Large-Scale Conversion of Forest to Agriculture in the Boreal Plains of Saskatchewan

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Abstract: *Despite widespread recognition of the importance of forest loss and fragmentation on biodiversity, the extent and rate of forest loss even in temperate regions remains poorly understood. We documented forest loss and assessed whether road density, rural developments, land quality for agriculture, and land ownership influenced the distribution and rate of change in forest cover for the entire boreal transition zone (49,846 km^2) of Saskatchewan, Canada. We used landscape data taken from the Canadian Land Inventory database in forest cover (1996) and Landsat thematic mapper data (1994) to study changes between 1966 and 1994. Forest covered 17,873 km^2 of the study area in 1966 and 13,504 km^2 in 1994. This represents an overall conversion of 73% of the boreal transition zone in Saskatchewan to agriculture since European settlement and an annual deforestation rate of 0.89% over the last 28 years, a rate approximately three times the world average. Mixed-regressive, spatially autoregressive models explained a considerable portion of the variation in forest cover (r^2 = 0.83) remaining on the landscape and suggested that less forest remained on lands that (1) were privately owned, (2) had soils with high suitability for agriculture, (3) had high road density, and (4) were in the southern portions of the study area. Strong spatial autocorrelation in the data indicated that areas of remaining forest tended to be spatially clustered. Our ability to predict where deforestation occurred between 1966 and 1994 was poor when we excluded the spatial autocorrelation terms from our model, but it was clear that deforestation was more likely to occur on privately owned lands than on those managed by the provincial government. Despite dramatic changes to forested areas in the boreal transition zone, and despite the importance of this area to a wide variety of forest-dwelling wildlife, no programs are in place to slow or halt deforestation.*

Conversión a Gran Escala del Bosque a Agricultura en las Llanuras Boreales de Saskatchewan

Resumen: *A pesar del amplio reconocimiento de la importancia de la pérdida de bosques y de la fragmentación sobre la biodiversidad, la extensión y la tasa de deforestación aún en regiones templadas permanece poco entendida. Documentamos la pérdida de bosques y evaluamos si la densidad de caminos, el nivel de urbanización en zonas rurales, la calidad del suelo para la agricultura y el régimen de propiedad privada influyeron en la distribución y la tasa de cambio de cobertura forestal en la zona de transición boreal (49,846 km^2) de Saskatchewan, Canadá. Utilizamos paisajes clasificados según la base de datos del Inventario Canadiense de Tierras (1966) y datos de mapas temáticos Landsat (1994) para estudiar los cambios en la cobertura forestal entre 1966 y 1994. En el área de estudio había 17,873 km^2 de cobertura forestal en 1966 y 13,504 km^2 en 1994. Esto representa una reconversión total del 73% de la zona de transición boreal en Saskatchewan a la agricultura desde la colonización europea y una tasa anual de deforestación del 0.89% en los últimos 28 años, una tasa aproximadamente tres veces mayor que el promedio mundial. Los modelos de regresión mixta, espacialmente autoregresivos explicaron una parte considerable de la variación en la cobertura forestal (r^2 = 0.83) remanente en el paisaje y sugirieron que hay menos bosque en terrenos que (1) eran propiedad privada, (2) tenían suelos adecuados para la agricultura, (3) tenían una alta densidad de caminos y (4) estaban en la porción sur del área de estudio. La robusta autocorrelación espacial de los datos*

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Paper submitted May 1, 2001; revised manuscript accepted January 23, 2002.

indicó que las áreas de bosque remanente tendían a estar espacialmente agregadas. Nuestra capacidad para predecir donde ocurrió la deforestación entre 1966 y 1994 fue pobre cuando excluimos de nuestro modelo los términos de autocorrelación espacial. Sin embargo, hubo una mayor probabilidad de deforestación en terrenos de propiedad privada que en los administrados por el gobierno de la provincia. A pesar de los cambios dramáticos en las áreas forestales de la zona de transición boreal, no hay programas para disminuir la deforestación, no obstante la importancia de esta área para una amplia variedad de vida silvestre que habita los bosques.

Introduction

Loss and fragmentation of forests is a major global conservation issue (Harris 1984; Hunter 1990; Myers 1996). Although considerable research on the effects of these processes has been conducted, rates of forest loss and the anthropogenic and environmental factors that influence loss and fragmentation remain poorly understood (Turner et al. 1996; Wickham et al. 2000). There is a need for information on why forested landscapes change, and how anthropogenic and ecological factors interact to influence forest loss and fragmentation (Turner 1987; Baker 1989; Dale et al. 1993; Wear & Flamm 1993; Spies et al. 1994; Turner et al. 1996). Evaluating factors influencing changes to forested landscapes increases the likelihood that integrated management for sustainable human use and the maintenance of environmental values will be attained. However, this paradigm of ecosystem management has been considered primarily for forested systems where the major anthropogenic disturbances are forestry, recreation, or residential development (Lee et al. 1992; Wickham et al. 2000). It is less clear whether ecosystem management strategies can be achieved in forested areas undergoing conversion to agriculture (Whitney & Somerlot 1985).

Attempts to determine rates of deforestation at large spatial scales have been conducted mainly in tropical areas (e.g., Dirzo & Garcia 1992; Dale et al. 1993; Fox et al. 1995; Stoorvogel & Fresco 1996). Less is known about deforestation of temperate areas of North America (Zipperer 1993; Kress et al. 1996; Zheng et al. 1997), particularly the boreal forest of western Canada. Dramatic changes to forestry policy in this region over the last two decades have generated a new awareness of the potential risks to biodiversity involved with large-scale landscape change (Cummings et al. 1994; Stelfox 1995). Unlike forestry, agricultural development in western Canada is largely unregulated, unencumbered with provincial operating rules, and more likely to result in permanent conversion of forest habitat. As early as the 1950s, Davidson (1952) recognized the risk to the boreal forest posed by rapid expansion of agriculture along its southern border. Agricultural policy in this region has not changed significantly over the past 50 years, and direct conversion of forested lands to cereals, oilseeds, and pasture continues unabated.

