimizing boundary areas by planning fewer, larger plantations rather than many smaller ones.
- avoiding planting on dispersal-prone sites such as ridges.
- avoiding planting in habitats that are particularly susceptible to invasion, such as stream floodplains.
- avoiding planting upwind of sensitive areas such as parks.
- planting outer rows of less invasive species.
- monitoring of plantations to detect any problems.

In the case of species and situations where widespread planting has already happened, assessment should still take place, making use of the experience that has already been gained with the behaviour of this species in the western boreal environment. Priority for such assessments should be given to suspected problem species, such as caragana in the forest fringe. A negative assessment could lead to prohibiting further distribution of the species or to control programs.

6.5 Policy recommendations

The western boreal forest should be monitored carefully for signs of systematic decline of native tree species arising from climate change or other factors, as this information is crucial to evaluating the need for introduction of exotics.

Replanting of harvested sites with seed stock of natives trees from sources distant from the planting site should be allowed, if distant seed sources are better adapted than local sources to the changing climate.

Policies for introduction of exotic trees should vary according to land ownership and land-use/ecological objectives. For example:

- in protected areas that serve as ecological benchmarks, exotic plantations should not normally be approved, except in special cases (e.g. reclamation of contaminated sites or scientific research)
- on crown forest land, exotic plantations may be acceptable in some situations, but require a high level of assessment and planning prior to approval
- on privately owned land, government should assess exotic introductions and prohibit the distribution of problem species

Individual exotic tree species should be subject to a standardized assessment process. **The assessment should include the following** (as recommended by the International Union for the Conservation of Nature (IUCN 1987)):

- a study of factors which limit distribution and abundance in native range
- an estimation of the probability the species will so increase in numbers as to cause environmental damage
- an estimation of the probability of invasion of habitats beyond the desired introduction zone
- a study of all phases of the relevant biological and climatic cycle
- an estimation of the risk of interbreeding with native species
- a study of the risk of introduction of diseases and parasites the exotic may host
- an evaluation of the threat to native species
- a presumption against the introduction of an exotic for which a control does not exist
- a summary of benefits and risks

In addition the assessment should include:

- an analysis of the net effects on timber supply
- an analysis of the net effects on carbon sequestration
- an analysis of the effect on the fire regime
- an evaluation of the degree to which the proposed introduction might compensate for the decline of keystone native species, and thereby contribute to ecosystem diversity and resiliency
- an analysis of the potential contribution of the introduction to ecosystem diversity and resiliency in the face of climate change

Controlled planting trials, with appropriate monitoring and evaluation, should precede widespread planting:

- planting trials should include control measures in the event of invasion or spread of disease into adjacent ecosystems.
- priority for trials should be given to provenances of native species from warmer/drier climates than the planting site, and to exotics native to warmer/drier parts of North America.

Assessment of previous introductions should take place; priority should be given to suspected problem species (such as caragana in the forest fringe) and to trial plantings of exotics from past decades (to examine survivability and invasiveness)

Using the results of the above assessments, governments should regulate which exotic tree species are acceptable for widespread planting, and the conditions or guidelines under which such planting could occur.

Government should develop guidelines for the location and design of exotic plantations, aimed at minimizing invasion risks.

As with other developments, widespread planting of an exotic tree species should be preceded by and subject to an environmental assessment, including public and stakeholder consultation.

Federal and provincial governments should jointly review their current policies related to introduction of exotic tree species, to determine whether new legislation or regulations are needed, and to avoid duplication.

There should be communication to stakeholders and the public about the challenges of a changing climate to the western boreal ecosystem, and options for adaptation. Governments should articulate their current policies regarding introduction of exotic species.

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APPENDIX 1: DEFINITIONS

Native species are species occurring in their natural range; i.e. their presence is not due to intentional or accidental human introduction. The terms "**exotic**", "**alien**" and "**non-native**" are all used to refer to species that are not native to a given area (i.e. occurring outside of their natural range). According to Mosquin (1997), "alien" was used by the 1992 United Nations Convention on Biological Diversity and is now the preferred term. However, "exotic" is more familiar to many ecologists and resource managers.

There is variation in the degree to which a species is exotic to a given area. For example, Peterken (2001) reviewed the long history of tree and shrub introductions in Britain. Some introductions were transfers from other continents while others were range expansions from continental Europe (some of these had been present in Britain in previous interglacial periods). Some exotic trees and shrubs represented new genera for Britain, some were new species in genera already present, some were new genotypes of species already present, while some just involved movement of genetic stock from one part of Britain to another. For the current study, it is useful to distinguish the following types of exotic plants (Mosquin 1997):

- **Locally exotic plants** – plants exotic to the region under consideration (e.g. a given province or ecoregion) but native to other parts of North America.
- **Plants exotic to North America**—exotic plants that were introduced to North America from other continents (usually Eurasia).

Exotic or alien plant species vary in their behavior and impacts. Various definitions have been proposed (Swarbrick 1991, White and Haber 1993, California Native Plant Society 1996, Williamson and Fitter 1996a, 1996b, Schwartz 1997, Mosquin 1997, ANPC 2000, Richardson et al. 2000, Haysom and Murphy 2003). The most important distinctions are among "exotic", "naturalized", and "invasive".

- **Naturalized plants** are exotics that reproduce and sustain populations in their new environment (Richardson et al. 2000, Haysom and Murphy 2003). This is by distinction with exotics that are introduced by humans but fail to reproduce, so that continued introduction is needed to maintain the species.
- **Invasive exotic plants** are naturalized plants that reproduce at a distance from parent plants and thus have the potential to spread over considerable areas (Richardson et al. 2000). This is by distinction with naturalized species that only reproduce in the area where they were originally introduced.

Invasive plants may be distinguished according to the types of habitats that are invaded. Many agricultural weeds pose little threat to our forests or grasslands because they require soil disturbance for establishment. Pysek et al. (1995) distinguished:

- **Invasive plants in seminatural habitats**—species that invade communities such as grassland, shrubland or forest.
- **Invasive plants in man-made habitats**—species that only invade highly disturbed habitats such as cultivated fields, road-sides, urban areas, etc.

"Weeds" are plants growing in sites where they are not wanted, and that usually have detectable economic or environmental effects (synonyms: plant pests, harmful species, problem plants)

(Richardson et al. 2000). The term "**environmental weeds**" is sometimes used to refer to weeds affecting natural vegetation (Richardson et al. 2000).

Some definitions of "**invasive**" are broader, encompassing not only spread from the location of introduction, but also invasion of natural habitats and impact on them. The following definitions were cited by Haysom and Murphy (2003):

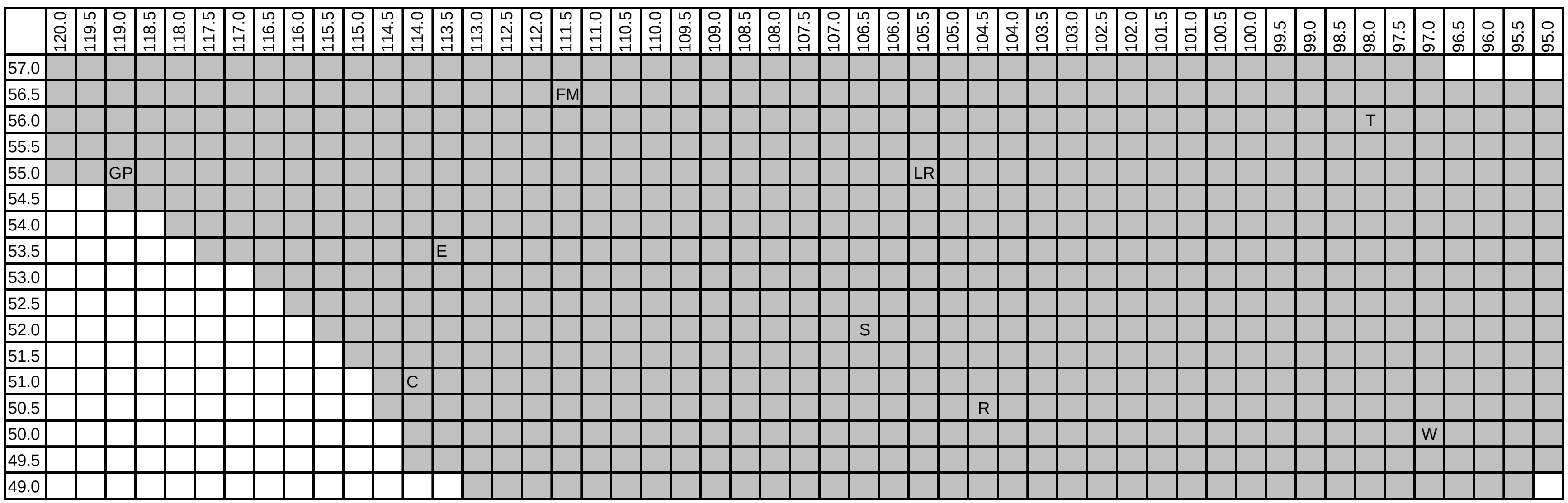
- "an alien plant spreading naturally…in natural or semi-natural habitats, to produce a significant change in terms of composition, structure or ecosystem processes" (Cronk and Fuller 1995, cited by Haysom and Murphy 2003).
- "an alien species that becomes established in natural or semi-natural ecosystems or habitat, is an agent of change and threatens native biological diversity" (Shine et al. 2000, cited by Haysom and Murphy 2003).
- "species introduced deliberately or unintentionally outside their natural habitats, where they have the ability to establish themselves, invade, out-compete natives and take over the new environment (CBD 2001, cited by Haysom and Murphy 2003).

While the term "invasive" often implies "exotic", as in the above definitions, it is important to specify whether invasive species are exotic. There are also native species that show invasive behaviour (for example, native tree species invading adjacent grassland).

APPENDIX 2. MODEL OUTPUTS FOR SUITABILITY OF PRESENT AND FUTURE CLIMATES IN THE PRAIRIE PROVINCES FOR SELECTED TREE SPECIES.

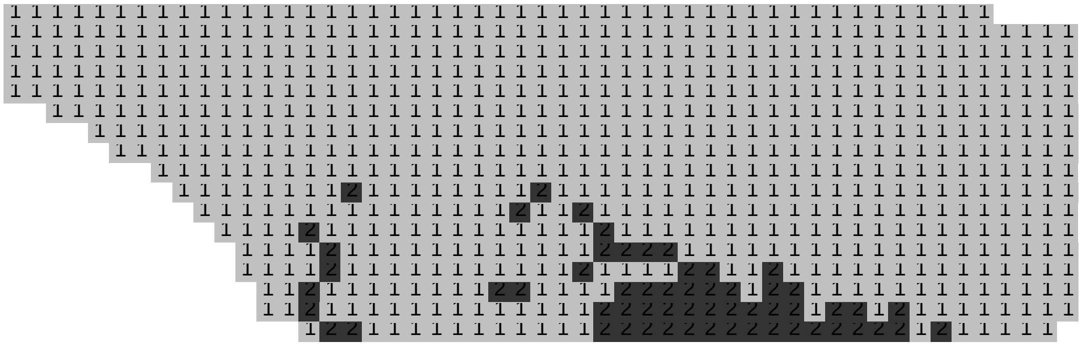
Suitability was determined by comparing derived climatic variables (growing degree-days, mean temperature of the coldest month, ratio of actual to potential evapotranspiration) at a grid of points to upper and lower thresholds for each species given by Thompson et al. (2000a, 2000b). Limits for Eurasian species were interpreted from thresholds for vegetation types given by Tchebakova et al. (1994). Details of methods are given in Section?. Suitability was determined for the present climate (1961-90 normals) and for three scenarios for the 2050s.

In this Appendix, results are presented on a coarse grid at half-degree intervals of latitude and longitude. The following map shows the study area mapped on this grid, with locations of selected cities for reference (GP – Grande Prairie, FM – Fort MacMurray, E – Edmonton, C – Calgary, LR – LaRonge, S – Saskatoon, R – Regina, T – Thompson, W – Winnipeg). In the range maps, dark shading indicates suitable areas, light shading indicates unsuitable areas, and blank cells are outside of the study area.



Manitoba maple (*Acer negundo*)

1961-90 normals



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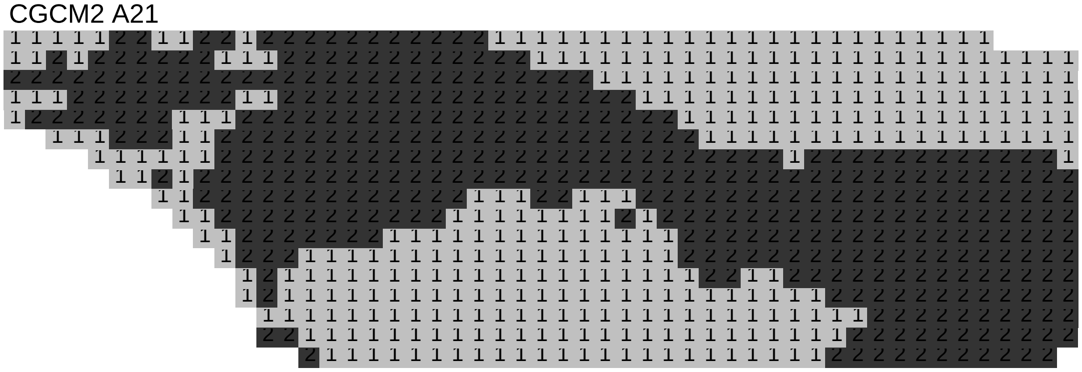

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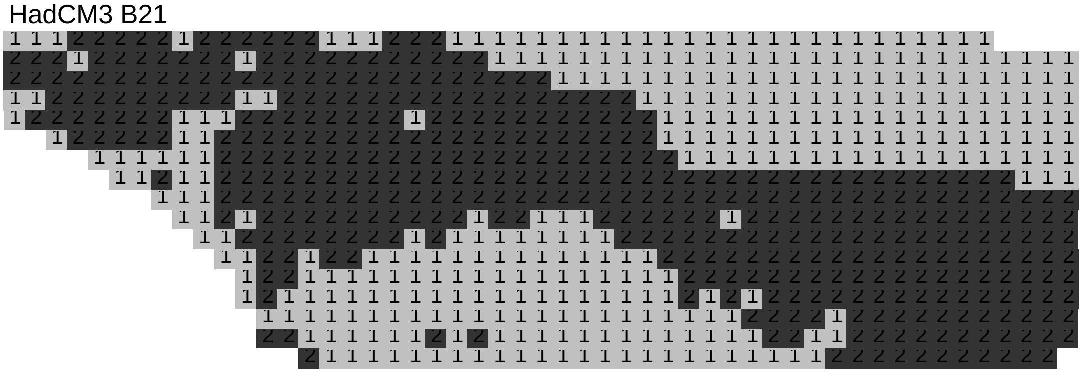

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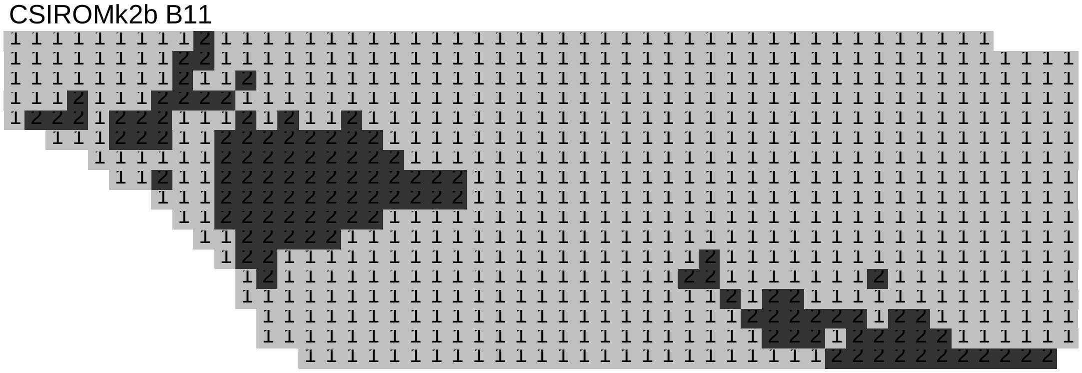
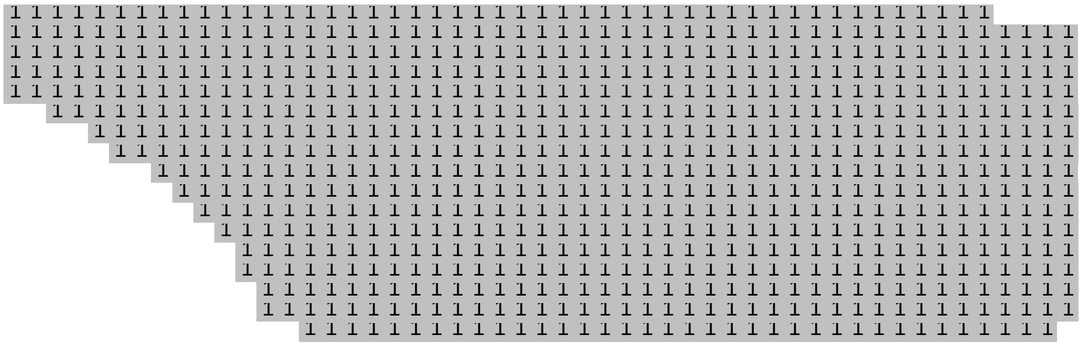


Red maple (*Acer rubrum*)

1961-90 normals



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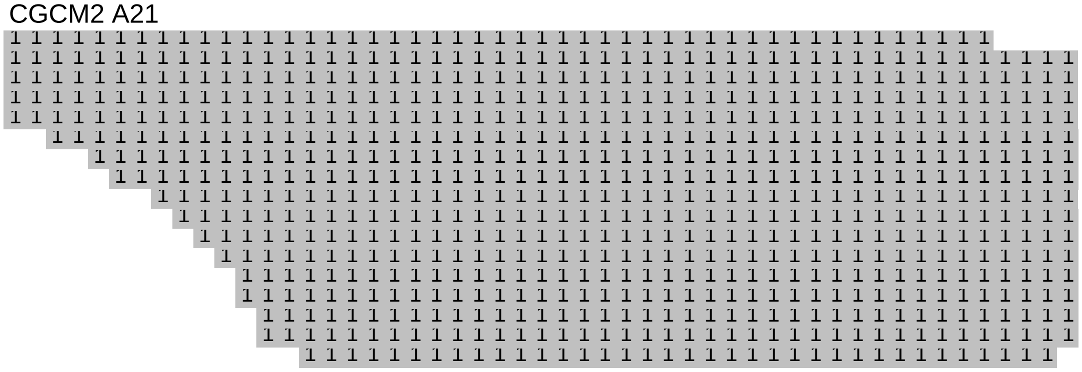

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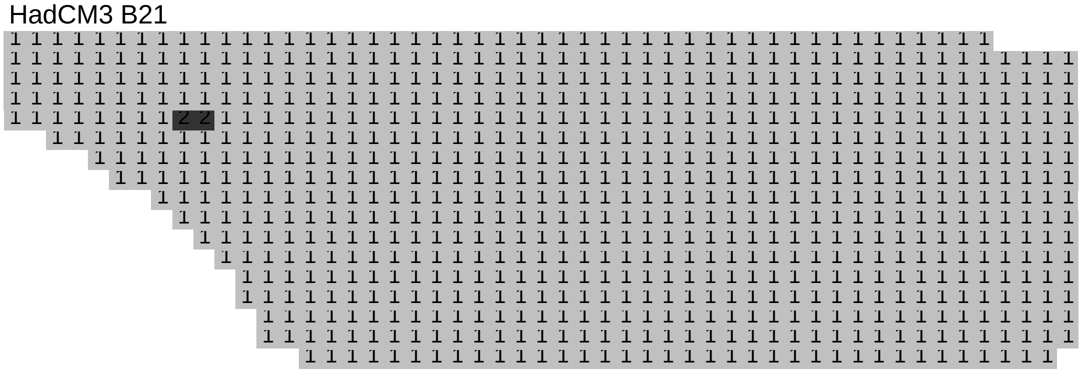