Understanding the factors driving landscape change in the boreal forest is critical for the management of many wildlife species, particularly birds. More than 200 species of birds breed in the southern boreal forest of western Canada (Robbins et al. 1986; Smith 1993; Price et al. 1995). Many (45%) of these species are long-distance Neotropical migrants, several of which are undergoing long-term declines (Askins et al. 1990; Bohning-Gaese et al. 1993). For species such as the Tennessee Warbler (*Vermivora peregrina*), Cape May Warbler (*Dendroica tigrina*), and Connecticut Warbler (*Oporonis agilis*), the western boreal forest represents almost all of their breeding range. Other species such as the Ovenbird (*Seiurus aurocapillus*) seem to reach much higher densities in the southern boreal forest of western Canada than they do in other forest ecosystems in North America (Bayne 2000). Clearly, the boreal forest is a critical biome for forest birds in North America and warrants considerably more attention than it has previously received.

The objectives of our study were to determine the amount of forest cover remaining in the boreal transition zone of Saskatchewan, establish rates of deforestation through retrospective analyses, and generate predictive models to evaluate which anthropogenic and environmental factors correlate with current forest pattern and loss. We hypothesized that land ownership would have a strong influence on landscape structure and land-cover change, with privately owned lands being more likely to be converted to agriculture. To test this hypothesis, we compared the amount of forest cover in the Canada Land Inventory (CLI) database (circa 1966) to that determined through Landsat imagery taken in 1993 and 1994. We also tested whether agricultural suitability (as indicated by soil quality, slope, and nutrient level), road density (accessibility), distance to nearest town (market distance), and spatial location could predict where remnant patches of forest were located in the landscape and where deforestation was most likely to occur.

Methods

Study Area

Our study area was within the mixed-wood section of the southern boreal forest of Saskatchewan (Kabzems et al. 1986) and extended from approximately lat. 54°N, long. 110°W, on the Saskatchewan-Alberta border to lat.

52°N, long. 102°W, along the Saskatchewan-Manitoba border (Fig. 1). We examined 49,846 km^2, comprising 95% of the 52,357 km^2 portion of the boreal transition ecozone of Saskatchewan between the Aspen Parkland and the commercial forest (Padbury & Acton 1994).

Upland forests in this area are typically a combination of trembling aspen (*Populus tremuloides*), white spruce (*Picea glauca*), jack pine (*Pinus banksiana*), and, to a lesser extent, balsam poplar (*Populus balsamifera*), white birch (*Betula papyrifera*), and balsam fir (*Abies balsamea*). Black spruce (*Picea mariana*), tamarack (*Larix laricina*), swamps, bogs, and fens occur in lowland areas. Native grassland comprised a small but unknown proportion of the pre-settlement landscape.

Historically, fire was the dominant disturbance regime in this area, with a pre-settlement fire interval of approximately 30 years (Weir & Johnson 1998). Fire suppression has increased the fire interval to about 200 years in the most heavily forested areas (Weir et al. 2000). Approximately 124,000 people live in the boreal transition ecozone (Acton et al. 1998). Land clearing for agriculture began in the late1800s by Metis settlers, with agricultural production records beginning in 1916. Agriculture is one of the major sources of employment in the region, with 11,500 farms having gross farm receipts of $740 million in 1990 (Statistics Canada 1993).

GIS Analysis

We obtained 98 digital map sheets created at a 1:50,000 scale from the Saskatchewan Digital Land Cover project. Data for this project came from Landsat thematic mapper (Landsat) bands 3, 4, and 5 taken from imagery collected between 1993 and 1994. Imagery was classified to a maximum of 25 cover types and geometrically rectified and cut on a national topographic series (NTS) basis (MacTavish 1995). Error assessment of two map sheets in the study area showed land-cover accuracy to be 97.6% and 97.7% for map sheets 73G/10 and 73G/15, respectively (MacTavish 1995).

Manipulation and querying of the classified imagery was done with Arcview (version 3.1, Environmental Systems Research Institute, Redlands, California). We reclassified the data so that all shrubs, bogs, and productive and nonproductive forests were considered forest cover. Cultivated lands, grasslands, farmsteads, and other anthropogenic disturbances (e.g., roads and towns) were classified as agriculture. Water was retained as a separate class.

Historical land-use data were obtained from the CLI digital coverage created around 1966. Data from the CLI were obtained from aerial photography and classified at a 1:50,000 scale (McClellan et al. 1967), but digital coverage of the final CLI land-use data set was available only at a