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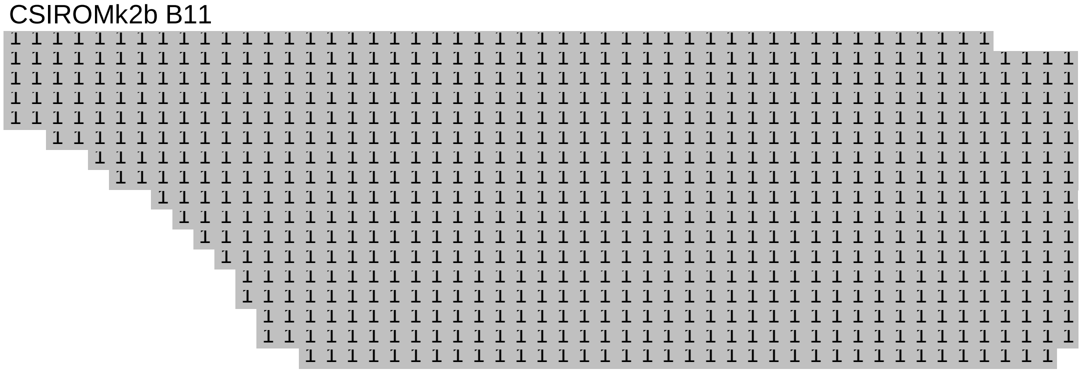
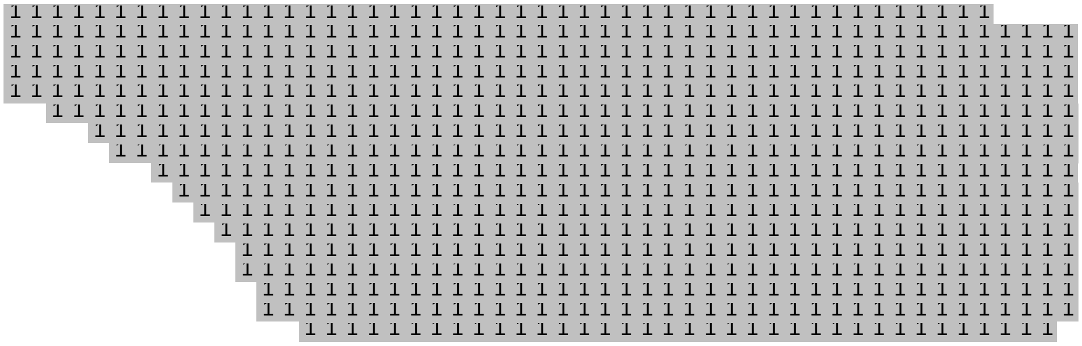


Sugar maple (*Acer saccharum*)

1961-90 normals



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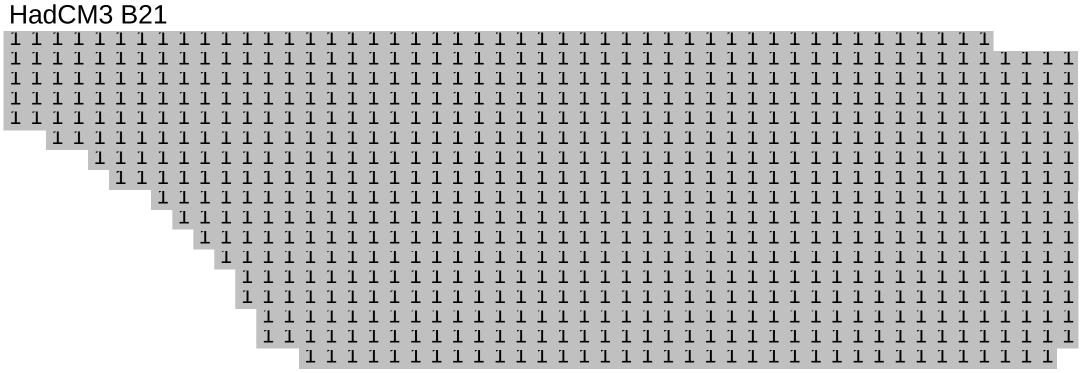

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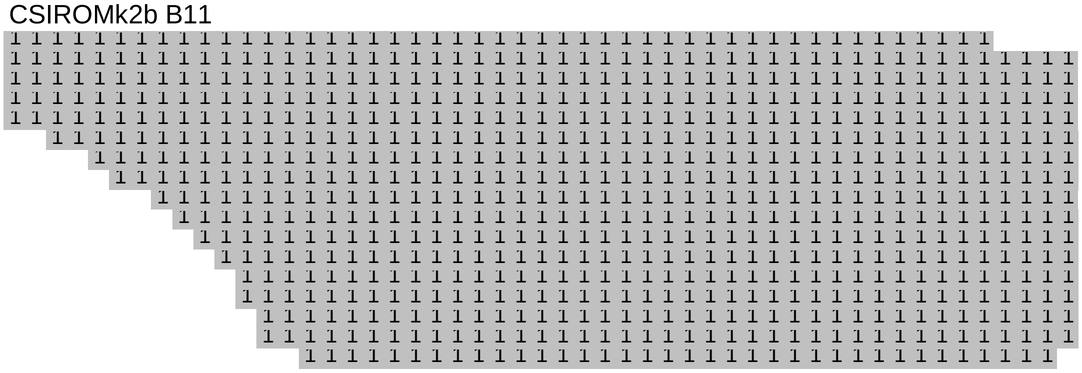
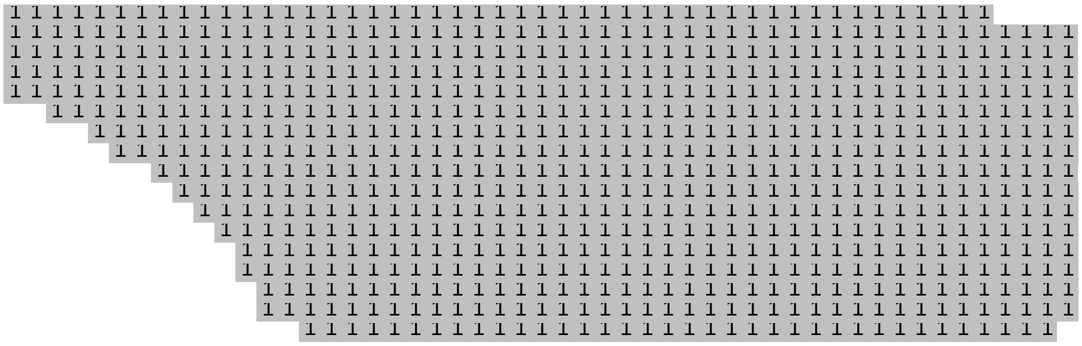


Yellow birch (*Betula alleghaniensis*)

1961-90 normals



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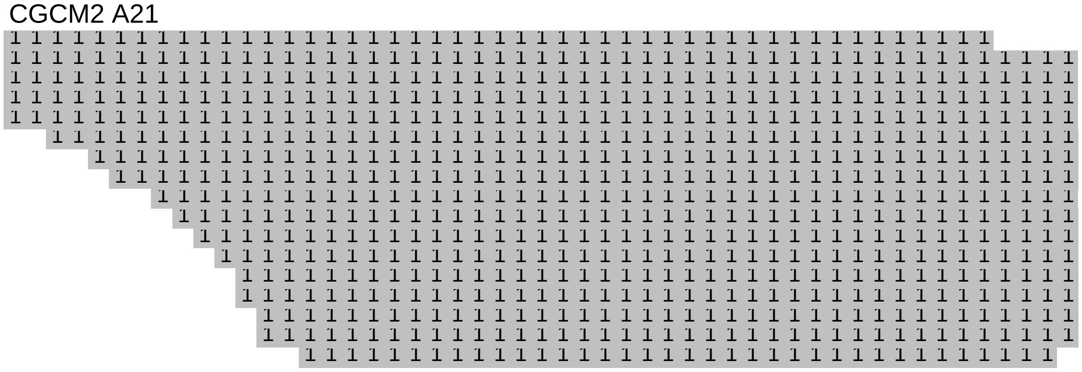

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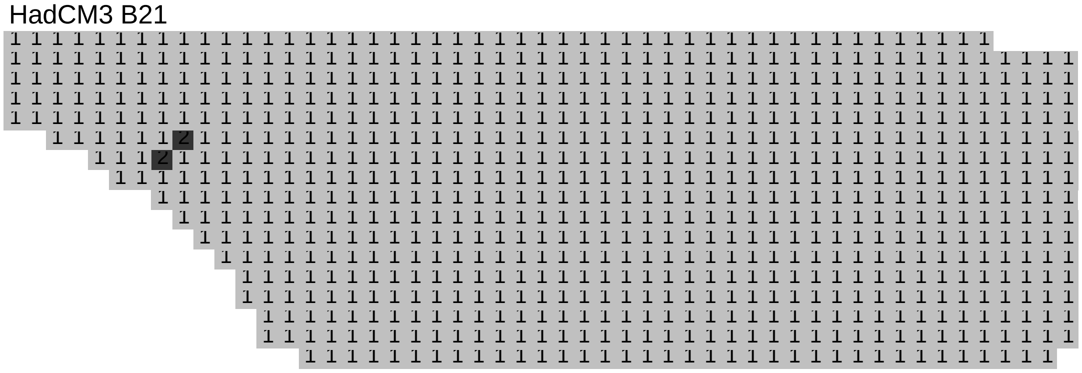

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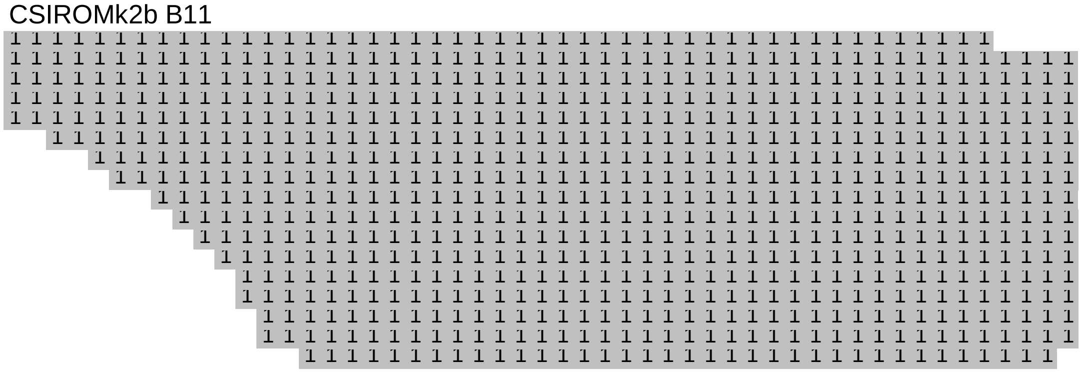
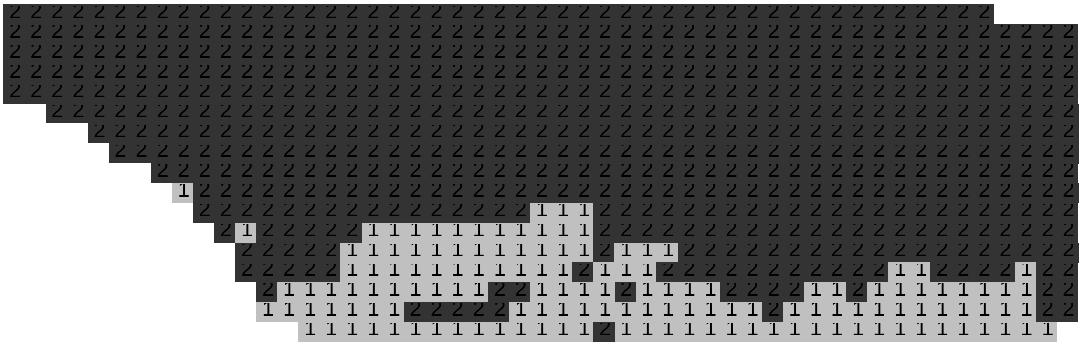


White birch (*Betula papyrifera*)

1961-90 normals



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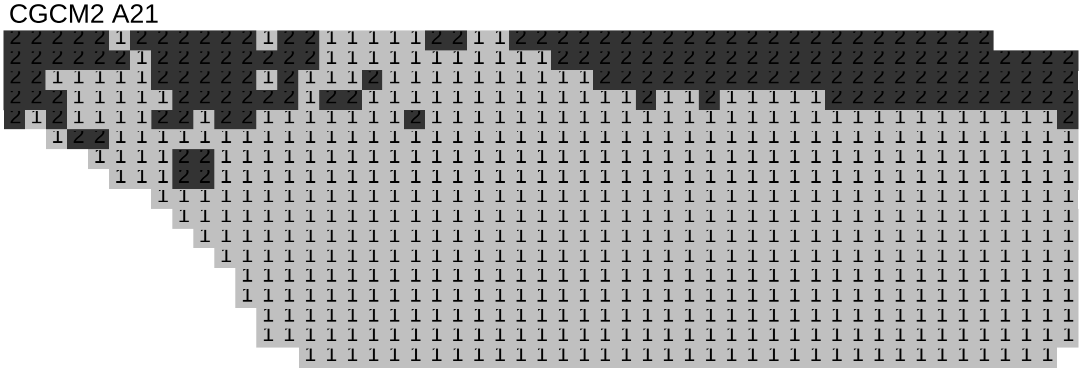

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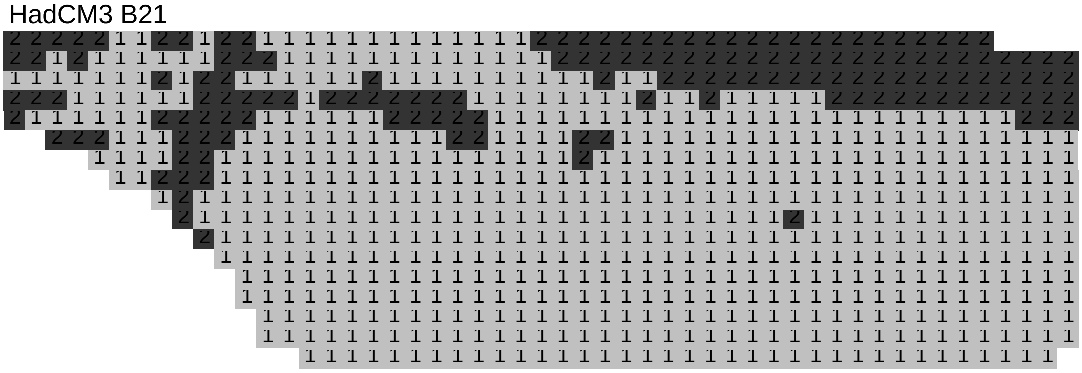

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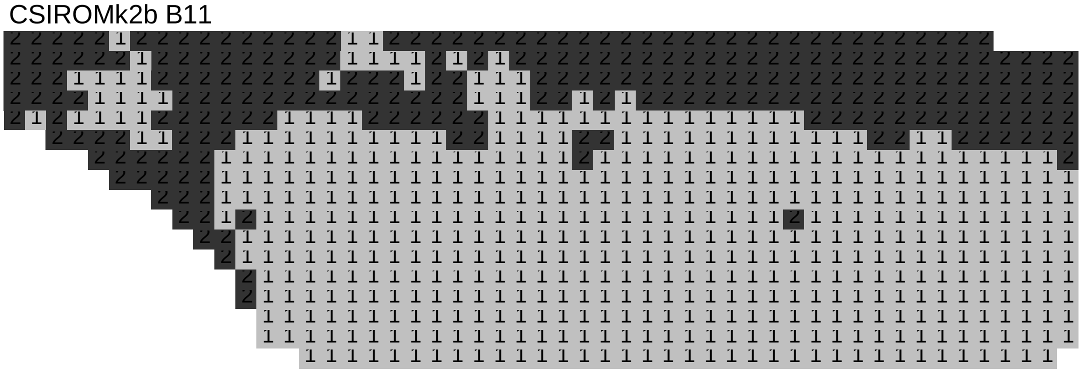

Green ash (*Fraxinus pensylvanica*)

1961-90 normals

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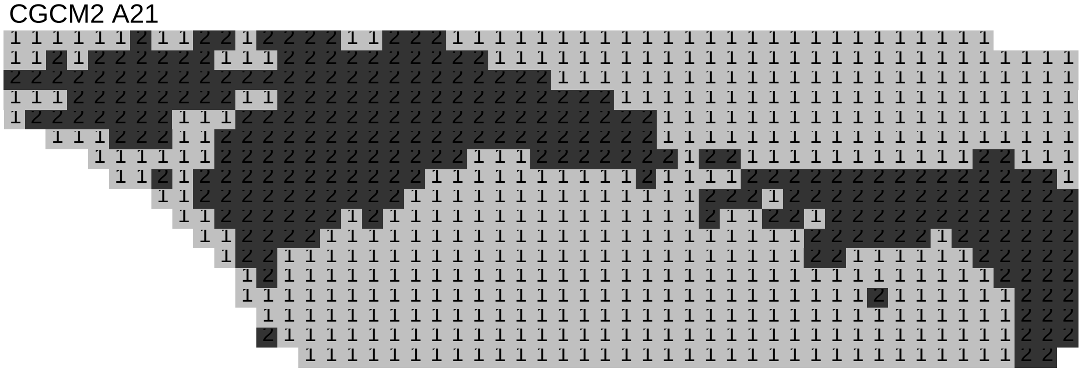

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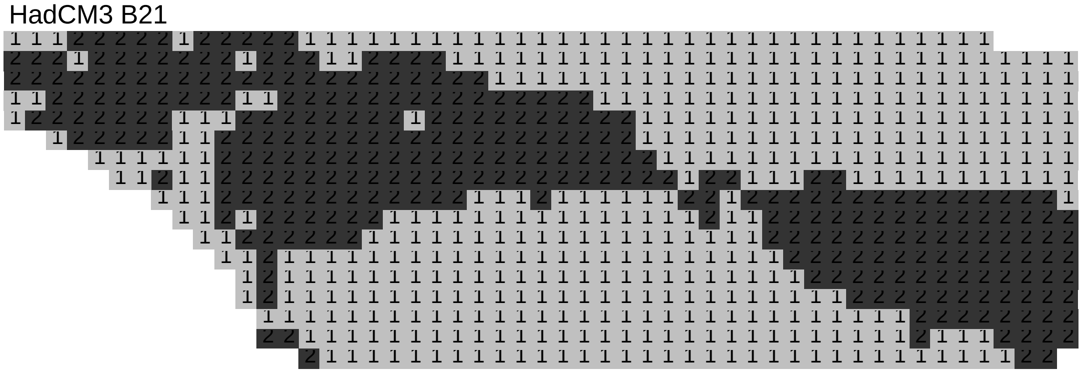

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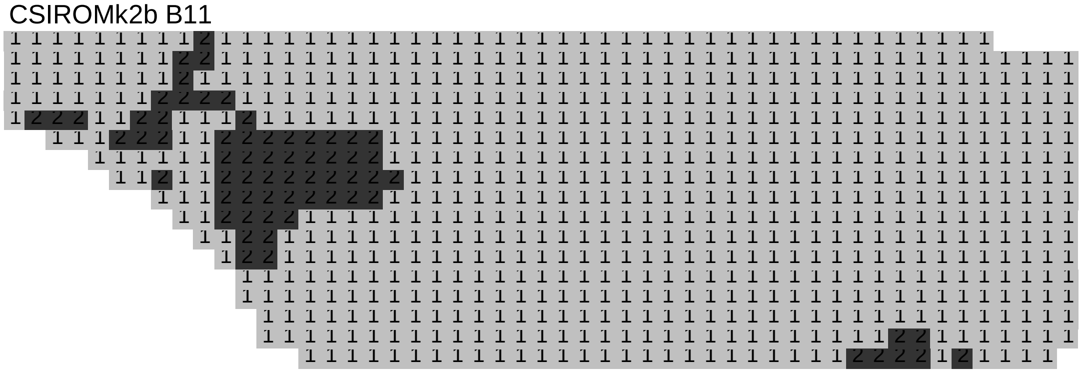
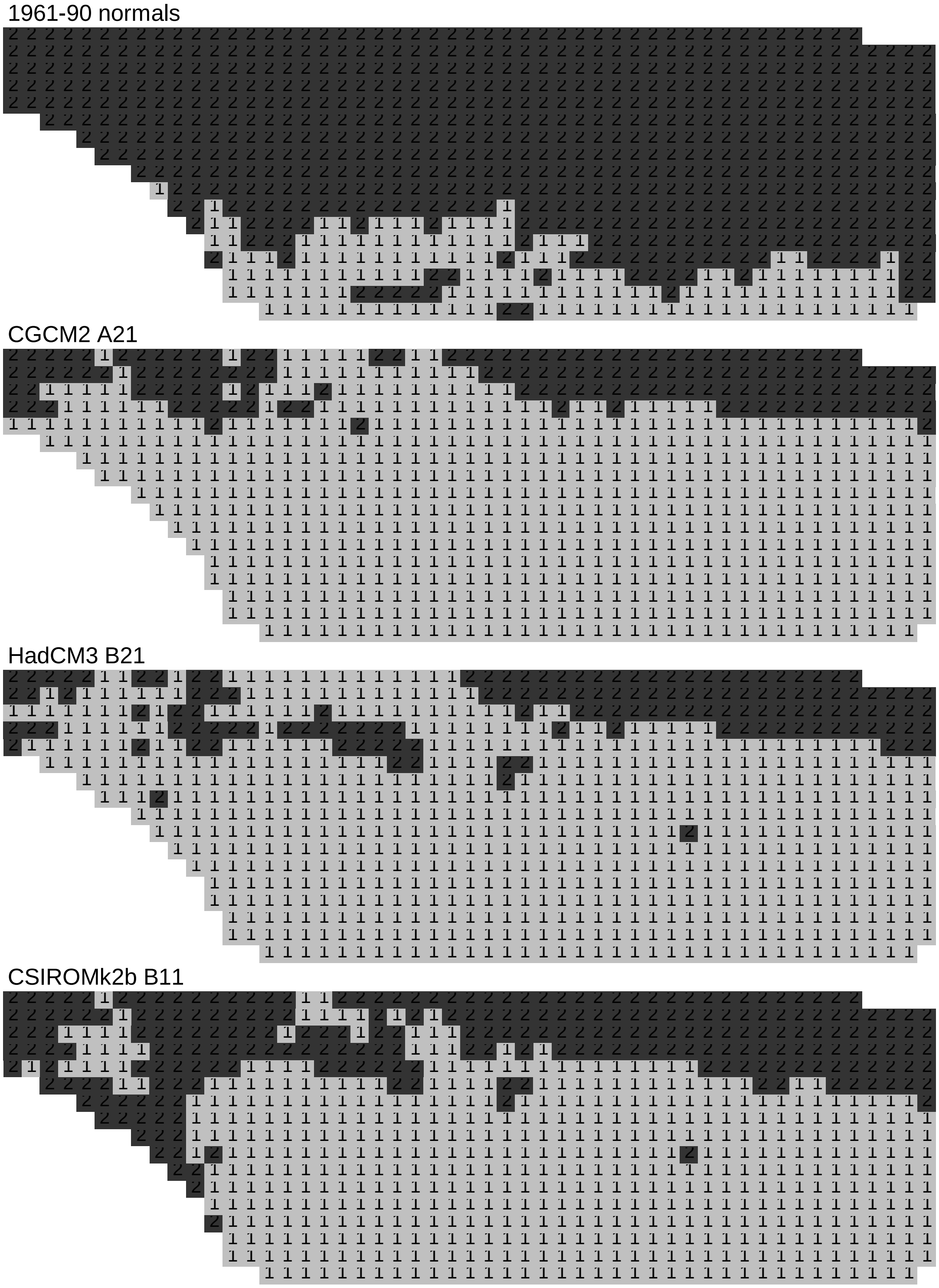
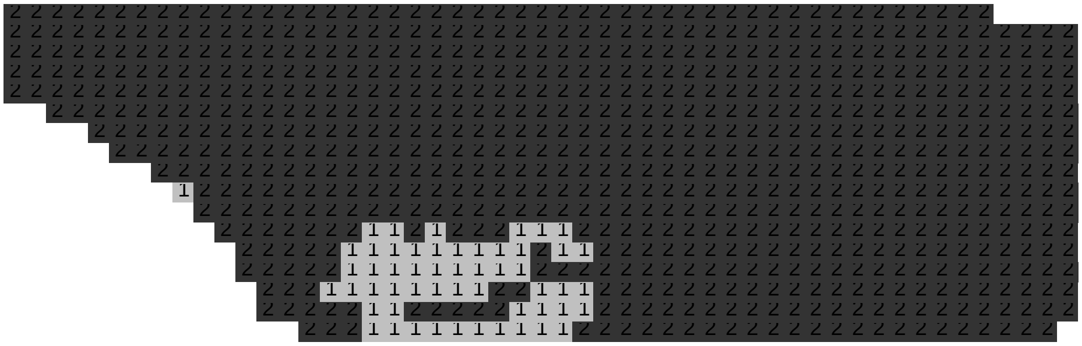


Balsam poplar (*Populus balsamifera*)



Trembling aspen (*Populus tremuloides*)

1961-90 normals



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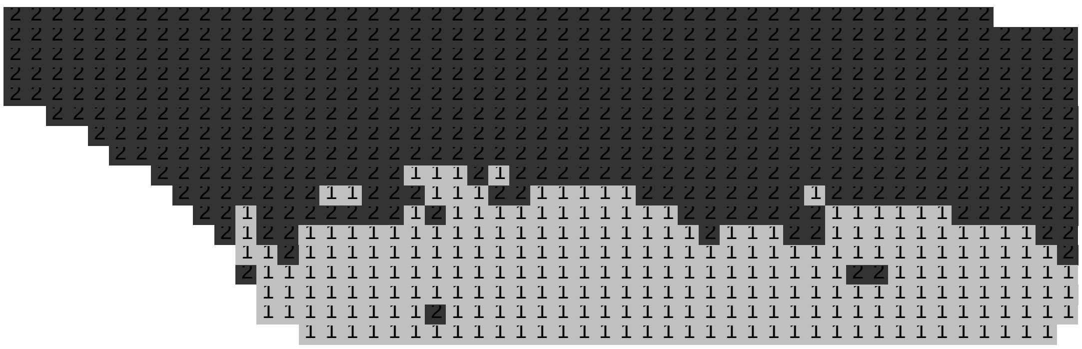

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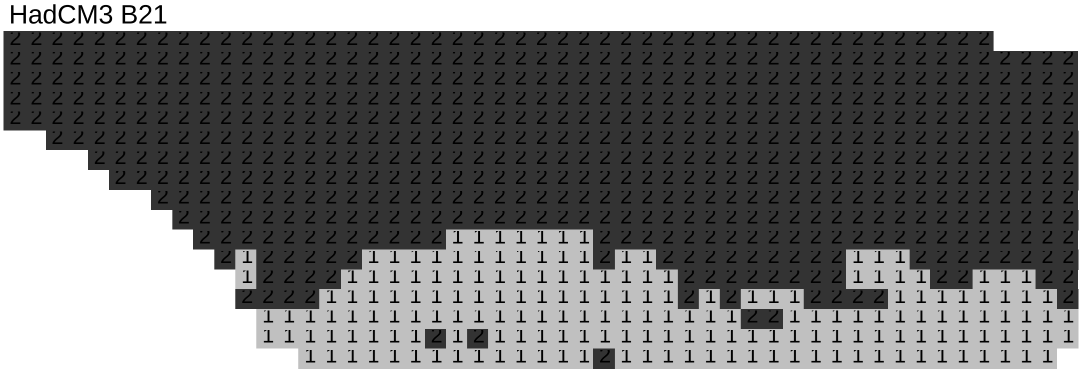

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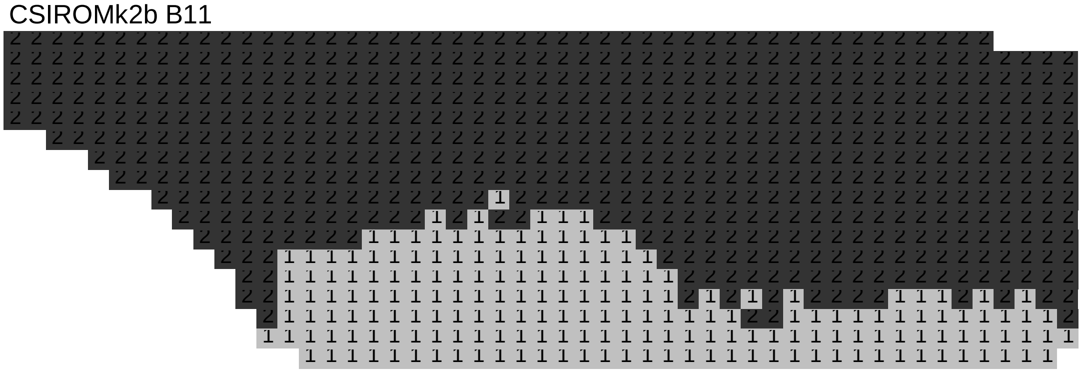
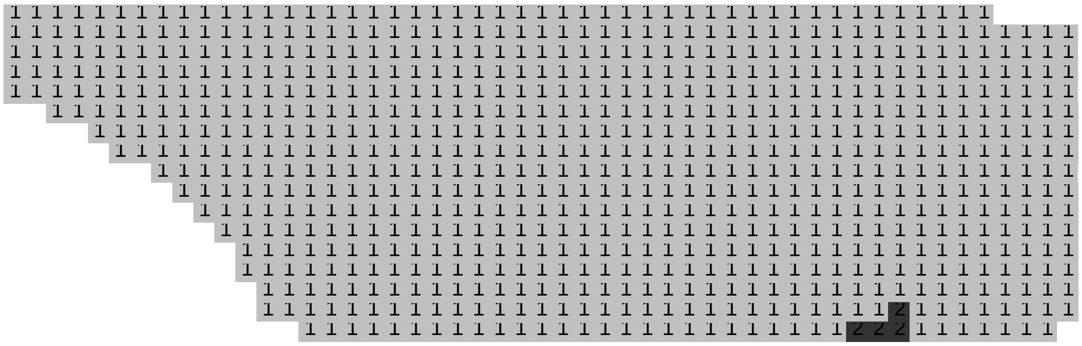


Bur oak (*Quercus macrocarpa*)

1961-90 normals



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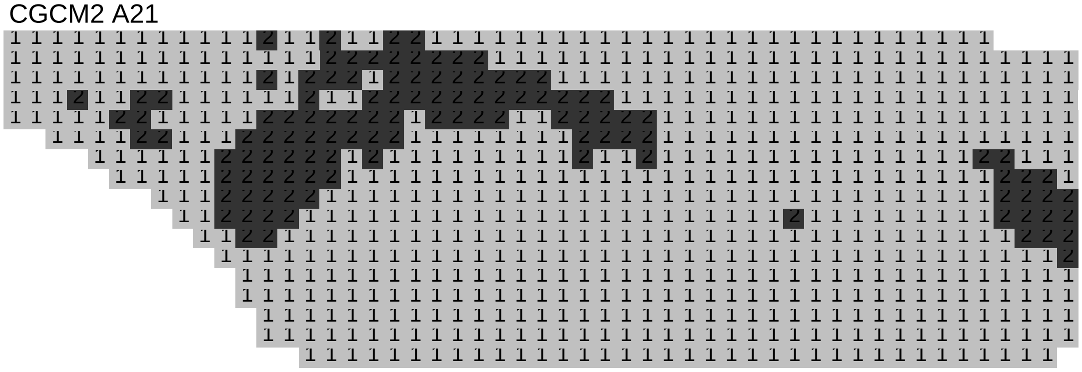

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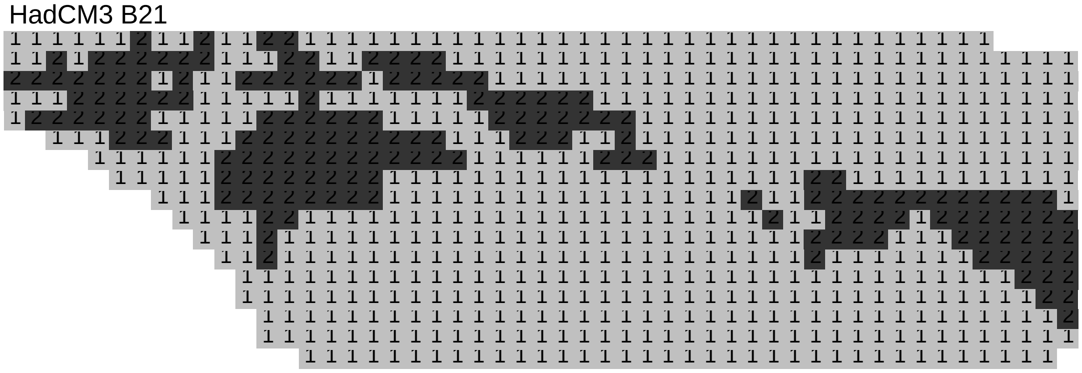

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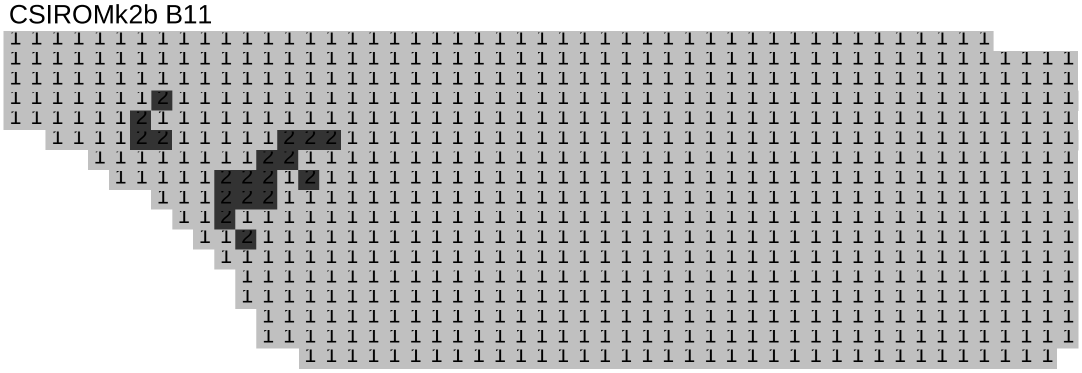
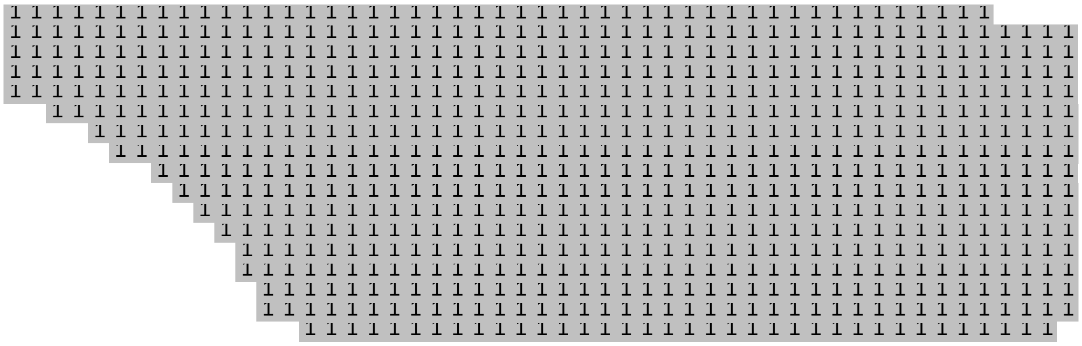


Red oak (*Quercus rubra*)

1961-90 normals



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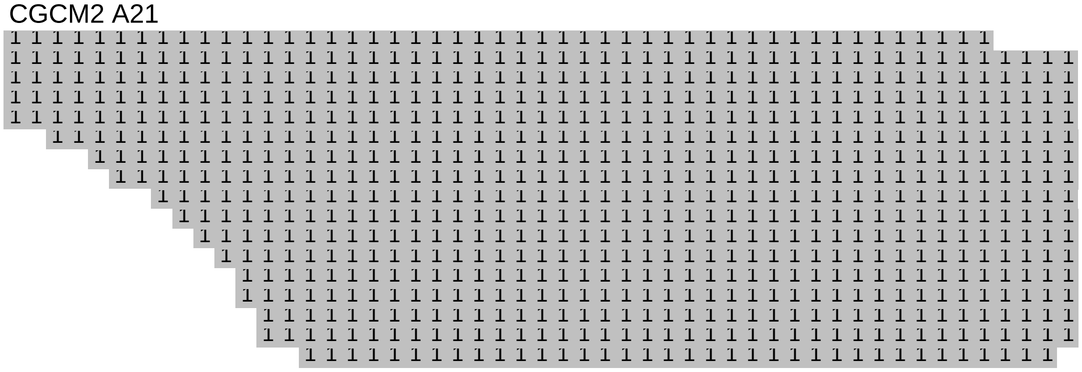

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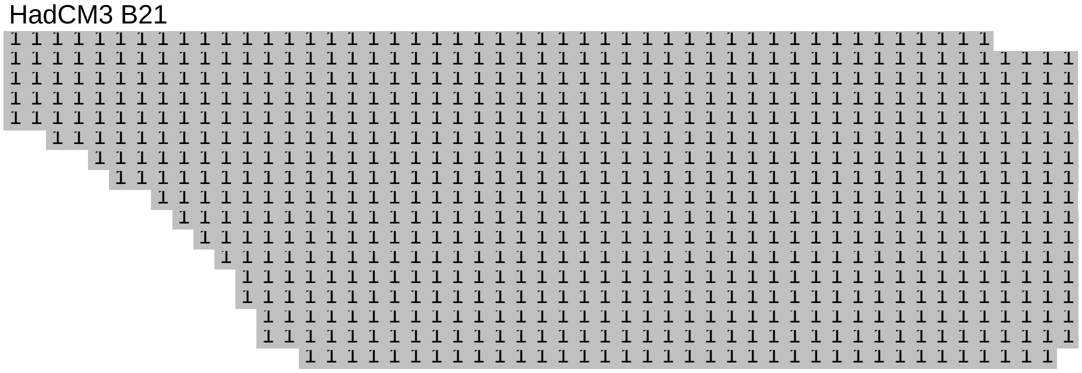