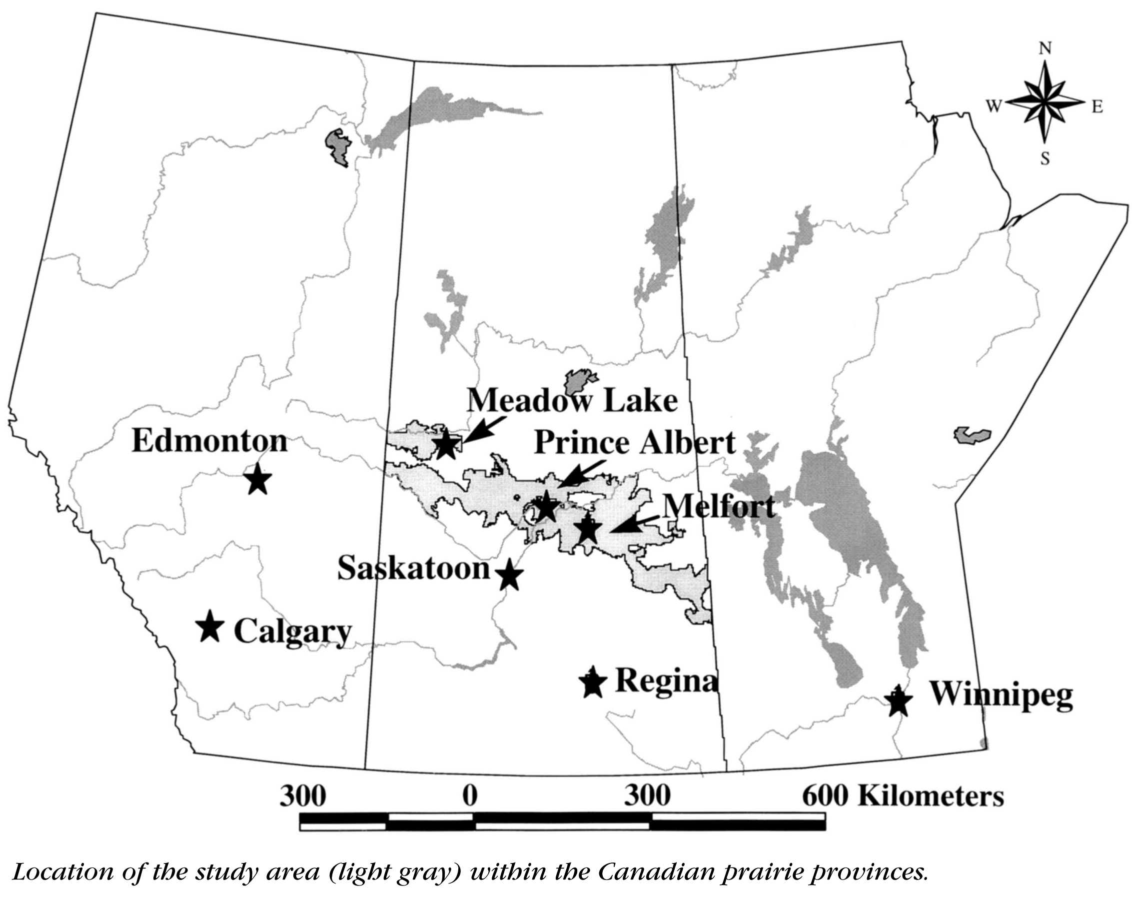

Figure 1. Location of the study area (light gray) within the Canadian prairie provinces.

scale of 1:250,000. Although not as fine as the 1:50,000 scale of the Landsat data, the CLI is the only accessible source of land-cover data on a broad scale for western Canada. To ensure compatibility between CLI and Landsat data, we compared data digitized from seven 1:50,000-scale NTS maps from a time period similar to that of the CLI data set (1957–1963). An overlay of 1000-ha hexagons was used to extract samples from the NTS and CLI data sets. For each map sheet, 30 hexagons were randomly selected for analysis. A total of 210 hexagons were selected for comparison.

To limit our study to the boreal transition zone, we used overlays to remove data not falling within the desired study area. We considered the boreal transition zone as all land between the commercial forest and the Aspen Parkland, which included the Turtle Lake Uplands (Padbury & Acton 1994). The northern boundary of the boreal transition zone was defined by boundaries of the forest management agreement (FMA), which are zoned for commercial forestry. The southern limit of the study area was the northern limit of the Aspen Parkland and was delineated with a georeferenced digital map from Padbury and Acton (1994).

To determine which environmental and anthropogenic factors influenced deforestation, we obtained georeferenced data on the location of roads, towns, rural municipalities, Indian reserves, and lands managed or leased by the provincial government. These data are part of the Saskatchewan Base Map data set, which was created at a scale of 1:50,000 from aerial photography collected between 1980 and 1995. To analyze which factors influenced forest cover and deforestation, we subdivided the study area into a continuous hexagon layer using cells of 50 km^2. The 685 full-sized hexagons covered 70% of the total study area. For each data set we calculated the percentage of forest cover within each hexagon, the percentage of privately owned land, the distance from the hexagon centroid to the nearest town of more than 100 people, and the total length of roads within the hexagon. Using the georeferenced land-quality scale provided in the CLI database (Table 1), we also computed the percentage of land suitable for cultivation (classes 1–4) in each hexagon.

Agricultural Statistics

We obtained tabular data on total area of land in agricultural production from Statistics Canada's Census of Agriculture, published every 5 years from 1916 to 1996. Total acreage of agricultural land reported per rural municipality was linked with our georeferenced data set so that only the 45 rural municipalities whose centroid was in the study area were examined. Because each rural municipality differed in size, we calculated the total percentage of land not in agricultural production. Land not in agricultural production included forest, water, native ungrazed prairie, and abandoned fields (hereafter termed nonagricultural land). The time series was incomplete for six of the northern-most rural municipalities and were excluded from our estimates.

Annual percent change in the amount of forest cover or nonagricultural land was calculated as

$$r_t = [(A_e - A_s)/A_s]\,(t^{-1})\,(100), \qquad (1)$$

where A_e is amount of land in a particular land use at the end of the period of observation, A_s is that amount at the start of the period of observation, and t is number of years between observations.

Statistical Analysis

To ensure that our deforestation rate estimates were not biased by differences in scale between the Landsat and CLI, we compared forest cover in the NTS and CLI data sets with a two-tailed Student's t test for paired samples. Comparisons were made only on the 202 hexagons that contained forest cover.