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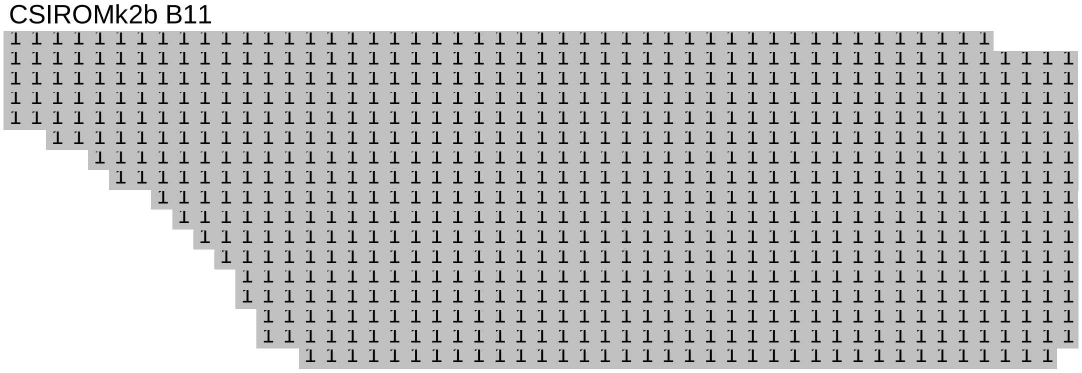
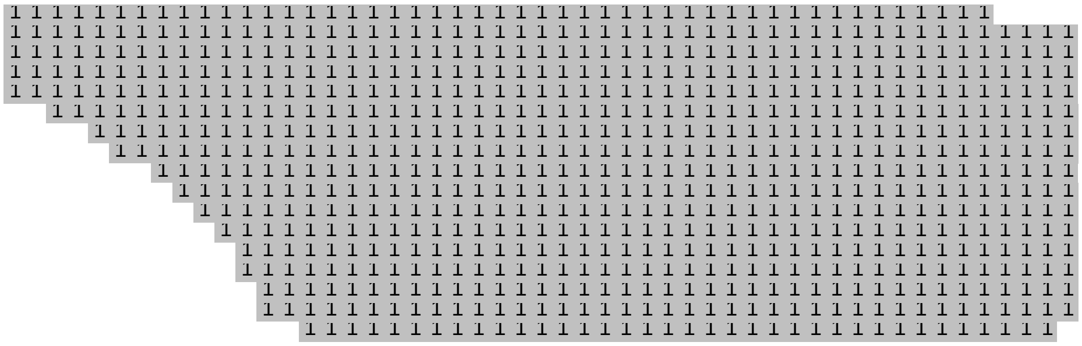


Basswood (*Tilia americana*)

1961-90 normals



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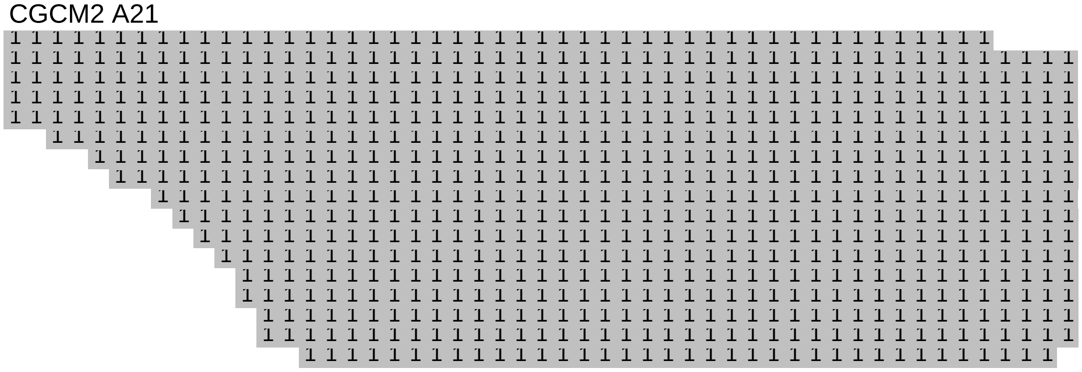

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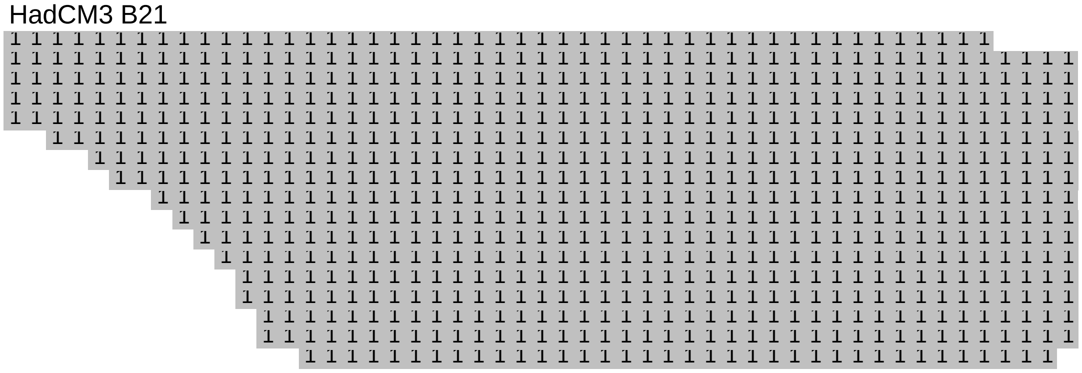

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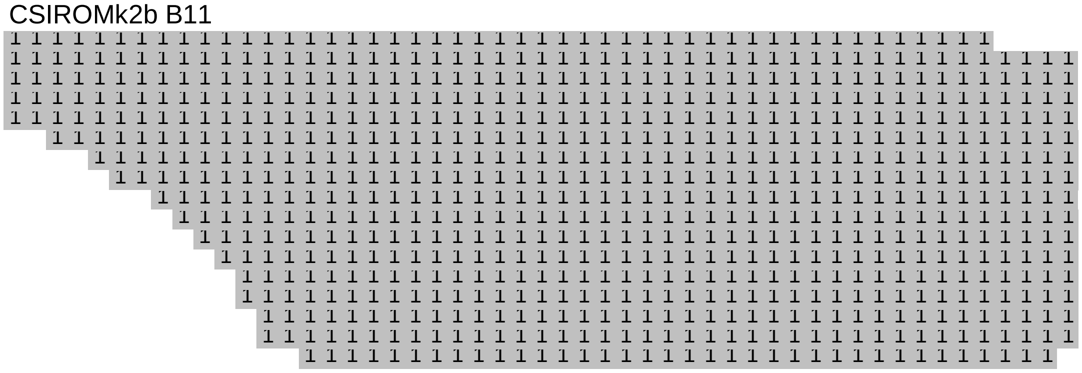
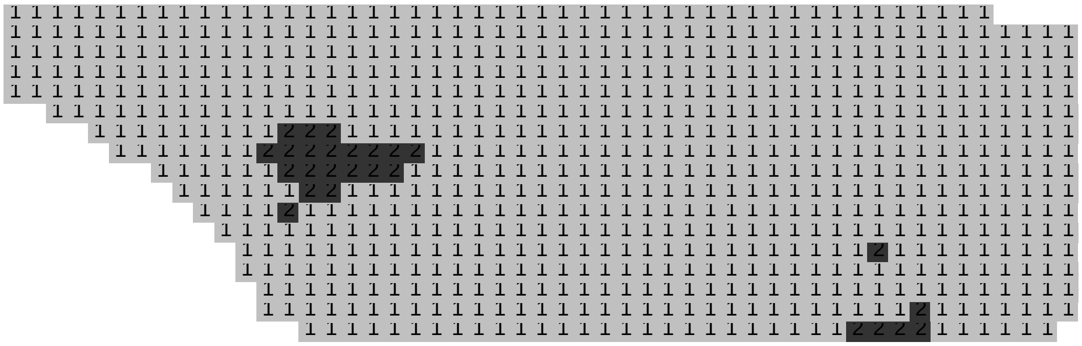


American elm (*Ulmus americana*)

1961-90 normals



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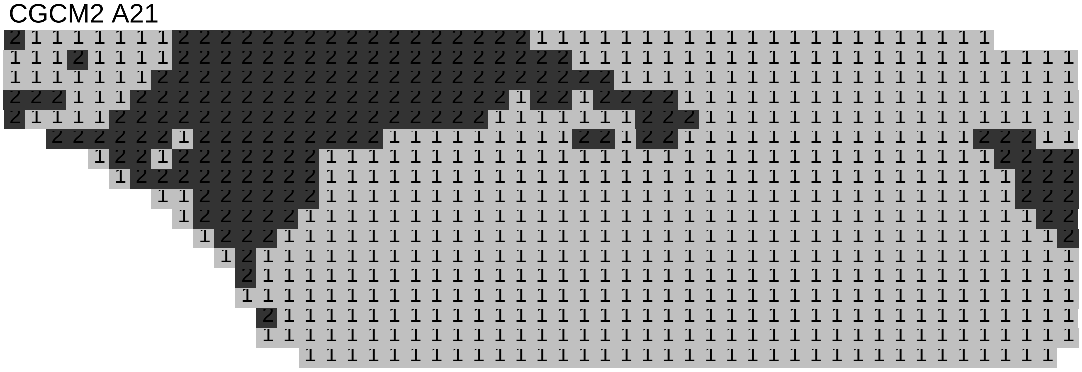

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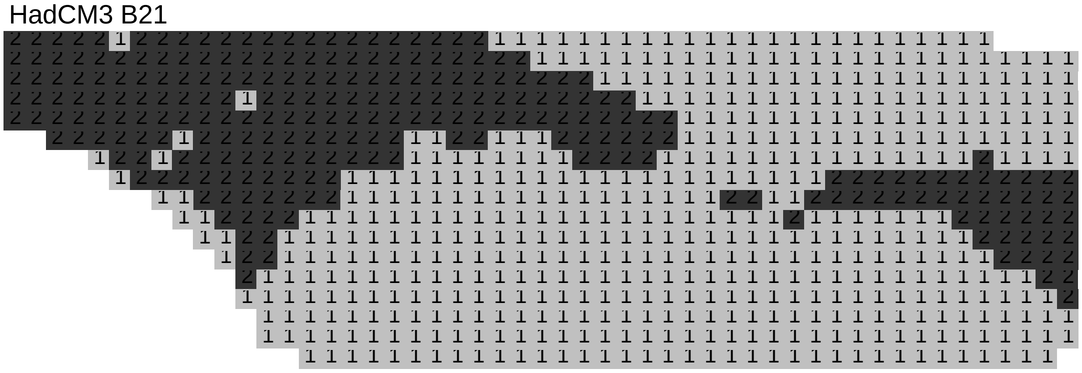

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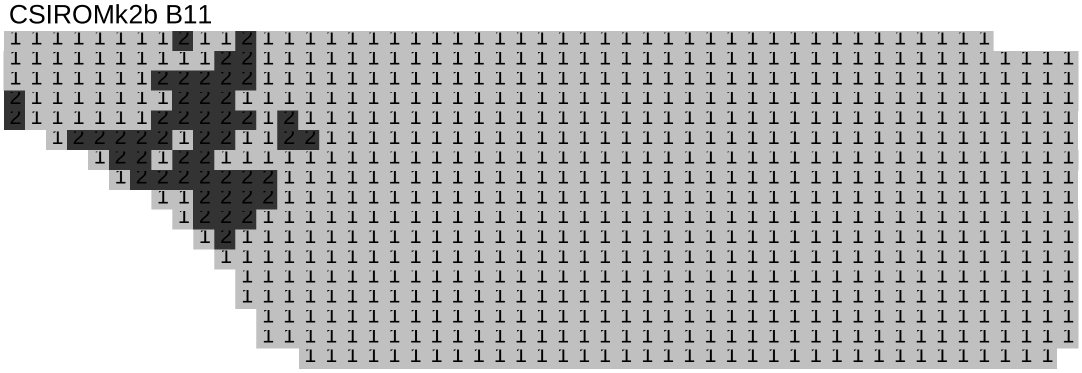
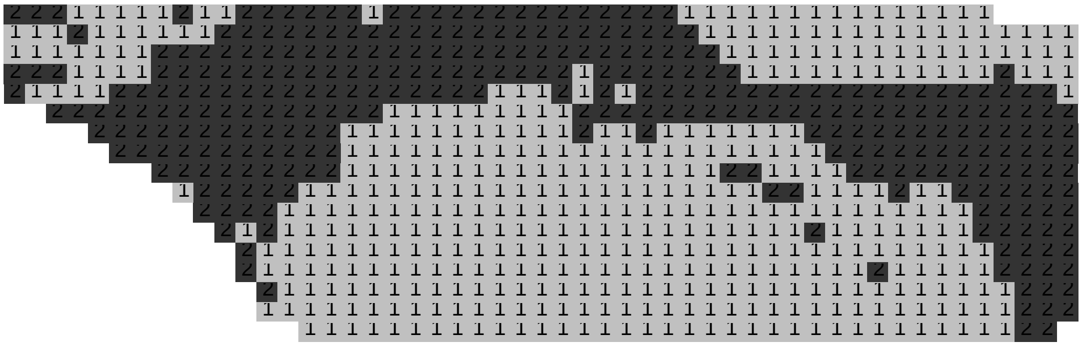


Balsam fir (*Abies balsamea*)

1961-90 normals



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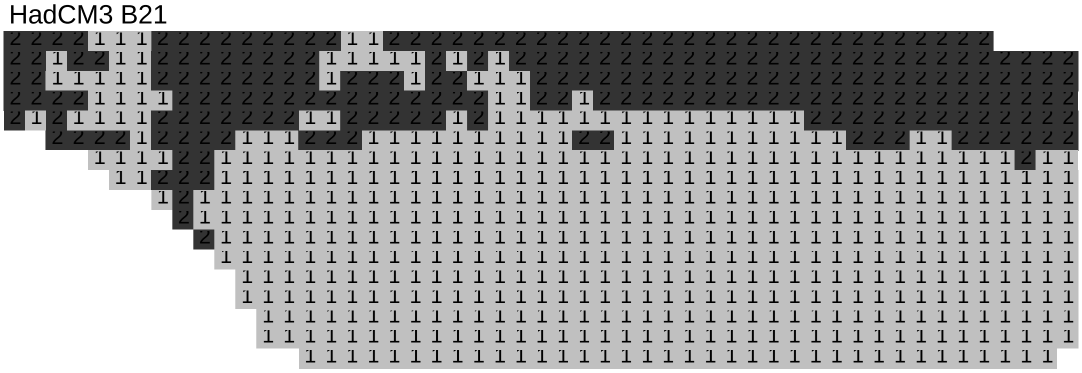

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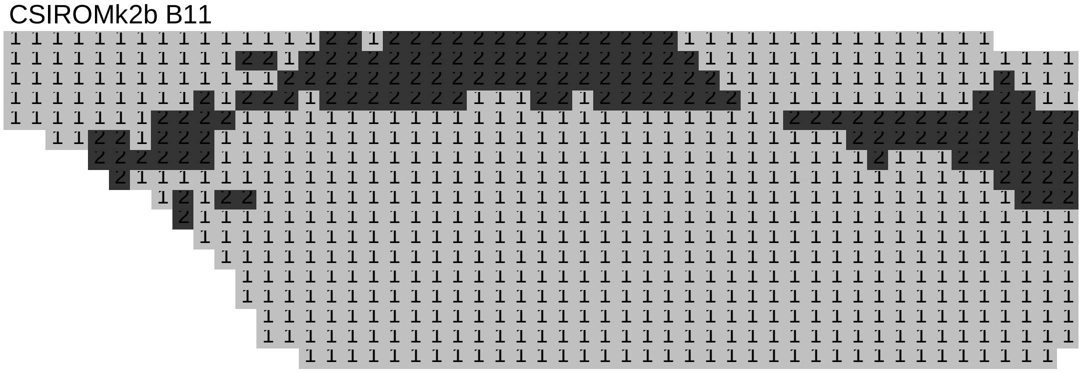
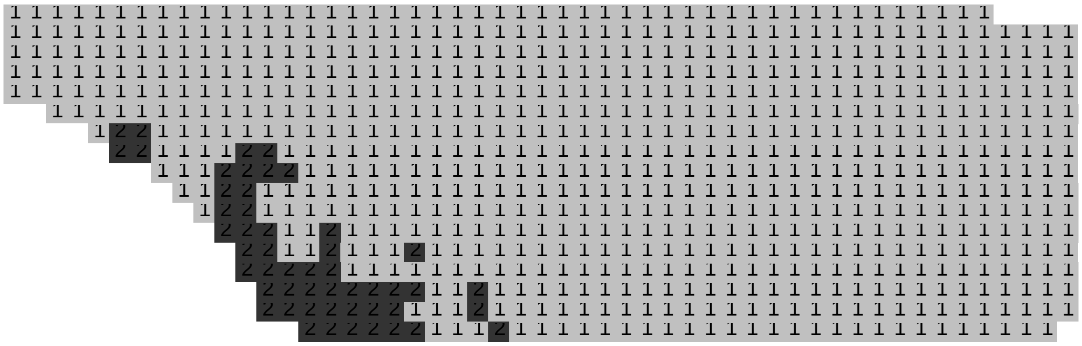


Rocky Mountain juniper (*Juniperus scopulorum*)

1961-90 normals



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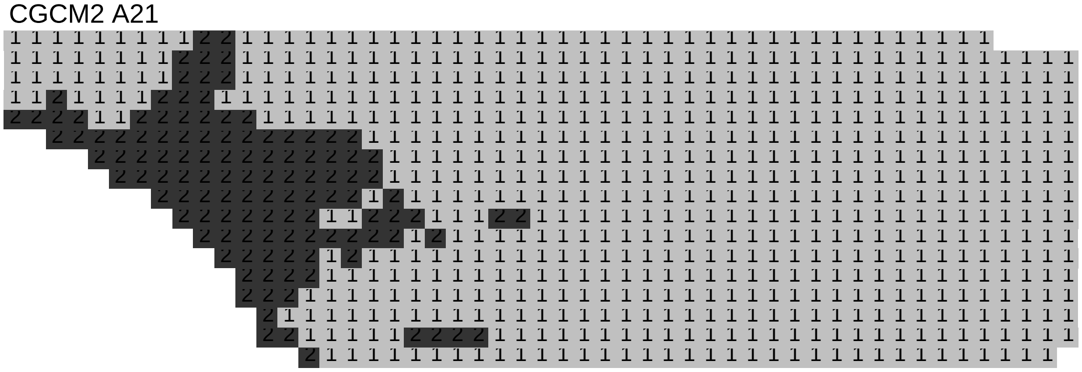

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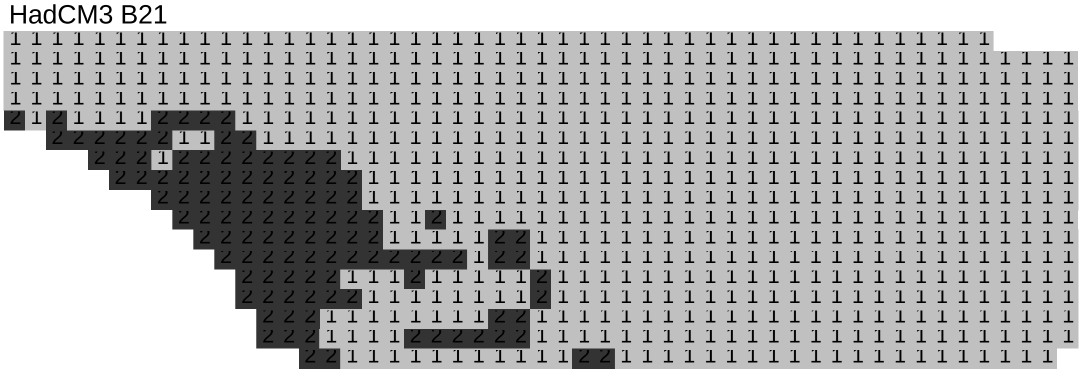

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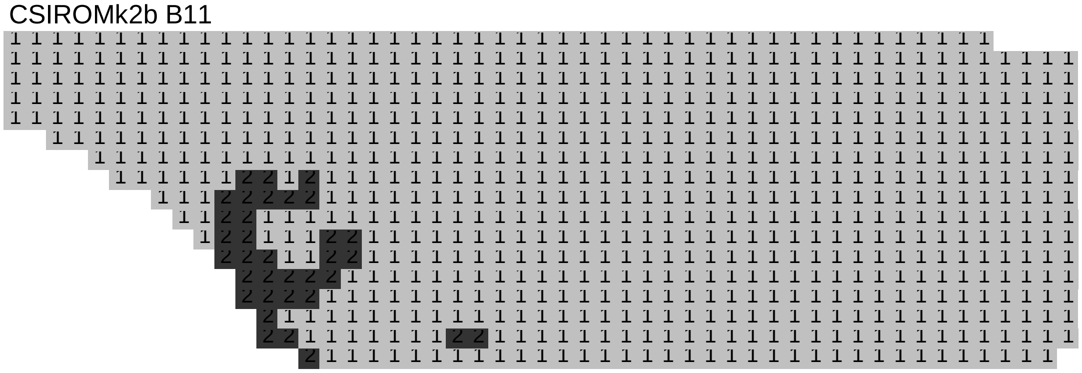
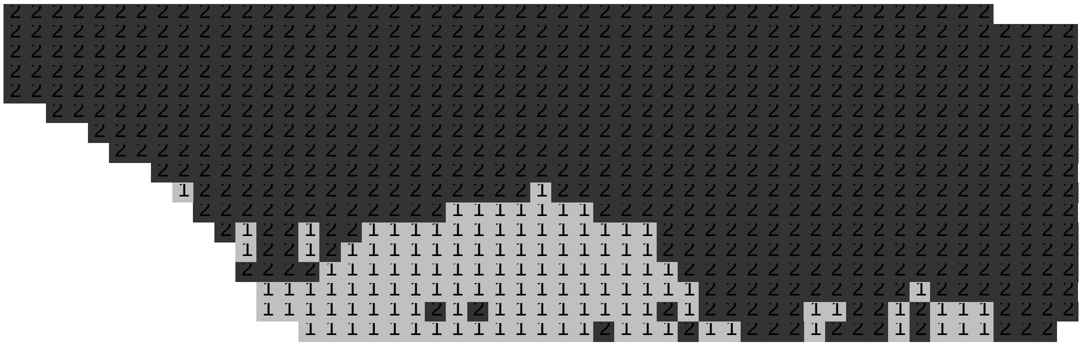


Tamarack (*Larix laricina*)

1961-90 normals



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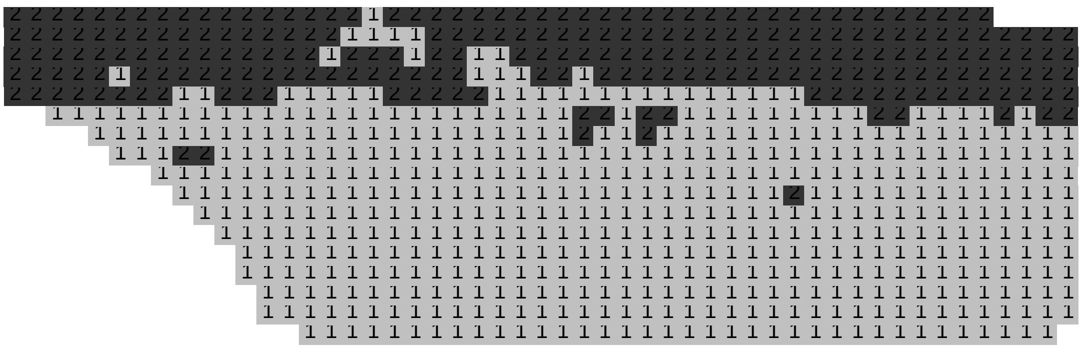

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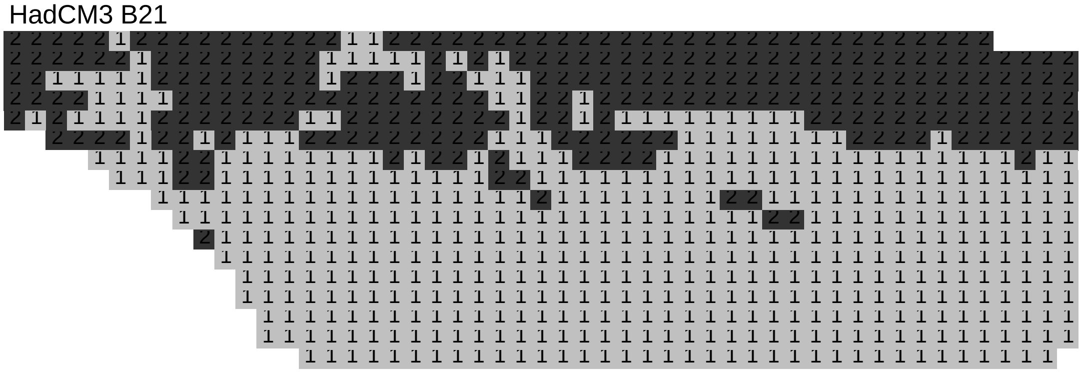

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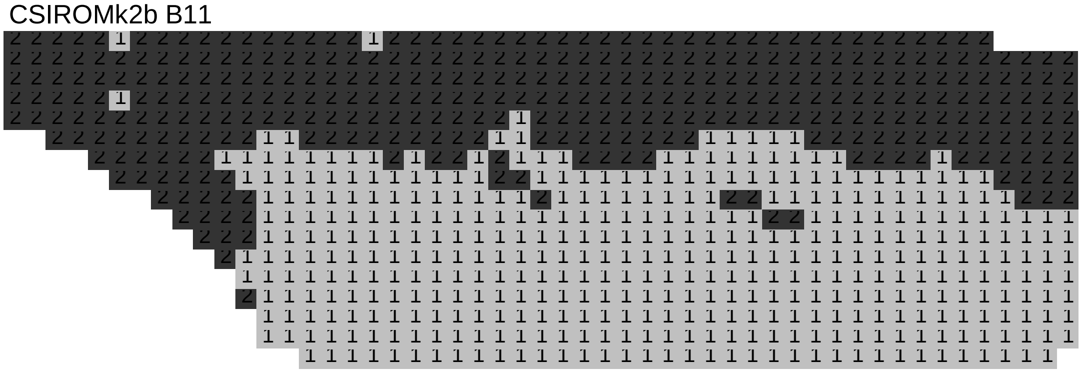
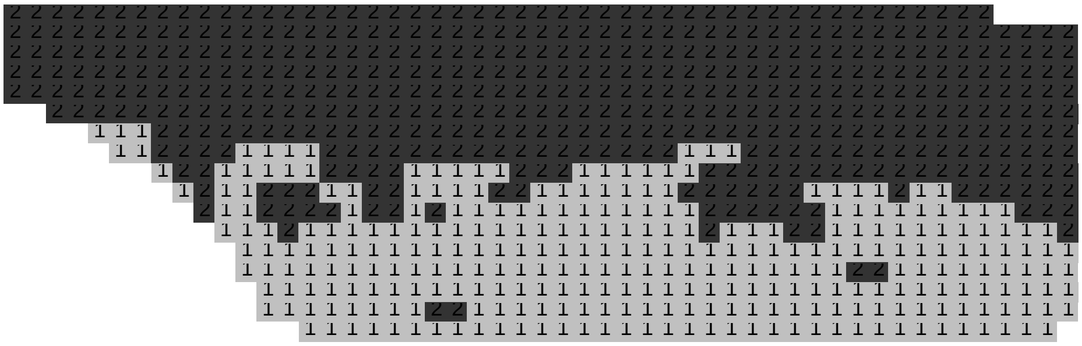


White spruce (*Picea glauca*)

1961-90 normals



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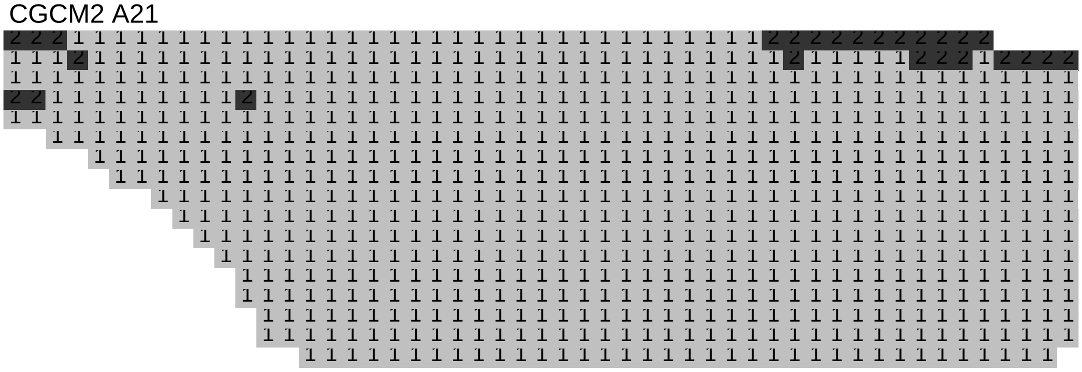

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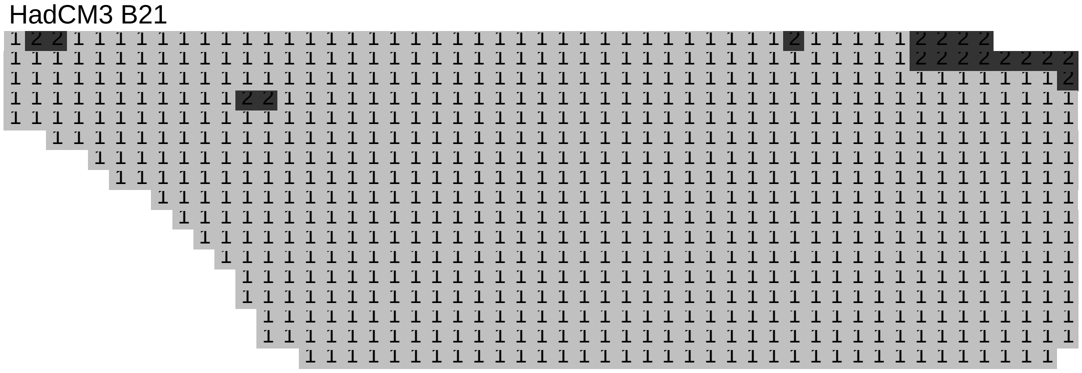

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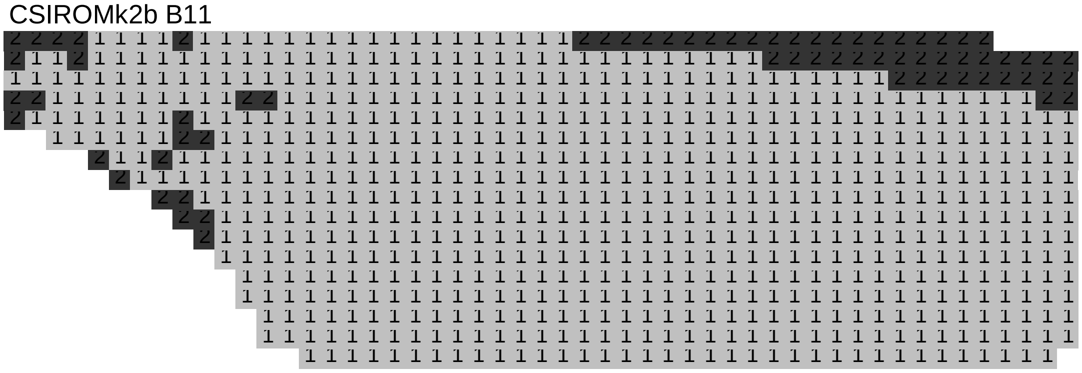
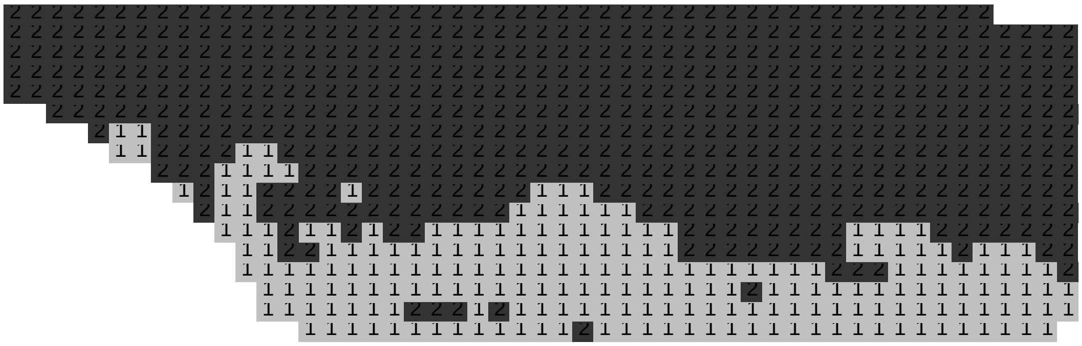


Black spruce (*Picea mariana*)

1961-90 normals



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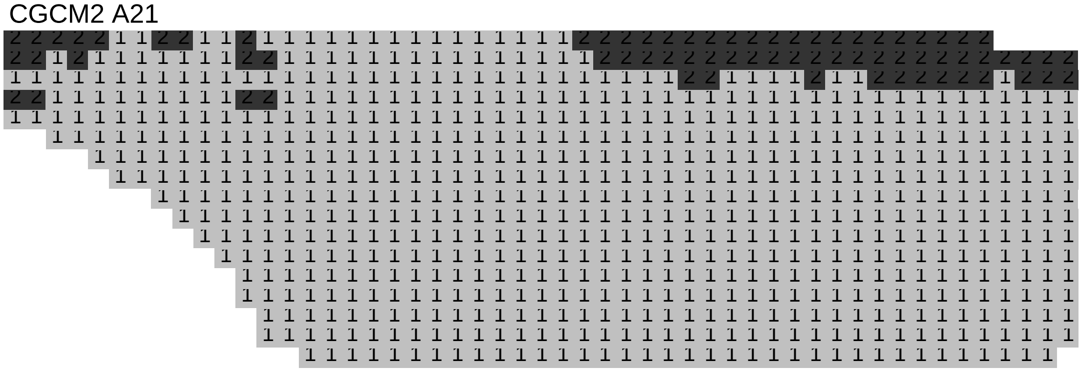

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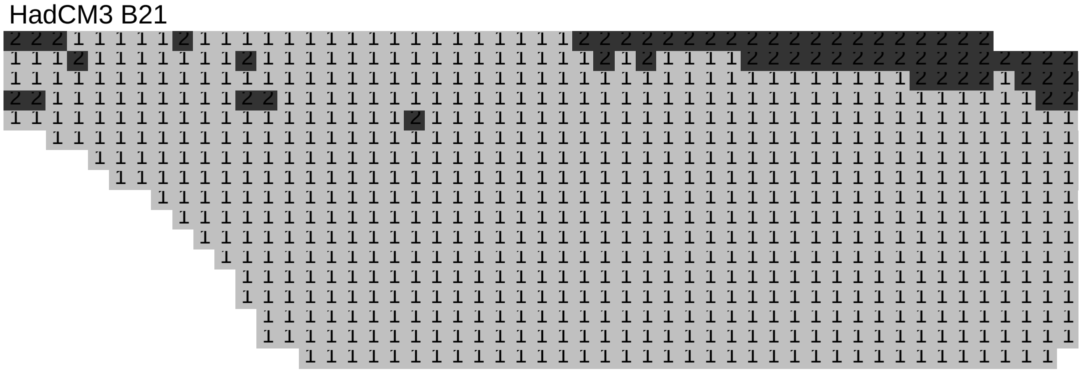

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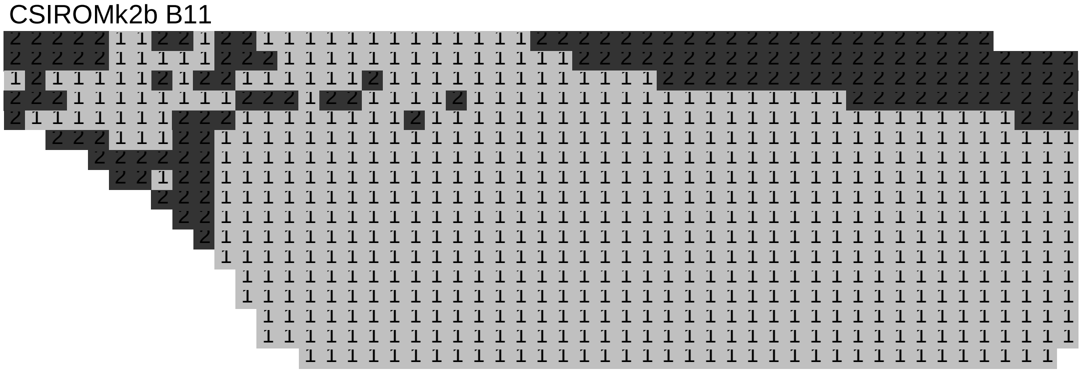
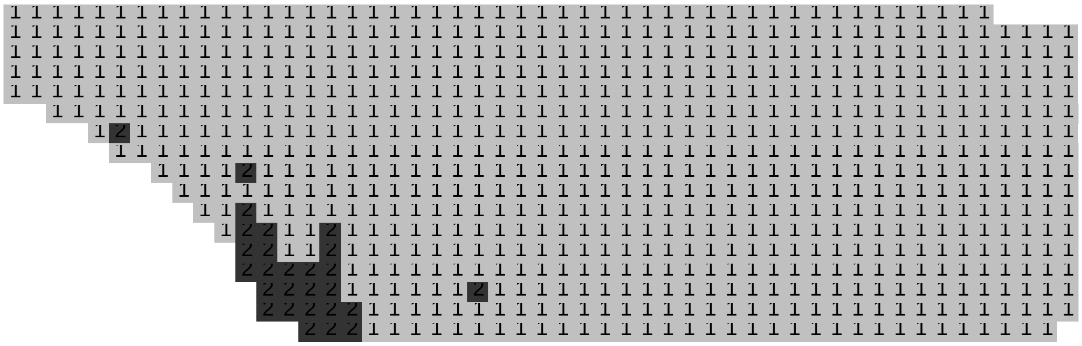


Blue spruce (*Picea pungens*)