In large-scale landscape analyses, the independence of observations in close spatial proximity is questionable. Prior to analysis, we computed Moran's spatial autocorrelation statistic for each variable and found significant positive spatial autocorrelation for most variables. Therefore, rather than use ordinary least-squares regression to develop predictive models, we used the program Spacestatpack to develop mixed regressive spatially autoregressive (MRSA) models (Pace & Barry 1997). For each dependent variable (forest cover in 1994, forest cover in 1966, and annual rate of change), an MRSA model was fit that included amount of privately owned land (km^2), total length of roads (km), amount of land suitable for cultivation (km^2), distance to the nearest town (km), and northing and easting coordinates. The model was built like a standard ordinary least-squares model, except that weightings from a spatial-weight ma-

Table 1. Land-quality classes used in the Canadian Land Inventory database to define the suitability of particular areas for agriculture.

Land-suitability class	*Limitations to agriculture*
1	no significant limitations
2	moderate limitations with moderate conservation practices required
3	moderate-severe limitations, only a limited range of crops can be planted
4	severe limitations to agriculture
5	forage crop improvement practices feasible
6	forage crop improvement not feasible
7	no capability for cropping or permanent pasture
8	unmapped areas
O	organic soils

trix (D) were applied to each parameter (Pace & Barry 1997). We fit the model by using 2, 4, 6, 8, 10, 12, and 16 neighbors to calculate D, choosing the model with the highest maximum likelihood as that with the best fit. In MRSA models, the spatially lagged variation in the dependent variable is also included. Estimation of MRSA parameters in Spacestatpak utilizes a maximum-likelihood routine. We calculated the significance of each independent variable by removing terms individually from the model and comparing model fit through likelihood-ratio tests. These tests involved two degrees of freedom because both the independent variable and its spatial weighting were removed from the model. Percent cover variables were arcsine-transformed, and all other variables were log-transformed. All data are reported as raw means ± 1 SD unless otherwise reported. Data are considered significant at $p = 0.05$.

Results

Scale of Data Set

There was no significant difference in the amount of forest cover between the 1:50,000–scale NTS and the 1:250,000–scale CLI datasets (NTS mean = 486.75 ha, CLI mean = 486.88 ha; df = 201, $t = 0.01$, $p = 0.992$). We therefore concluded that using data at a 1:250,000 scale would not unduly bias our estimates of deforestation rates.

Forest Cover

In 1994, 13,504 km^2 (27.1%) of the study area was forest (Fig. 2b). Water covered approximately 1812 km^2 (3.6%) of the study area, with the remaining 34,530 km^2 (69.3%) converted to agriculture or other human developments. The average amount of forest per 50 km^2 hexagon was 12.0 $\pm$ 10.2 km^2 (24%). The remaining forest was not equally distributed across the study area. The MRSA model that fit best controlled for spatial autocorrelation within a six-hexagon neighborhood. Percentage of forest within a hexagon was negatively correlated with percentage of privately owned lands ($\chi^2 = 93.8$, df = 2, $p < 0.001$). Of the 37,839 km^2 of privately owned land, 18.9% remained as forest. In contrast, 42% of the 1870 km^2 of Indian reserves and 55% of the 10,137 km^2 of lands managed or leased by the provincial government were forested (Fig. 3a). Percentage of land within a hexagon suitable for cultivation was negatively correlated with percentage of forest cover ($\chi^2 = 50.8$, df = 2, $p < 0.001$). Of the 9521 km^2 of lands not suitable for cultivation, 50% were forested. In contrast, 22% of the 40,324 km^2 of lands suitable for cultivation retained forest cover (Fig. 3b). Road density was negatively correlated with percentage of forest cover ($\chi^2 = 16.2$, df = 2, $p < 0.001$). More forest existed in the northern portion of the study area ($\chi^2 = 18.6$, df = 2, $p < 0.001$). Removing the locally averaged spatial variation reduced the fit of the overall model (r^2) from 86% to 62% ($\chi^2 = 354.4$, df = 8, $p < 0.001$).

In 1966, 17,873 km^2 of the study area was forested (35.9%), and 31,435 km^2 existed as agricultural lands (63.1%; Fig. 2a). Water covered 519 km^2 of the study area (1.0%; see below). The average amount of forest per 50 km^2 hexagon was 15.7 $\pm$ 10.2 km^2 (32%). Percentage of forest within a hexagon in 1966 was negatively correlated with percentage of privately owned land ($\chi^2 = 52.8$, df = 2, $p < 0.001$; Fig. 3a) and percentage of land suitable for cultivation ($\chi^2 = 53.8$, df = 2, $p < 0.001$; Fig. 3B) and was positively correlated with distance to the nearest town ($\chi^2 = 17.0$, df = 2, $p < 0.001$) and latitude ($\chi^2 = 20.6$, df = 2, $p < 0.001$). Removing locally averaged spatial variation reduced the r^2 from 83% to 59% ($\chi^2 = 368.4$, df = 8, $p < 0.0001$).

Deforestation

Between 1966 and 1994, 4368 km^2 of forest was lost, resulting in an annual rate of change in forest cover of -0.87% for the entire study area. Overall, 69% of the 685 hexagons examined lost forest cover (Fig. 2c), with 3.7 $\pm$ 6.5 km^2 of forest lost per hexagon. In total, 211 hexagons increased in forest cover, with 71% of these gaining more than 1 km^2. However, 31 hexagons had no forest in the CLI but averaged 2.8 $\pm$ 1.9 km^2 of forest in the Landsat images. For these hexagons, the mean patch size was 1.67 $\pm$ 1.18 ha, suggesting that small fragments were not recorded in the CLI. The 31 hexagons with no forest cover in the CLI were excluded from our MRSA analyses because the rate of change could not be estimated.