1961-90 normals



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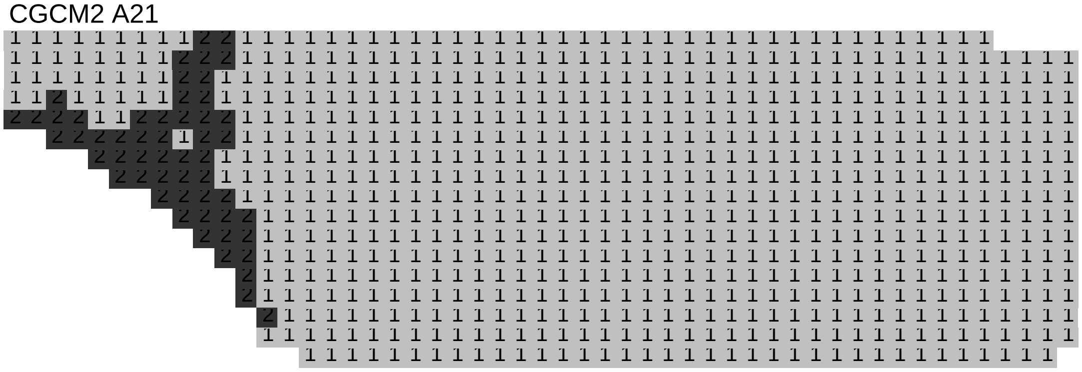

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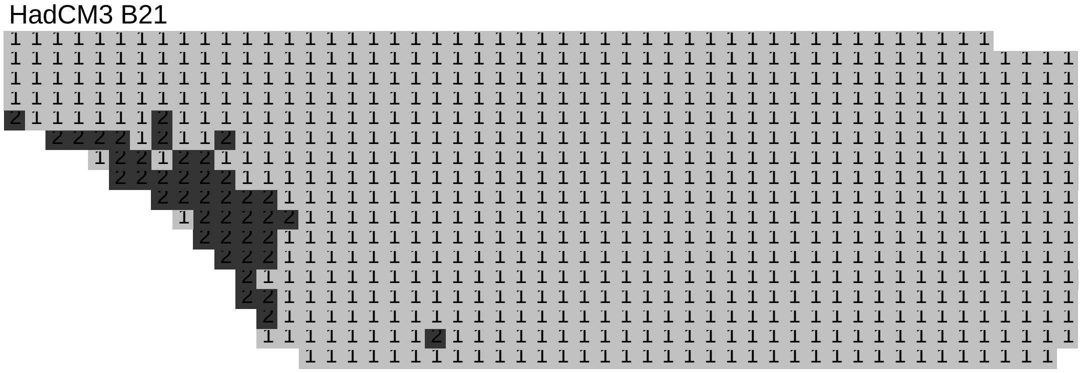

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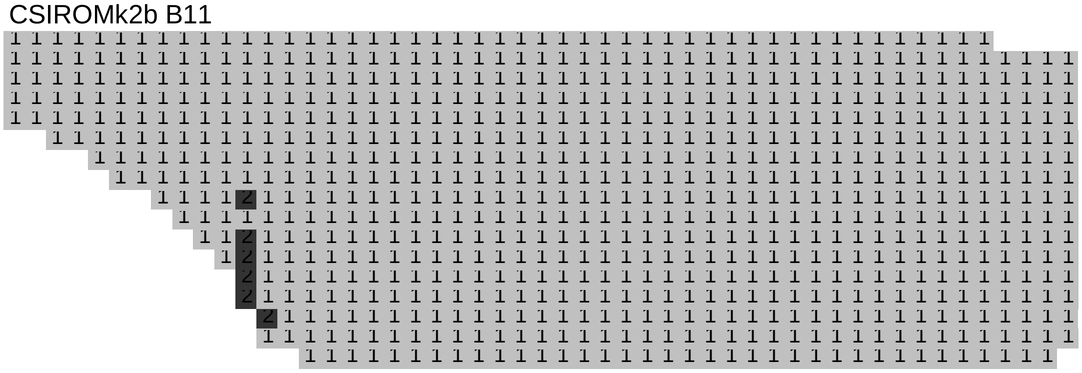
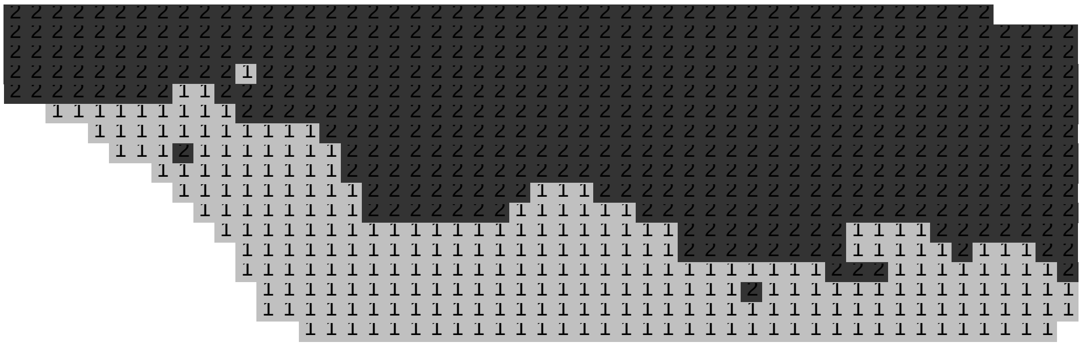


Jack pine (*Pinus banksiana*)

1961-90 normals



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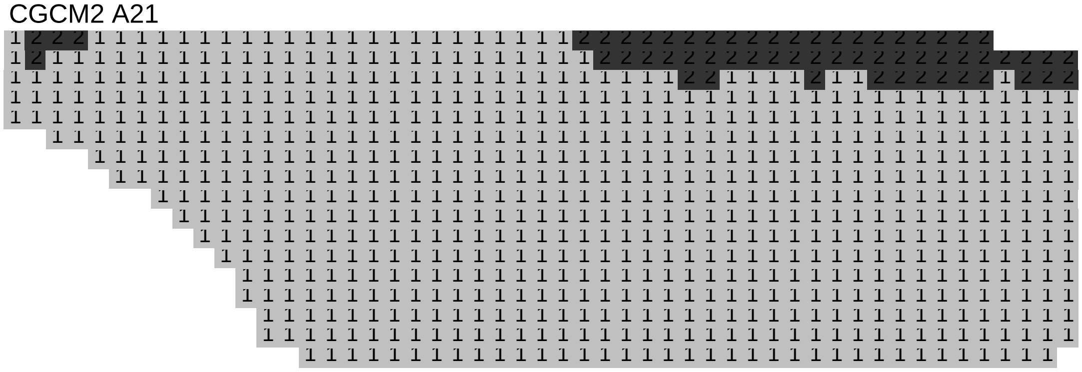

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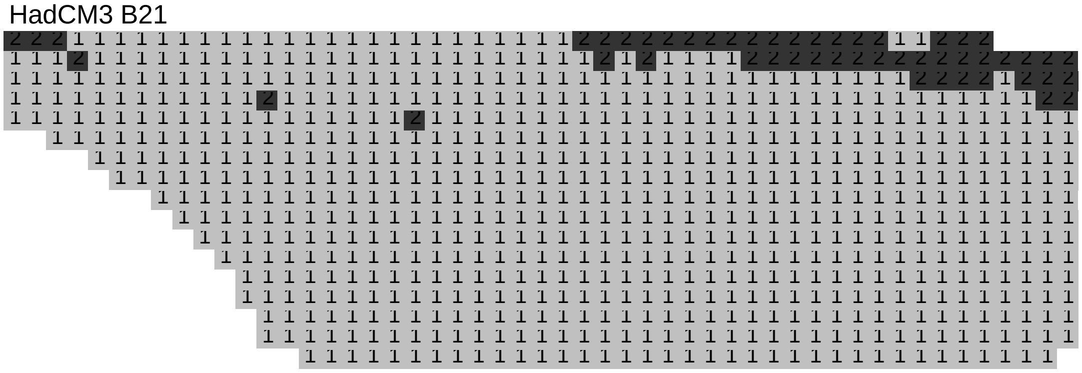

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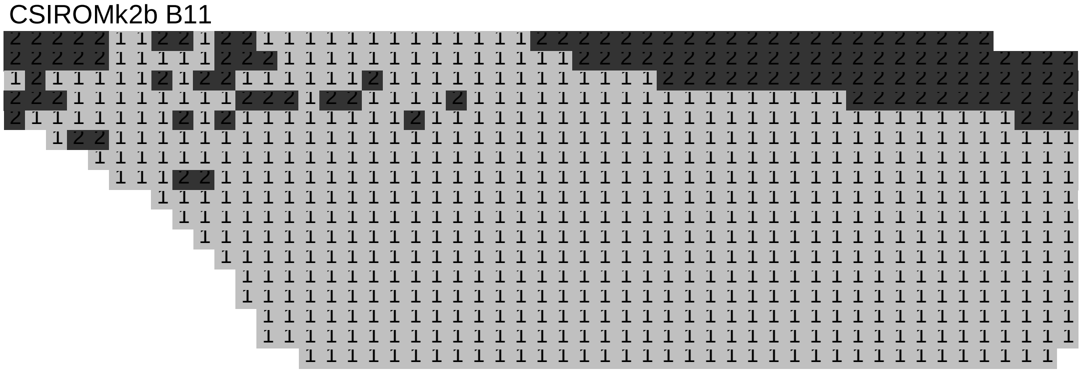
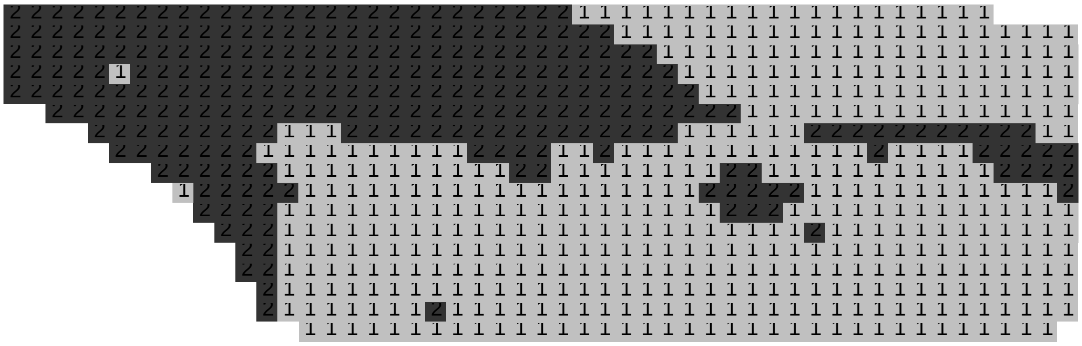


Lodgepole pine (*Pinus contorta*)

1961-90 normals



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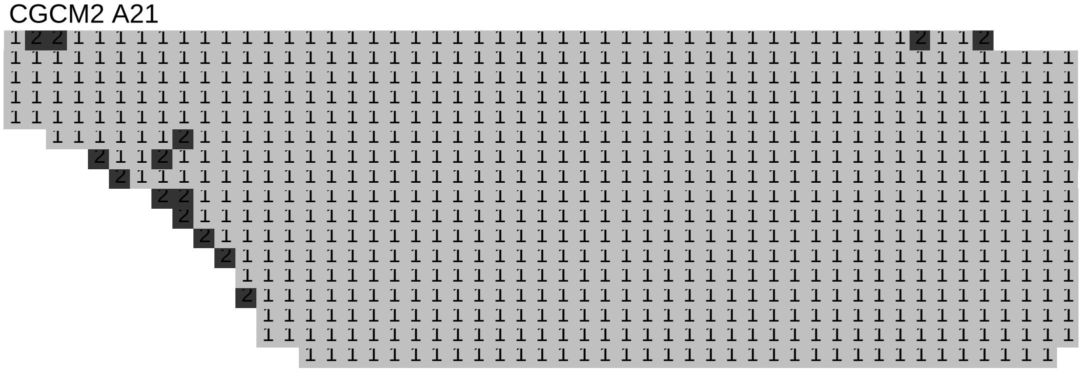

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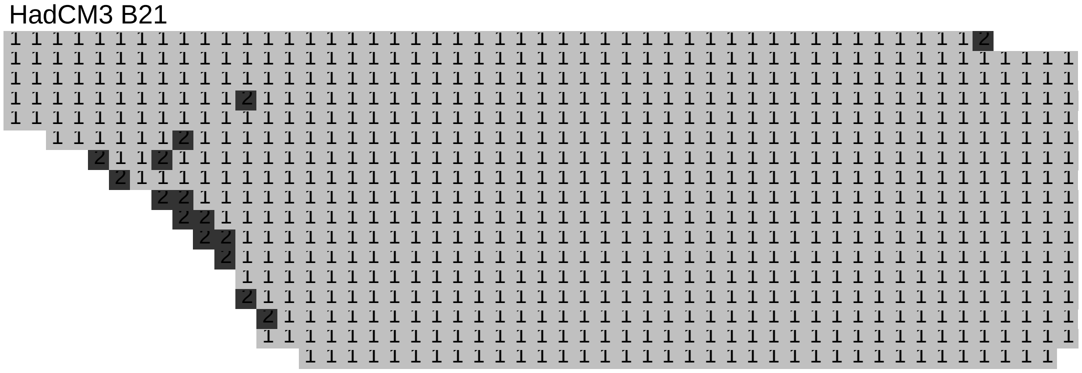

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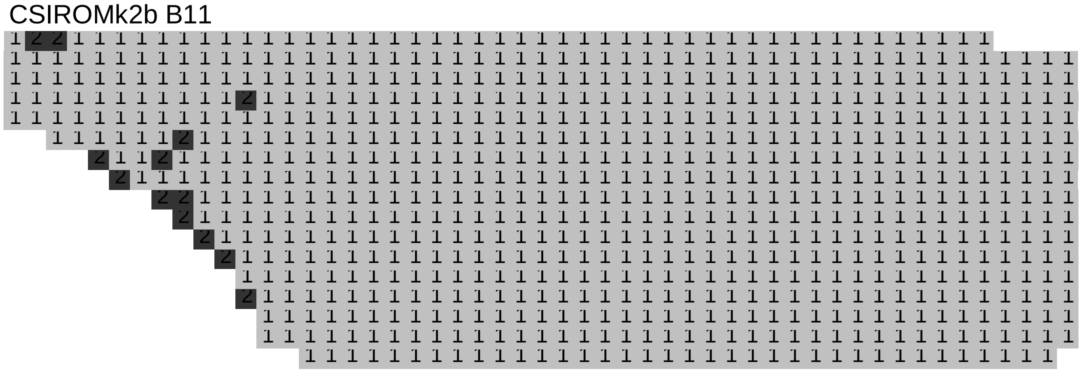
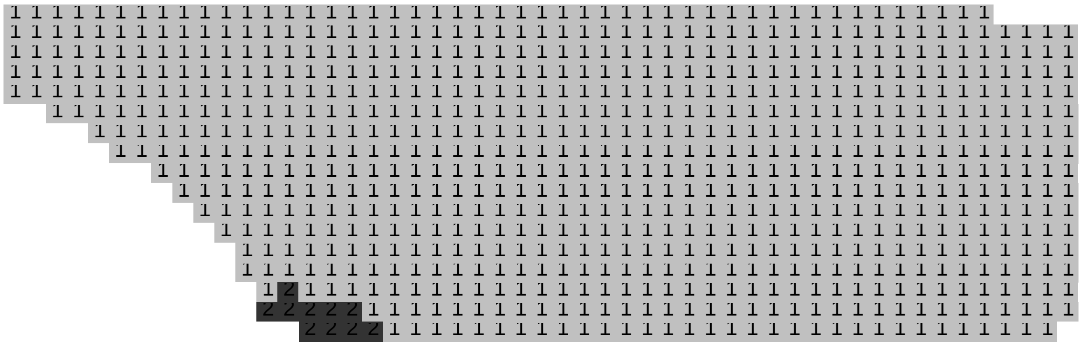


Ponderosa pine (*Pinus ponderosa*)

1961-90 normals



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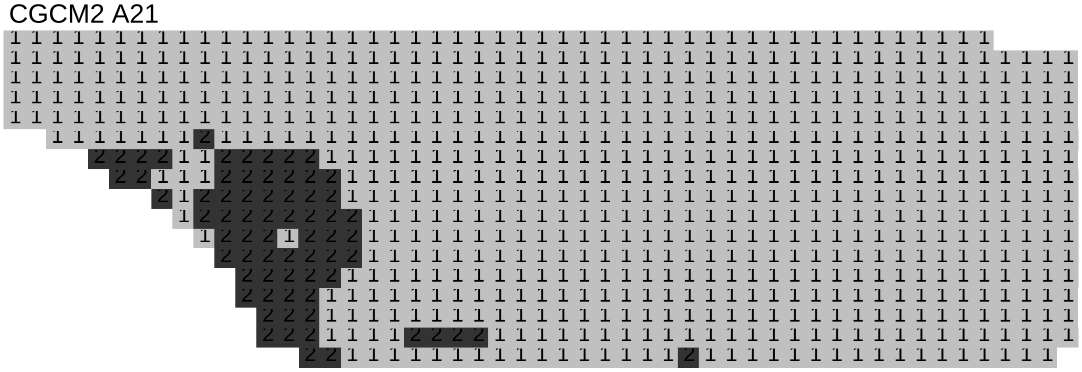

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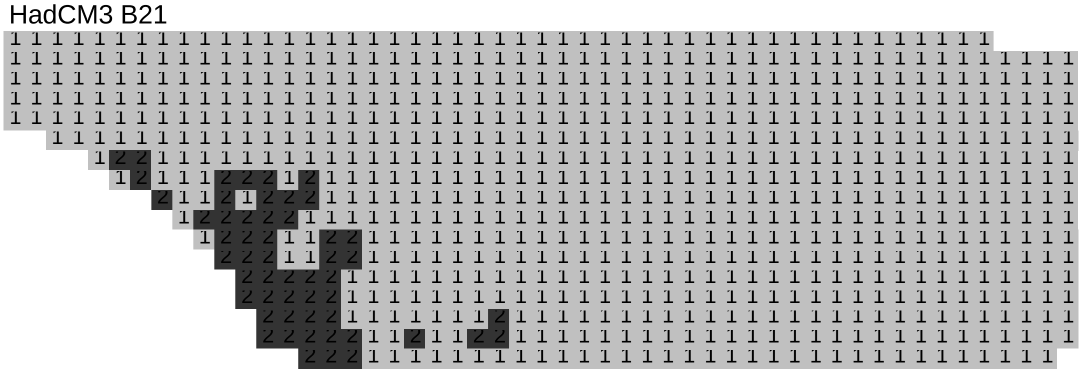

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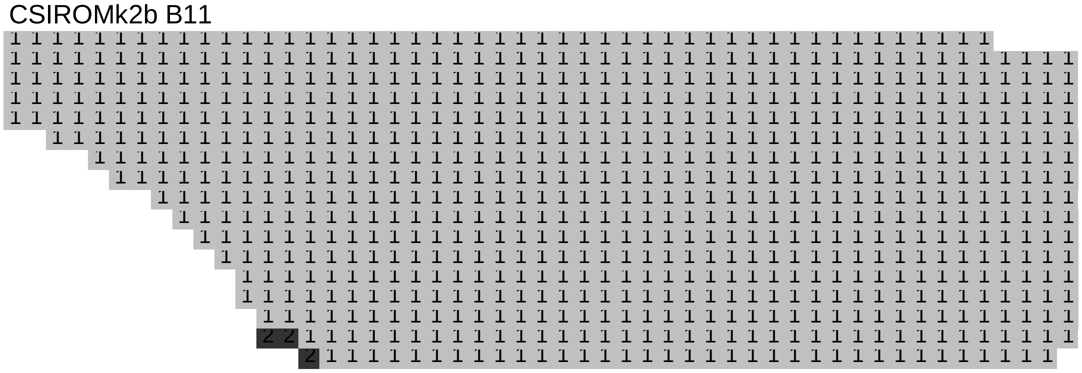
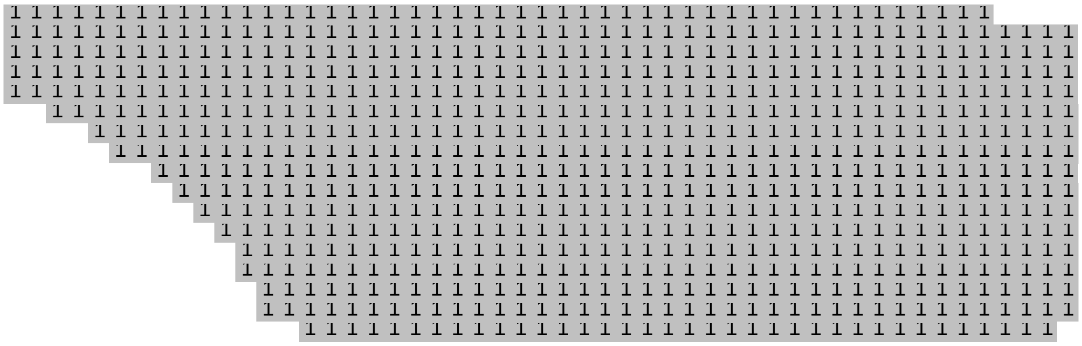


Red pine (*Pinus resinosa*)

1961-90 normals



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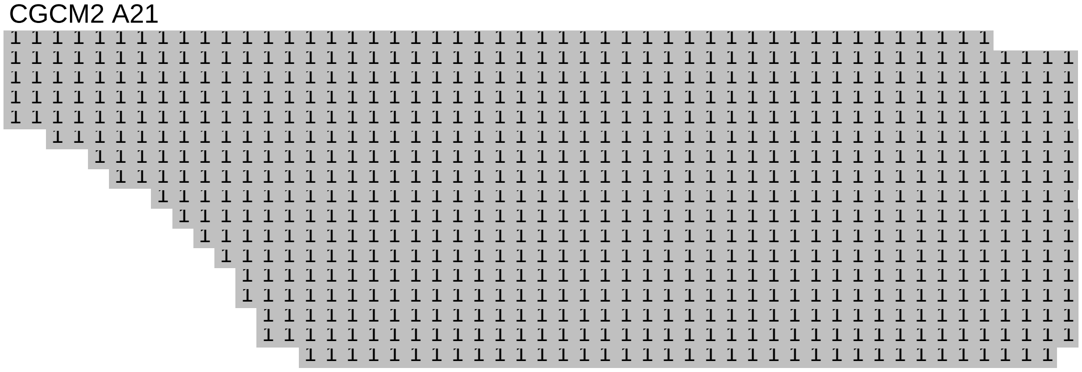

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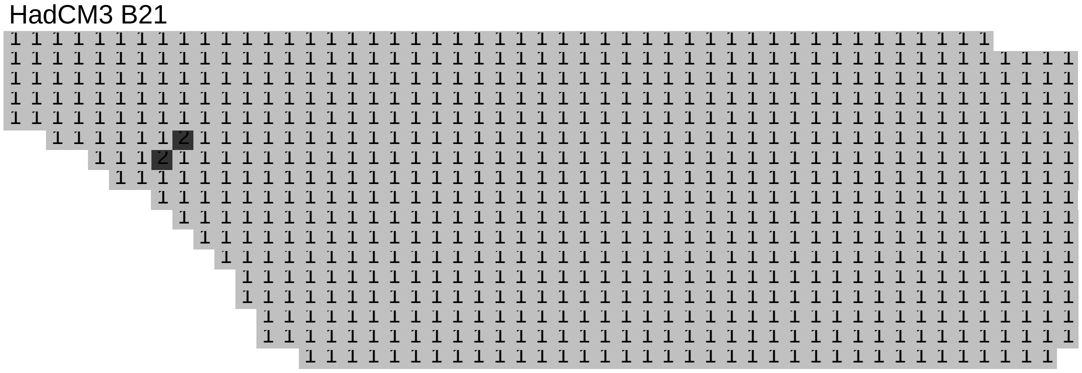

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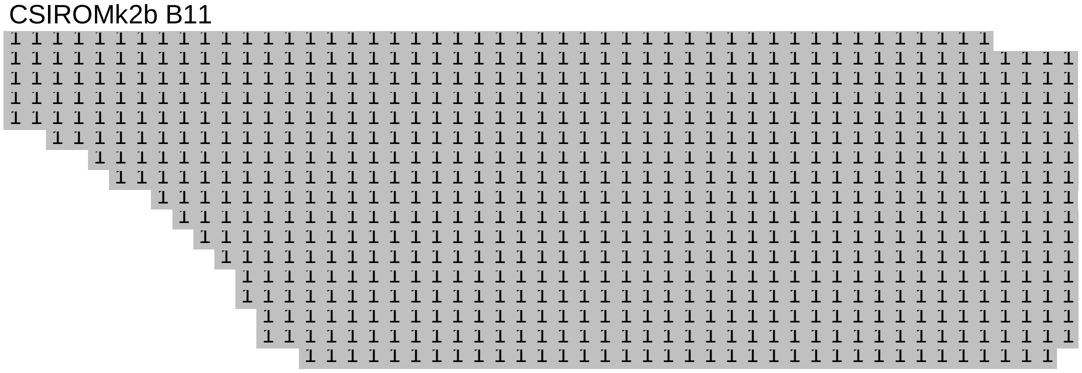
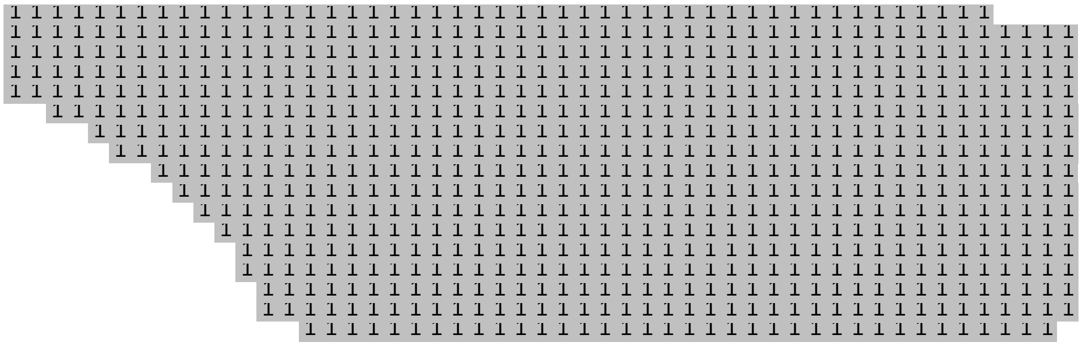


Eastern white pine (*Pinus strobus*)

1961-90 normals



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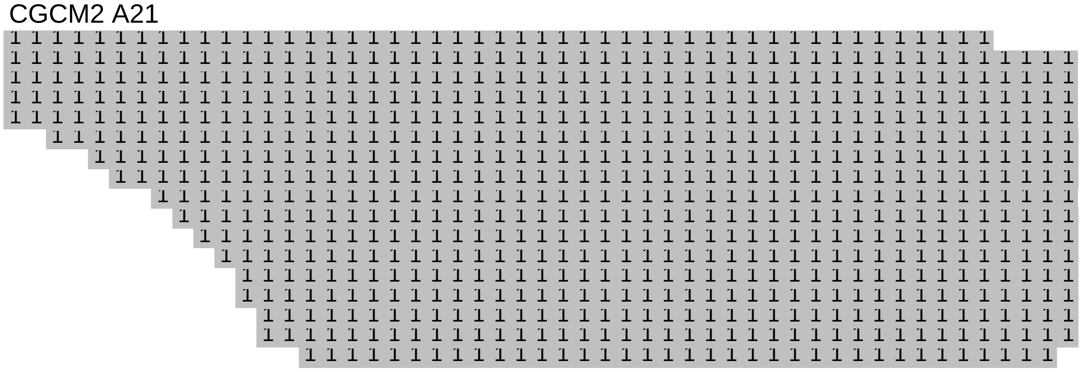

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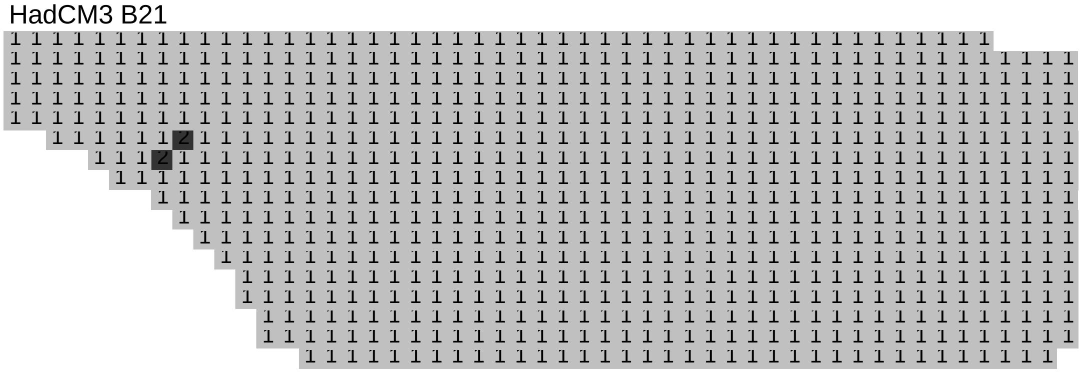

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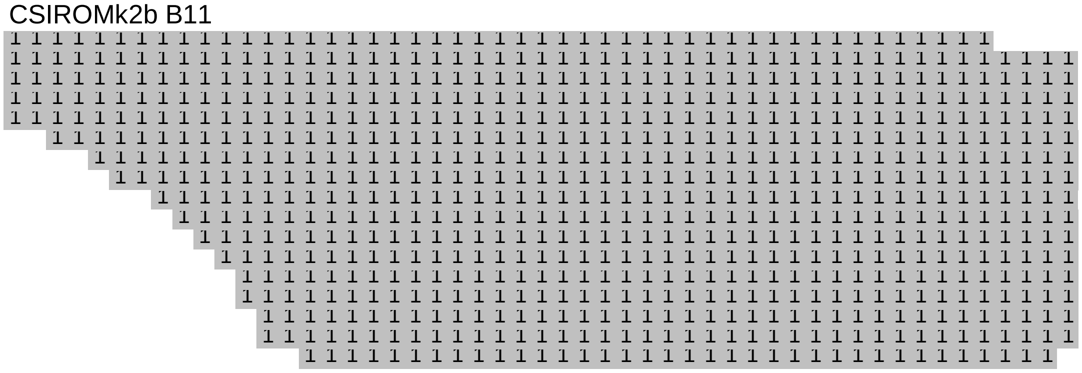
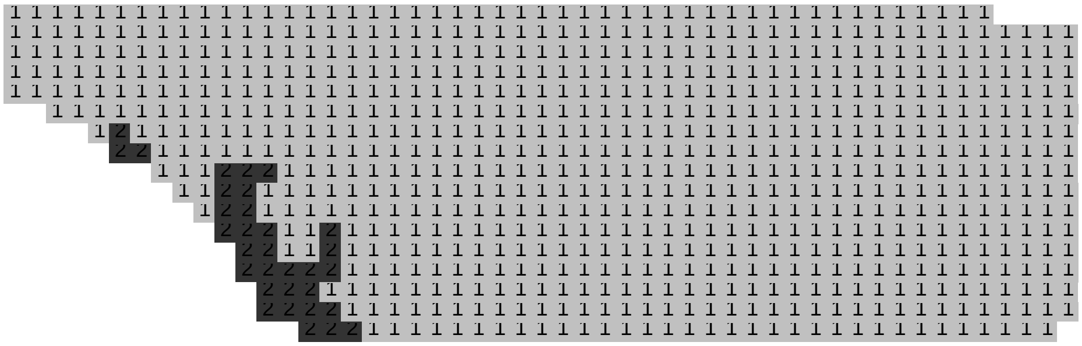


Douglas-fir (*Pseudotsuga menziesii*)

1961-90 normals



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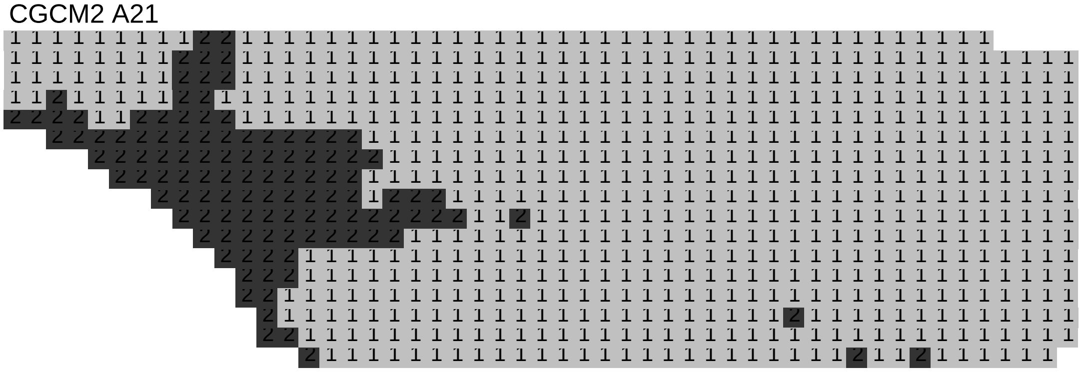

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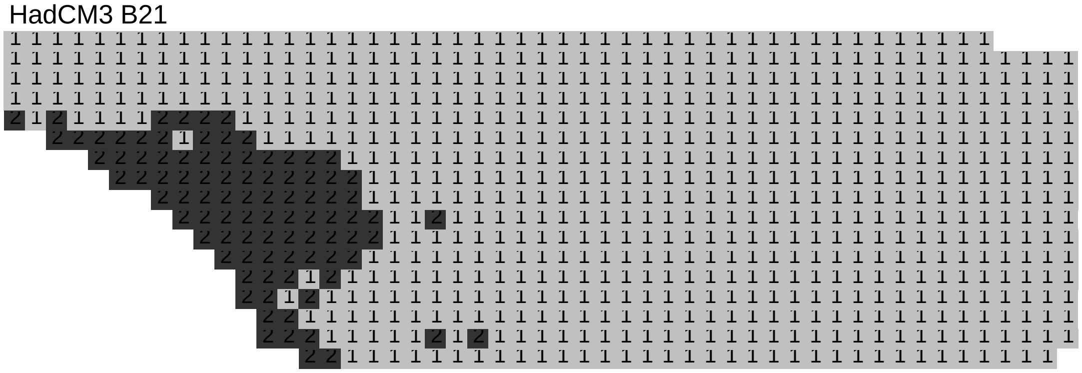

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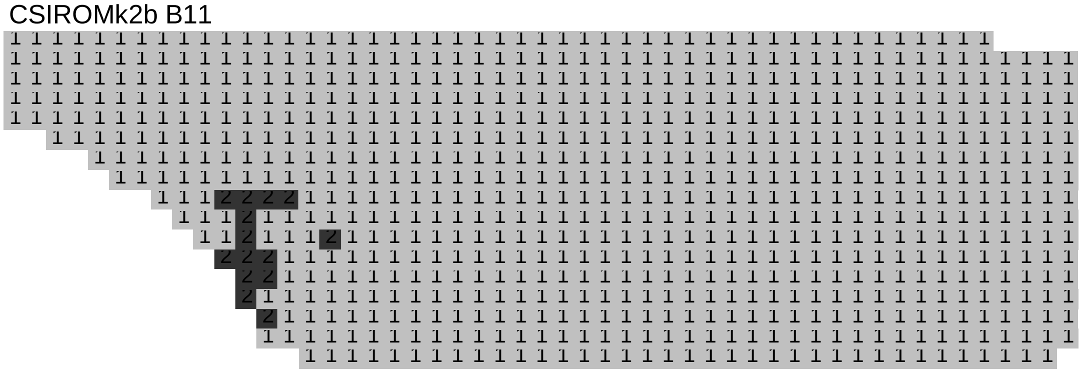
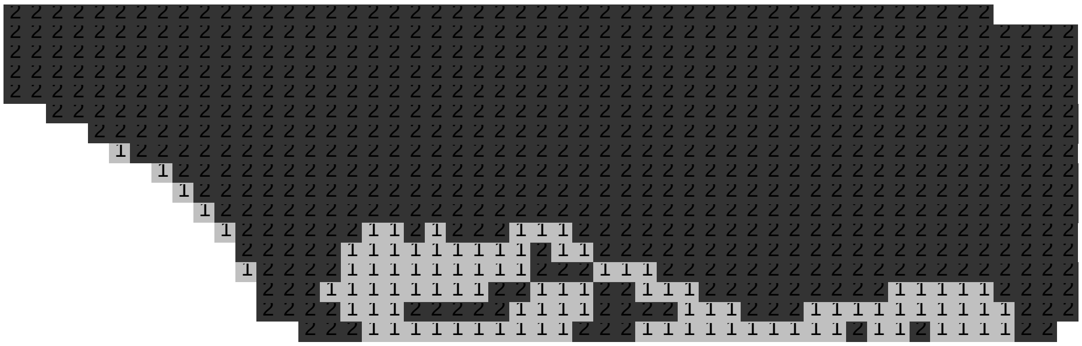


Scots pine (*Pinus sylvestris*)

1961-90 normals



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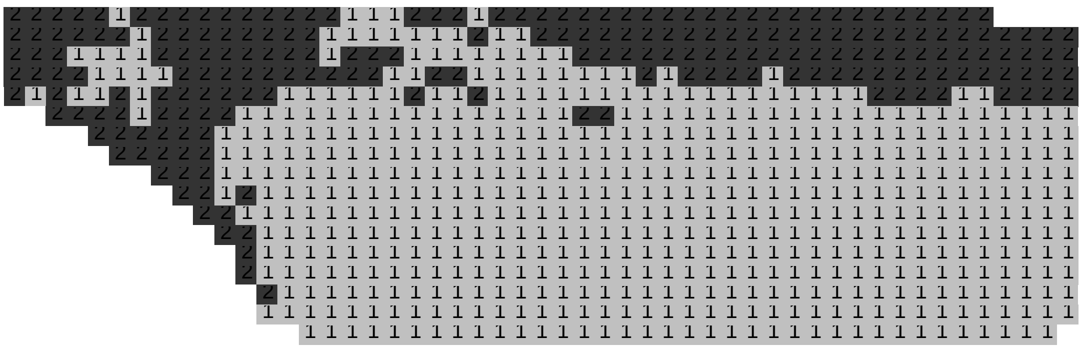