The annual rate of change in forest cover was negatively correlated with percentage of privately owned land. The annual rate of change was -1.13% on privately owned lands and -0.54% on public lands ($\chi^2 = 7.8$, df = 2, $p = 0.02$; Fig. 3a). Road density, suitability of land for cultivation, distance to nearest town, latitude, and longitude were not significant predictors of the annual rate of change in forest cover (all $p > 0.10$). The MRSA model accounted for 68% of the variation in the data, but the majority of the variance was explained by the strong spatial autocorrelation in the data (Fig. 2c). Removing the locally averaged spatial variation of the dependent and independent variables resulted in an ordinary least-squares model with an r^2 of only 4% ($\chi^2 = 159.8$, df = 8, $p < 0.001$).

Changes in Agricultural Production

Between 1916 and 1996, the amount of land not in agricultural production in the 39 rural municipalities examined (total area 54,222 km^2) decreased from 44,219 to 17,646 km^2 (Fig. 4). This resulted in an annual rate of

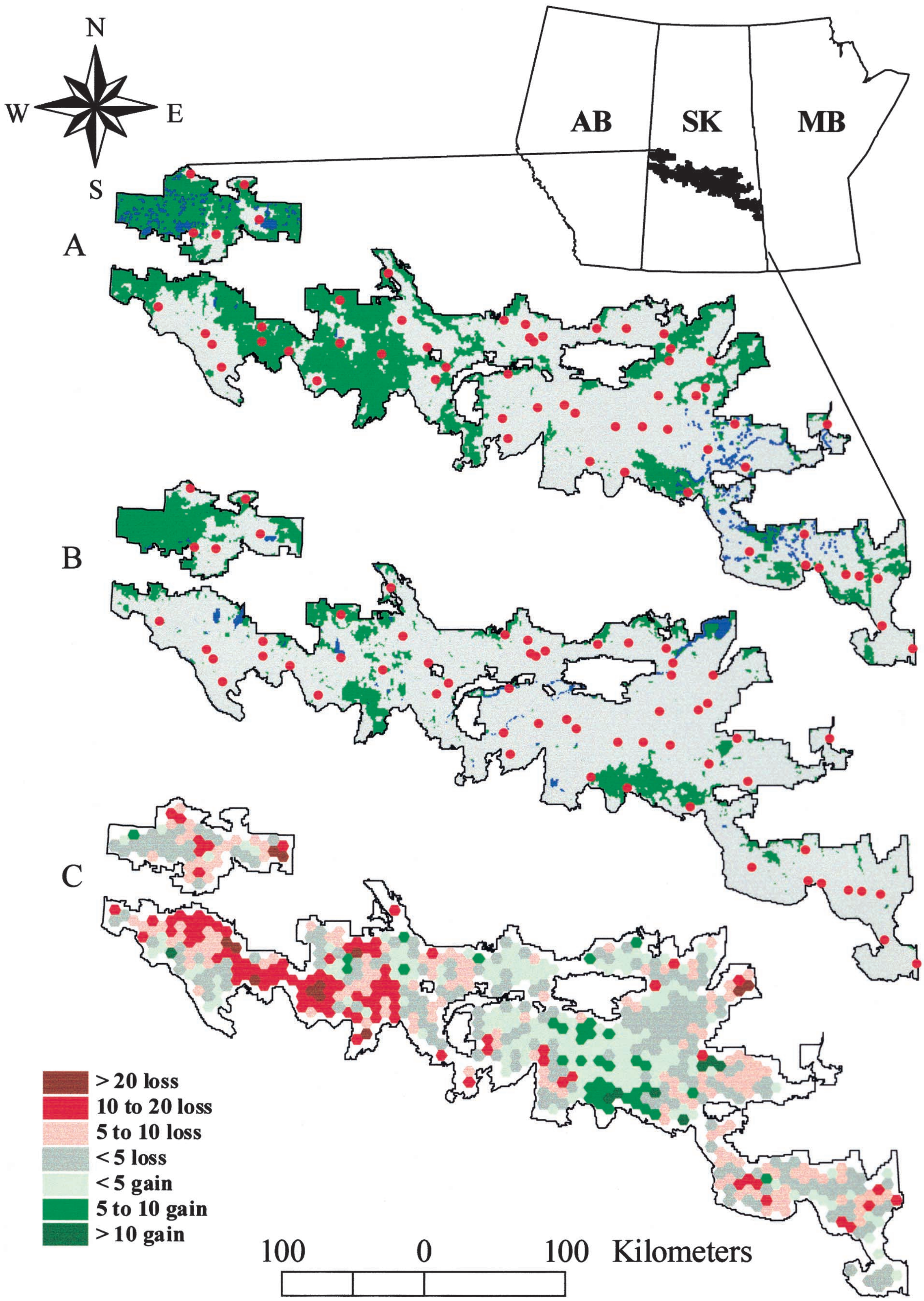

Figure 2. Study-area landscape in (a) 1966 and (b) 1994 and (c) absolute change in the amount of forested land (km^2) per 50-km^2 hexagon between 1994 and 1966. Location of the study area within the prairie provinces is shown in the inset. All towns of >100 people are shown as red circles on (a) and (b).

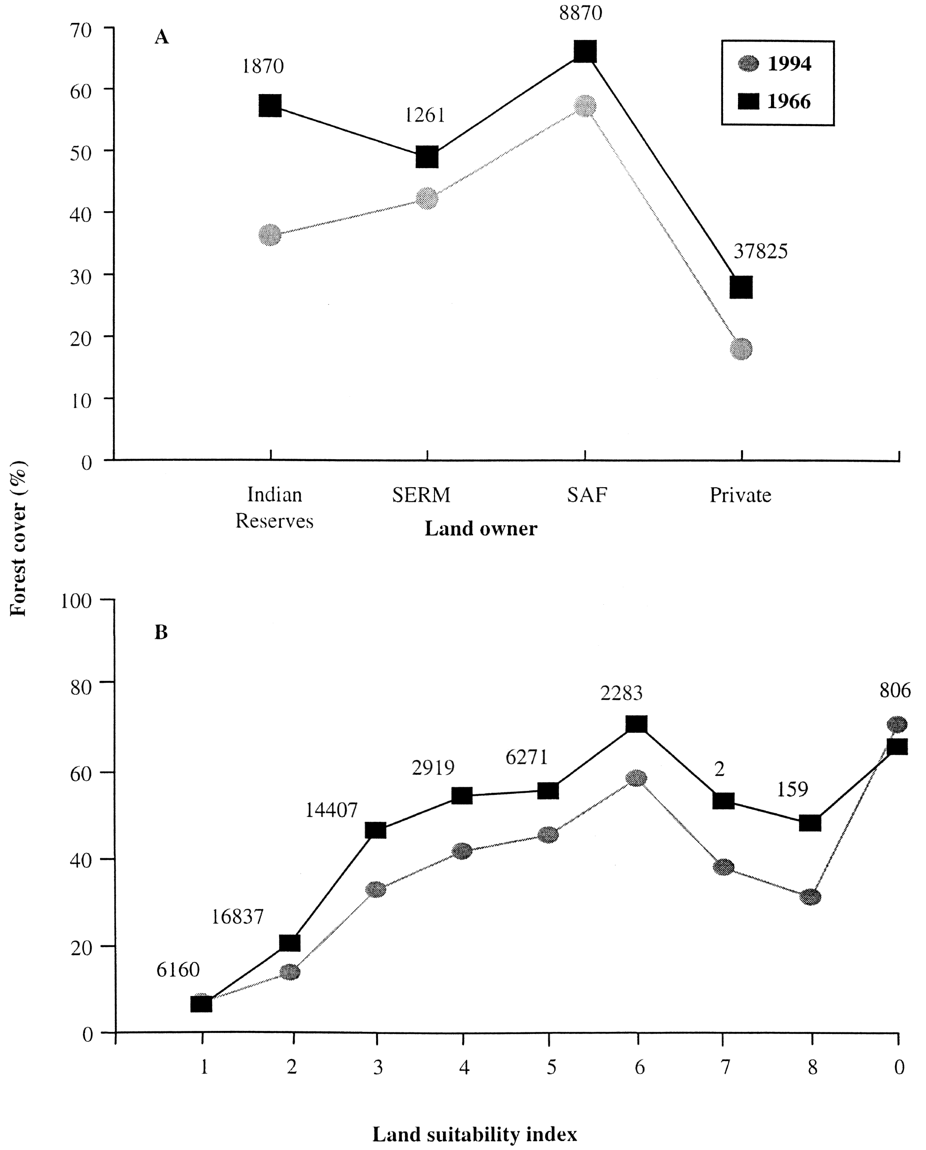

Figure 3. Percent forest cover in 1966 and 1994 in relation to (a) land-ownership class (Indian Reserves, public land managed by Saskatchewan Environment and Resource Management [SERM] or Saskatchewan Agriculture and Food [SAF], and private lands) and (b) land suitability for agriculture (1-4, suitable; 5-0, not suitable). Information on land-quality classes is given in Table 1. Total area in square kilometers of each land-ownership class is shown above the data points in (a) and (b).

change in nonagricultural land of −0.75% over the entire 80-year time series. On average, 76.9 ± 13.9% of the land area within a rural municipality was nonagricultural land in 1916, 27.8 ± 17.7% in 1966, and 26.4 ± 16.9% in 1996. Decreases in the amount of nonagricultural land and the rate of loss were not constant across time or space (Figs. 4 & 5). The annual loss of nonagricultural land was highest prior to World War II. with an annual rate of change of −1.85 ± 0.98% (Fig. 4). Most land clearing prior to World War II occurred in the central portion of the study area (Fig. 5), although all rural municipalities lost nonagricultural land during this period. Between 1946 and 1966, land clearing slowed in the central portion of the study area but increased in western and eastern portions (Fig. 5). The annual rate of change in nonagricultural land was −1.79 ± 1.18% during 1946–1966, although two rural municipalities saw an increase in the amount of nonagricultural land during this period. After 1966, the rate of land conversion slowed in most areas, with an average annual change in nonagricultural land of −0.13 ± 1.26%. Land conversion continued after 1966 in most of the study area, however, with 25 rural municipalities having less nonagricultural land in 1996 than they did in 1966 (Fig. 5).

According to the CLI-Landsat comparison at the rural municipality level, the average annual change in forest cover during the period 1966–1994 was −0.72 ± 0.94%. Only four rural municipalities gained forest cover (Fig. 5). Of these, only one reported an increase in the amount of nonagricultural land. The percentage of land within each rural municipality that was forested declined from 21.4 ± 12.3% in 1966 to 16.2 ± 9.2% in 1994. However, the absolute difference in percentage of forest within each rural municipality was not significantly correlated with percentage of nonagricultural land within each rural municipality (Pearson $r = 0.22$, $p = 0.17$), and the