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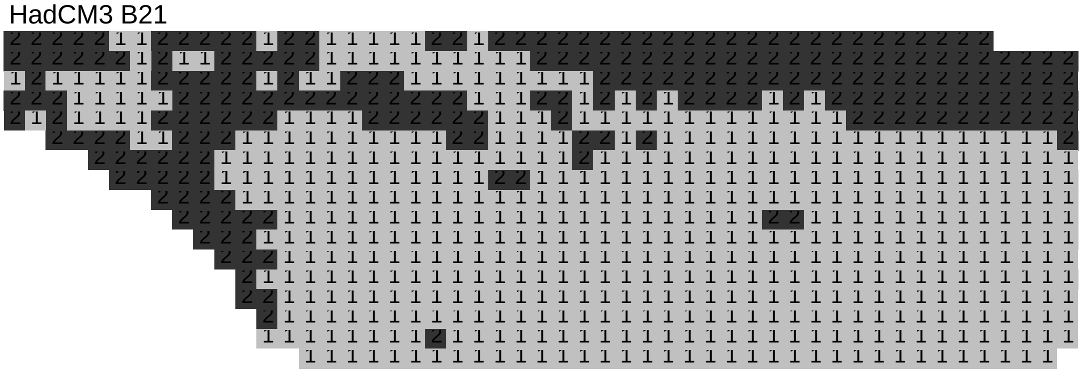

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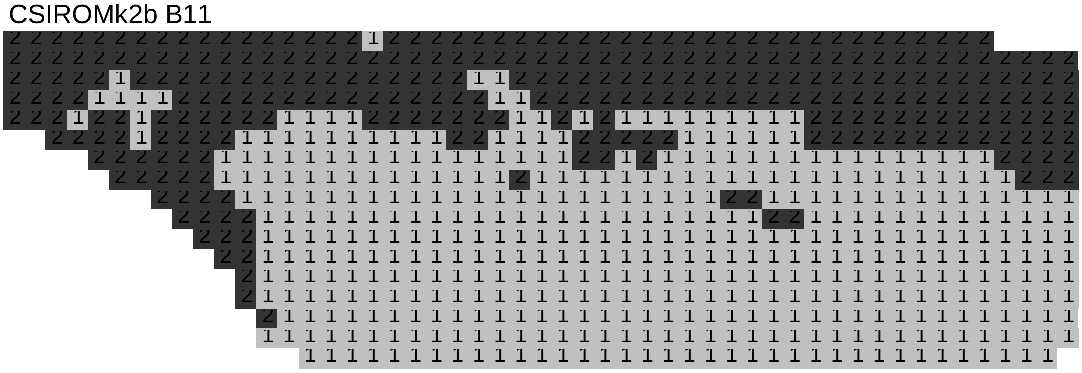
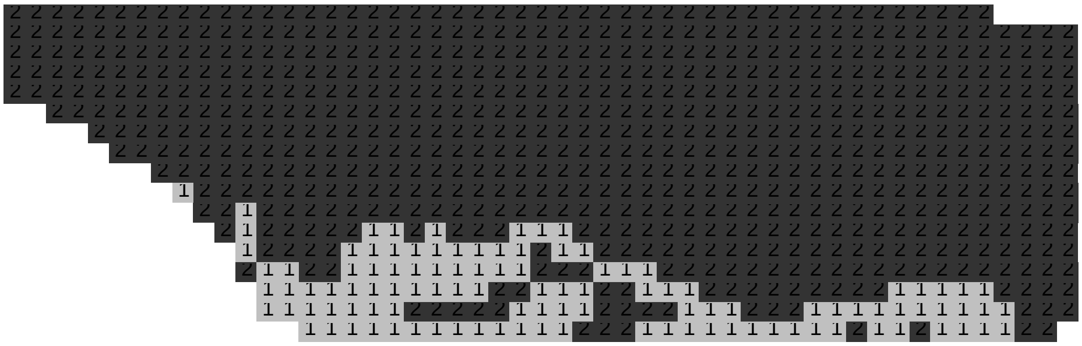


Siberian larch (*Larix sibirica*)

1961-90 normals



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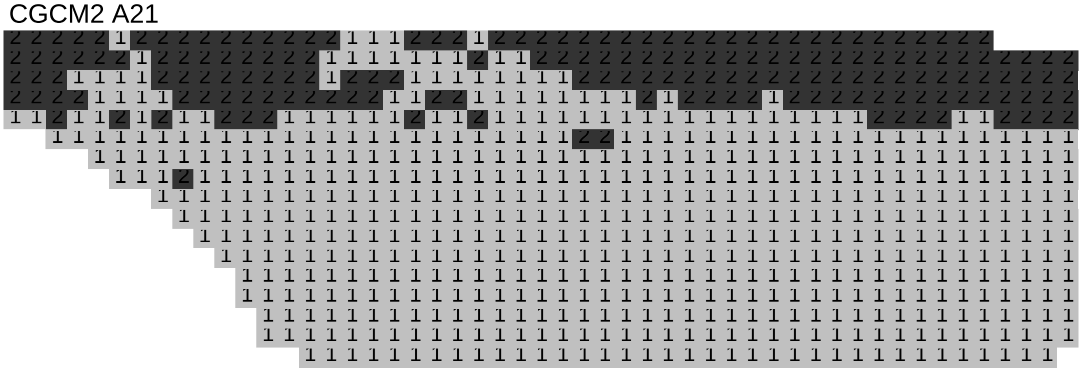

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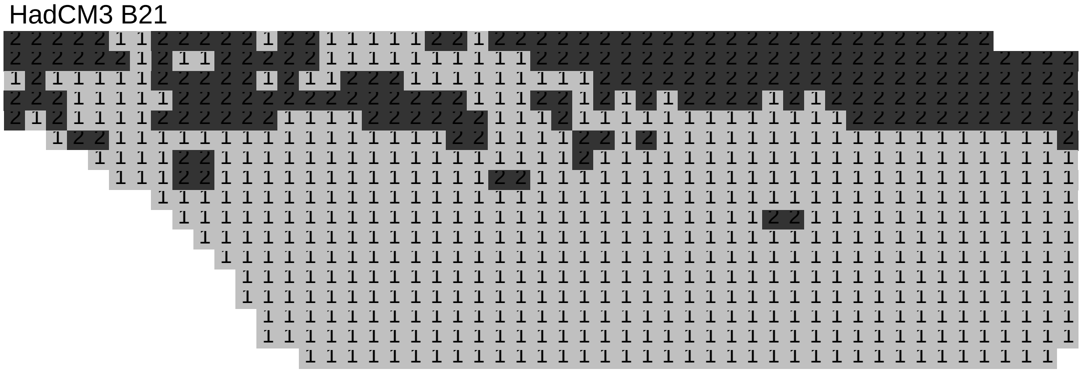

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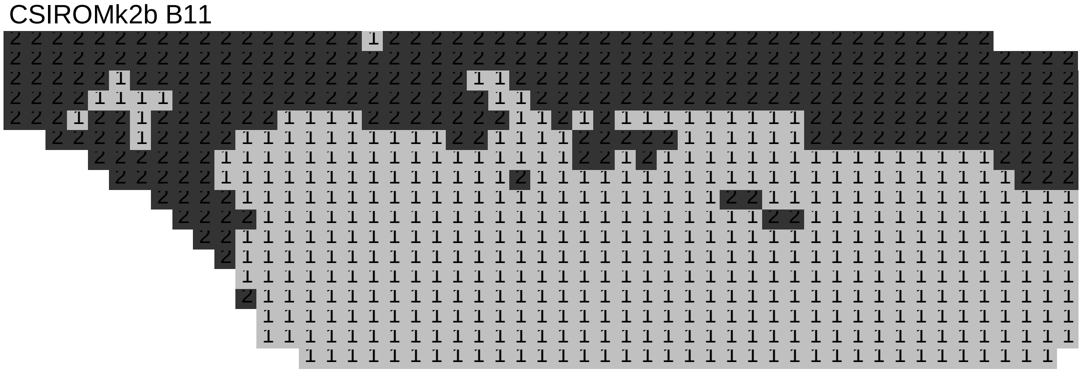

APPENDIX 3. STAKEHOLDERS' WORKSHOP, MARCH 21, 2006

- Jeff Thorpe, Saskatchewan Research Council, Saskatoon; 306-933-8172; thorpe@src.sk.ca
- John Vandall, Saskatchewan Environment, Regina; 306-787-8598; JVandall@serm.gov.sk.ca
- Rob Wright, Saskatchewan Environment, Regina; 306-787-2914; RWright@serm.gov.sk.ca
- John Thompson, Saskatchewan Environment, Prince Albert; 306-953-2343; JThompson@serm.gov.sk.ca
- Norm Henderson, Prairie Adaptation Research Collaborative, Regina; 306-337-2292; henderno@uregina.ca
- Darryl Sande, Weyerhaeuser Canada, Prince Albert; 306-953-1793; darryl.sande@weyerhaeuser.com
- Bruce Hesselink, PFRA, Indian Head; 306-695-5118; hesselinkb@agr.gc.ca
- Paul Weedon, Saskatchewan Forest Centre, Prince Albert; 306-765-2856; pweedon@saskforestcentre.ca
- Roman Orynik, Saskatchewan Forest Centre, Prince Albert; 306-765-2858; rorynik@saskforestcentre.ca
- Angus Carr, Saskatchewan Forest Centre, Prince Albert; 306-765-2855; acarr@saskforestcentre.ca
- Mike Bendzsak, Saskatchewan Forest Centre, Prince Albert; 306-765-2869; mbendzsak@saskforestcentre.ca
- Gene Kimbley, Prince Albert Model Forest, Prince Albert; 306-922-1944; gkimbley@sasktel.net
- Barb Thomas, Genstat Consulting, Edmonton; 780-432-4230; bthomas@ualberta.ca; thomasba@alpac.ca

AGENDA:

8:30 a.m.	Welcome – introductions – objectives for the day	Jeff Thorpe/Norm Henderson/John Vandall
8:45 a.m.	Current status of climate change and response by Canada and Saskatchewan	John Vandall
9:00 a.m.	Ecological threats from introducing exotic trees	Jeff Thorpe
10:15 a.m.	BREAK	
10:45 a.m.	Introducing exotic trees as an adaptation to climate change	Norm Henderson
11:45 a.m.	LUNCH (provided	
1:00 p.m.	Breakout groups will be assigned to answer a series of questions about future policy for introduction of exotic trees.	All
2:30 p.m.	BREAK	
3:00 p.m.	Reports from breakout groups and discussion	Jeff Thorpe/Norm Henderson/John Vandall
4:00 p.m.	FINISH	

SUMMARY OF ISSUES:

INTRODUCTION

- There is a conflict between the desire to find useful new plant species, and the increasing global agenda focused on dangers of exotic invasion.
- Introduction of exotic species is considered one of the major threats to biodiversity.
- But introduction of exotic species may be useful in adapting to climate change.

RATIONALE FOR INTRODUCING EXOTIC TREE SPECIES

- Exotic species may be more productive, easier to manage, or yield more valuable products than any native species.
- Exotic plantations may be useful in reclaiming disturbed land or afforesting grasslands.
- Wood supply from fast-growing exotic plantations may relieve pressure on native forests.
- Exotic species may even have positive ecological functions, such as providing habitat for native animals.
- Introduction of exotic species, or of genotypes within species, may be useful in adapting to climate change.
- May increase ecosystem diversity and/or resiliency.

THREATS FROM INTRODUCING EXOTIC TREE SPECIES

- Economic threats (e.g. plantation failure)
- Disease threats (e.g. exotic tree accompanied by exotic pathogen)
- Genetic threats (e.g. gene flow into native forests)
- Site degradation threats (e.g. soil acidification)
- Biodiversity threats (e.g. effects on habitat for other species)
- Aesthetic threats (e.g. change in traditional character of landscape)
- Invasion threats – considered to be the most important.

ASSESSING THE INVASION PROBLEM

- Impacts of exotic species would be of little concern if they stayed where they were planted.
- Species used in exotic forestry tend to be fast-growing and seed heavily, and are therefore likely to be invasive.
- The greatest damage results from invasive species that alter ecosystem function (e.g. forming dense canopy that excludes other species).
- There are many examples from around the world of introduced forest trees causing serious invasion problems (e.g. lodgepole pine in New Zealand).
- Most of the reported tree invasions have occurred in warmer climates than that of the boreal forest. There are few examples of invasive exotic trees in Canada, but some intentionally introduced exotic shrubs have become serious problems.
- Research has shown that one of the best predictors of which species will become invasive is invasive behaviour elsewhere. Invasiveness is also more likely for species with wide native ranges, and with high reproductive capacity.

- Invasion is most likely to occur in areas with low vegetation cover, especially disturbed sites.
- Several systems have been developed for screening proposed introductions to prevent invasion problems. In the American system, exotic tree species from other continents are considered to pose a greater threat than those that are native to other parts of North America.

POLICY ON INTRODUCTION OF TREE SPECIES

World policy experience

- Conservation organizations advocate the use of native species
- Policy guidelines of the International Union for the Conservation of Nature say that no exotic species should be used in natural habitats, and that exotic species should be introduced into semi-natural habitats only when there are exceptional reasons for doing so. For species meeting these criteria, the IUCN recommends detailed benefit/risk assessments and controlled field trials prior to widespread introduction.
- Most governments do not have strong policies against introduction of exotic species
- U.S. legislation provides lists of prohibited species (i.e. species are presumed innocent until proven guilty). Conservationists have argued that the emphasis should be shifted to lists of allowed species (i.e. species are guilty until proven innocent).
- Sweden has gradually increased restrictions on planting of the exotic lodgepole pine.
- Recent policy in South Africa and New Zealand has placed the legal onus on those introducing exotic species to prevent their spread to adjacent land.

Canadian policy experience

- Canadian legislation is largely aimed at plant diseases, not at plants themselves.
- Provincial weed acts are aimed at agricultural weeds.
- Policy for provincial forests generally requires regeneration of native trees following timber harvesting.

A different perspective: introduction of exotic species for adaptation to climate change

- The new ecosystems that result from climate change can be expected to unlike what we see now and probably different from ecosystems seen previously.
- Climate change may require abandoning the laissez-faire approach and assisting the movement of species to newly suitable habitats.
- The idea of protecting representative examples of natural ecosystems may become meaningless, and be replaced by focus on maintaining resilience, diversity and connectivity.
- The effects of exotic species should not be assumed to be negative; they may make positive contribution to ecosystem function and integrity.
- The key question is not whether species is exotic, but whether it contributes to biodiversity preservation, or causes problems because of exponential population growth.
- If the object is helping the system adjust to climate change, it may be desirable to use invasive species (those that will reproduce successfully in the new environment).

AFTERNOON DISCUSSION ON POLICY QUESTIONS

QUESTION: What is the current policy?

Indian Reserves: are there any limits on exotics? Probably not?

Saskatchewan FMAs: 20 year management plan assumes regeneration with native species. Widespread planting of exotics would require changing the 20-year plan, which would require new environmental assessment and public consultation. There is provision for experimental planting of extoics.

Regulations under Saskatchewan's Forest Resource Management Act say that a permit would be needed to use exotics in reforestation.

In Alberta FMAs, there is no approval for using exotics for commercial reforestation—only for research plots. Policies are laid out in the Alberta Tree Improvement Standards.

Saskatchewan crown agricultural lands: forests on these lands are managed by Saskatchewan Environment, so their policies would apply. They probably would not allow exotic plantations.

Private land: generally no restrictions. However, GMO trees are restricted by the Canadian Food Inspection Agency, and would require approval to use on any kind of land. This would apply to any species listed by CFIA, but there probably are no tree species listed. A tree species could be restricted by CFIA if there is a disease or insect associated with it.

In Saskatchewan, an exotic plantation could be considered a development that would require environmental impact assessment.

QUESTION: What should be the general policy on introduction of exotic species (by any definition) in the western boreal forest?

- **Prohibit exotic introductions**
- **Allow exotic introductions in some situations**
- **Allow exotic introductions anywhere**
- **Actively encourage exotic introductions**

The consensus of the group was: allow exotic introductions in some situations (depending on what the management objectives are).

However, some members of the public would advocate prohibiting exotic introductions.

The response to this question may depend on whether you accept that climate change is happening. You are more likely to agree that we should actively encourage exotic introductions is you agree that climate change is happening.

QUESTION: Should the policy be different depending on land tenure and the type of land management and objective?

- **Parks and protected areas**
- **Crown forest land used for commercial forestry**
- **Crown agricultural land (e.g. grazing leases, community pastures)**
- **Private land**

For invasive species, if you allow it on one kind of land you allow it anywhere—so you should not differentiate in terms of prohibiting invasive species.

But can you infringe on private land rights? You can if the species qualifies as a noxious weed.

The public does differentiate, e.g. they accept herbicides on private land but not public land.

QUESTION: The example of a proposed hybrid poplar plantation was used to further explore this question. Responses from the group:

Protected areas:

- one opinion: prohibit on protected areas because they are ecological benchmarks.
- second opinion: there may be situations even in protected areas where it might be acceptable, such as reclamation of a contaminated area.
- the first opinion can be used as the rule, while the second might be an allowed exception.

FMAs:

- acceptable for research
- if it passes the assessment based on the research, then it is acceptable to plant with some restrictions (e.g monitoring)
- another opinion: it is acceptable if its purpose is to replace natives that won't survive because of climate change. If it's just for fibre production, it is not acceptable.

Private land – acceptable; generally do not interfere with decisions on private land, except in the case of invasive species.

Other comments on the proposed hybrid poplar plantation:

- Genetic pollution is not likely to be a big impact, because of low seed viability, narrow germination requirements and short lifespan of seed, and dilution by larger quantity of native seed.
- There is lower risk tolerance on crown land than on private land.
- Planting a mixture of genotypes lowers the risk of catastrophic failure.

QUESTION: Should the policy be different depending on the origin of the exotic species?

- **Different genotypes of same species**
- **Species that are exotic to the region but native to other parts of North America**

- **Hybrid on N.A. with Eurasian species**
- **Species that are exotic to North America (Eurasian species)**

One opinion: We should not differentiate by continent of origin. A circumpolar boreal species may be more acceptable than a North American species from another ecoregion.

Moving seed or pollen from another part of the world presents a lower risk than moving plants, because of transport of diseases.

Genetic relatedness may increase acceptability

QUESTION: Should exotic species proposed for introduction be subject to a screening process?

- **costs versus benefits**
- **potential invasion problems**
- **differentiate between invasive and non-invasive exotics**

Yes, they should be.

What about private land? One opinion: there should be screening even on private land. This could be applied by working with nurseries which are the source of the planting material.

In Saskatchewan, species can be restricted from entering the province by minister's order under FRMA

QUESTION: Should we favour non-invasive exotics, or target exotics we expect to spread and be self-sustaining?

The answer depends on your objectives. From a tree improvement perspective, it is better if it doesn't regenerate. From an adaptation to climate change perspective, it is better if it does.

Even if regeneration is considered desirable, experimental trials are still needed to find out how it grows, how invasive it is, whether it breeds into the native species.

QUESTION: Should individuals or agencies that introduce exotic trees be liable for control of spread away from the planting site?

On FMAs, the province approves their plan. So is the company or the province liable for control? This would be subject to negotiation between company and government

There is a need for ongoing risk assessment, updating ratings of species, defining thresholds for rejecting a species.

QUESTION: In the light of climate change, should we openly and officially abandon as impractical the objective of maintaining the boreal forest as it is in terms of species composition?

One opinion: we already have.

Another opinion: we could be wrong about climate change. Maybe it won't change as much as we think. The forest ecosystem has a lot of natural variation, and we do not know whether it is changing outside that natural range of variability.

ADDITIONAL COMMENTS

The report should identify gaps in knowledge.

Recommendations should relate to management objectives.

There should be consideration of different perspectives on risk.

In Saskatachewan, FMA holders don't want to be perceived as proponents of exotic plantations. They would need strong reasons to even consider planting exotics. But in Alberta, it is the forest industry that wants the opportunity to plant hybrid aspen or hybrid poplar on crown land.