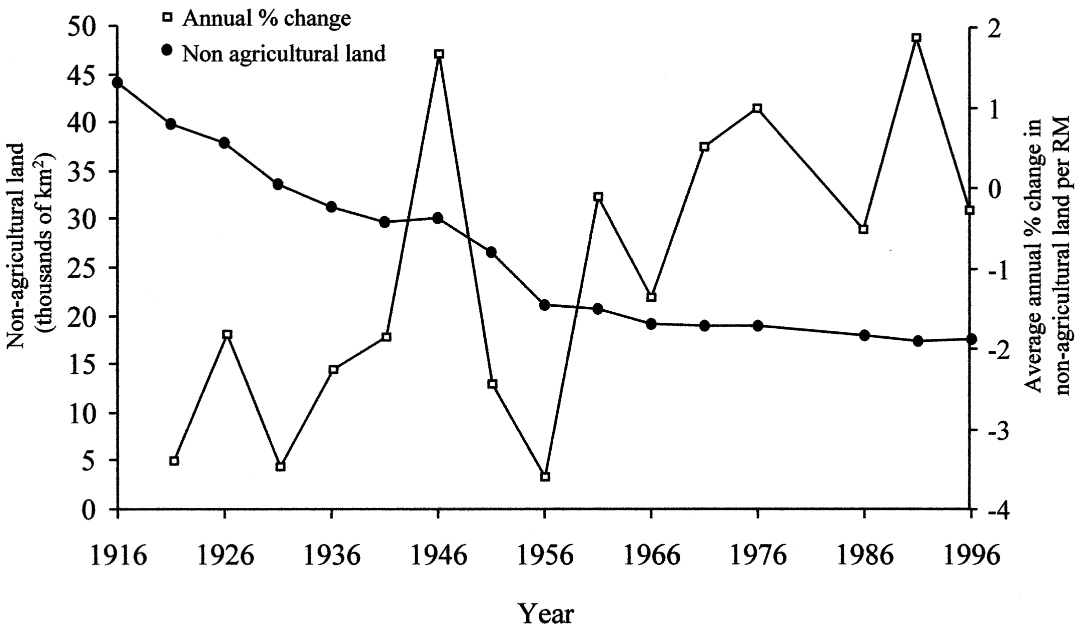

Figure 4. Total amount of nonagricultural land between 1916 and 1996. Also shown is the rate of change in the amount of nonagricultural land per 5-year period (RM, rural municipality).

rate of change in the amount of non-agricultural land was not correlated with the rate of change in the amount of forested land (Pearson $r = -0.13, p = 0.43$).

Discussion

Factors Influencing Forest Pattern and Deforestation Rates

Although the most dramatic rates of deforestation in our study area occurred prior to and just after World War II, the rate of deforestation we quantified between 1966 and 1994 was higher than the world average of 0.3% per year (Food and Agriculture Organization 1999). In the southern portions of our study area there is little forest left, with almost all arable land converted to grain or cattle production. Northern areas closer to the boundaries of the commercial forestry zone retain more forest cover and are in a less-fragmented state. In 1966, however, the more heavily forested portions of our study area, particularly in the northwest, have experienced some of the highest rates of deforestation in the world in recent times (data presented herein vs. that of the Food and Agriculture Organization [1999]). Our province-wide results are consistent with studies from across the boreal plains ecozone, which have shown deforestation rates ranging from 0.87% to 1.76% per year (Environment Canada 1991; Alberta Environmental Protection 1999; Cumming et al. 2001; Fitzsimmons 2002). These results also fall within the range found in other temperate forests worldwide (Wear & Flamm 1993; Spies et al. 1994; Turner et al. 1996; Cushman & Wallin 2000).

Land ownership and land quality significantly influenced the distribution of forest on the landscape. Privately owned had far less forest than did areas managed by the provincial government or native bands. Private landowners and native bands were more likely to clear their land between 1966 and 1994. Although lands in private ownership tended to be of higher quality for agriculture than public lands, deforestation occurred in almost every land-quality class. High-quality agricultural lands that were privately owned were also more fragmented than lower-quality public land. In part, this stems from the higher road density on private lands. The road density in our study area was extremely high and resulted in a highly patchy landscape with a great deal of forest-edge habitat (Thorpe & Godwin 1999).

The models we created were good predictors of landscape pattern and explained a large proportion of the variance in the data. On the other hand, our ability to predict deforestation was weak. Although the annual rate of deforestation was higher on privately owned land, land ownership explained relatively little of the variation in the data. Instead, spatial autocorrelation between neighboring hexagons explained most of the variation in deforestation rate, suggesting that deforestation across the region occurred in a clumped pattern.

The amount of water present in the study area differed considerably between the Landsat images and the CLI. Flooding caused by dam creation was partially responsible for these increases. In addition, much of the study area suffered from drought during the 1960s, so many prominent lakes in the study area were considerably smaller in the CLI than in the Landsat imagery. However,

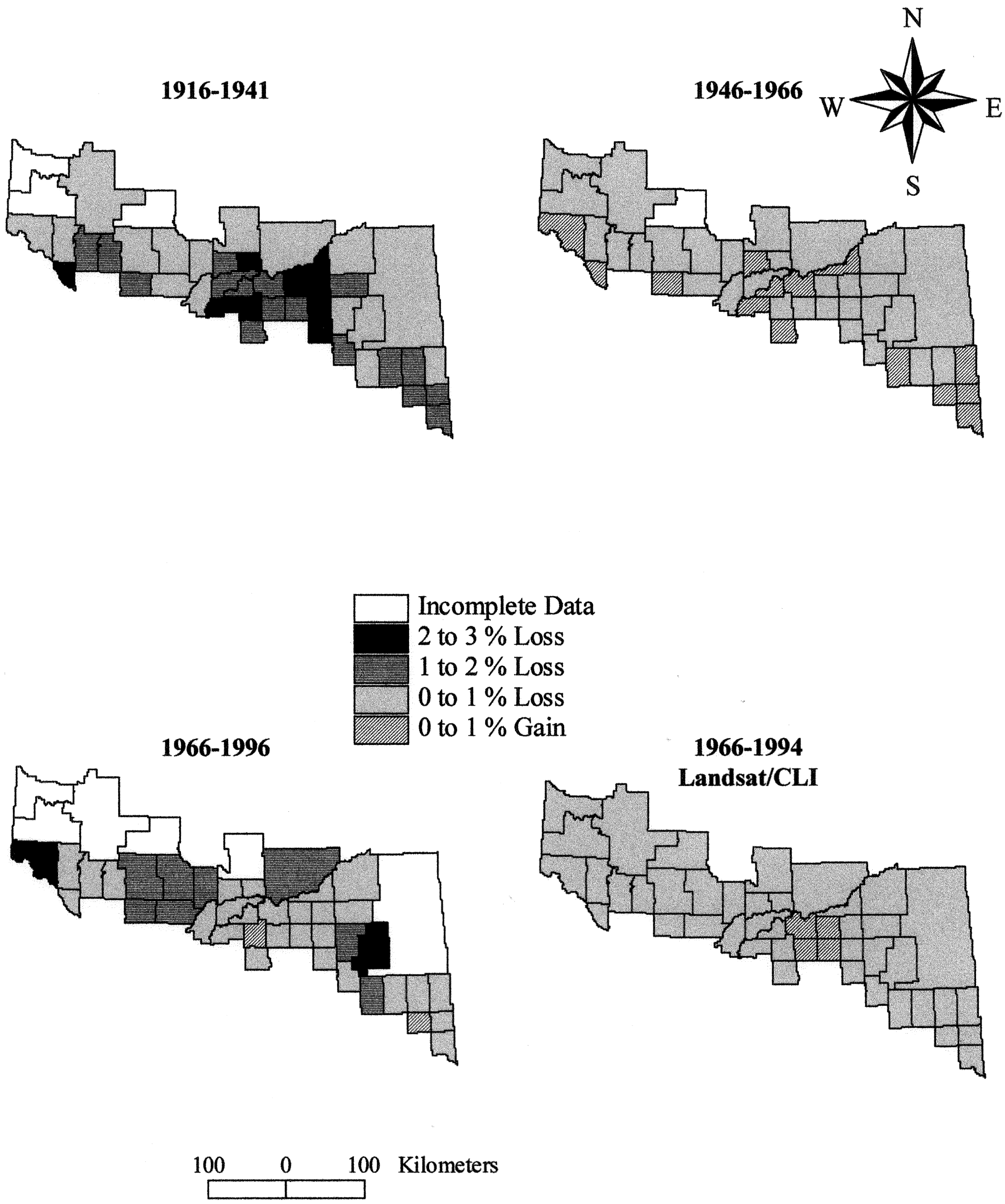

Figure 5. Annual absolute rate of change (%) per rural municipality (RM) for three time periods as determined by agricultural census data and forested land as determined by data from the Landsat thematic mapper and the Canadian Land Inventory for the period 1966–1994.

classification errors in the original CLI data may also have been a factor. For example, the full extent of the north and south Saskatchewan river system is not shown in the CLI dataset, but when we removed those portions of the 1966 study area that were covered by water in 1994, the annual rate of change in forest cover was still −0.75%.

Measures of landscape pattern are highly dependent on the grain (e.g., spatial resolution), and extent (total area) of the data (Turner et al. 1996). Although the CLI was created at a scale similar to that of the Landsat imagery, the final resolution was different enough that small patches seemed to have been missed in the CLI that were picked up in the Landsat imagery. This resulted in "reforestation" of some areas, likely because small forest patches were counted in the Landsat imagery but not in the CLI. This makes our estimate of deforestation conservative.

Conservation Implications

Although the southern boreal mixed-wood forest of Saskatchewan has undergone rapid deforestation, provincial land managers have done little to discourage conversion of forest to agricultural land. In a survey of land managers across Canada, managers in Saskatchewan showed the least concern for loss of forest along the boreal fringe (Fox & Macenko 1985). Similarly, the Saskatchewan State of the Environment Report for the Boreal Plain Ecozone did not even list deforestation by agriculture as a concern (Saskatchewan Environment and Resource Management 1995). The apparent indifference of both the government and public toward forested lands in the boreal transition zone is caused by a combination of factors, including (1) the predominant position of the conventional agriculture sector in the provincial economy, (2) lack of perceived economic and ecological value of privately owned forest, (3) the abundance of forest on public lands, and (4) economic incentives that promote direct conversion of forest to agriculture.

The notion that "wheat is king" has pervaded agricultural policy in Saskatchewan for decades. Recent programs such as the Gross Revenue Insurance Plan (GRIP) were intended to protect farmers against production losses and price fluctuations. However, GRIP may have resulted in direct conversion of forested land into marginal cropland, because pay-outs under GRIP were a function of the amount of seeded acreage multiplied by a guaranteed price per acre. The Grazing Lease Improvement Program of the 1970s provided direct payment for felling trees, breaking land, and clearing or burning bush. The imprint of these policies can be seen on the landscape today, with privately owned lands retaining far less forest cover than public lands.

Although lands managed or leased by the provincial government retained more forest cover than land in private ownership, the quality of these forests may have been compromised because they are often used as unimproved grazing land for cattle (Thorpe & Godwin 1999). Grazing by cattle can dramatically reduce the suitability of forest for wildlife, particularly for birds nesting on the ground or in shrubs (Donald et al. 1998). Penetration of forested habitat by cattle can also have indirect effects on wildlife, including forest structural changes (Jenkins & Parker 2000) and increases in parasitism by the Brown-headed Cowbird (*Molothrus ater*) (Coker & Capen 1995; Hobson & Bayne 2000*a*).

Outside the agricultural portion of the boreal plains ecozone, Saskatchewan has one of the largest publicly owned commercial forestry zones in Canada (Farm Woodlot Association of Saskatchewan [FWAS] 1991). This large volume of available timber, in conjunction with one of the lowest stumpage fee rates in Canada as of 1996 (Canadian Council of Forest Ministers 2000) and high provincial subsidies, allow forestry companies in the region to transport wood over large distances at relatively little cost. Consequently, little effort has been put into the development of sustainable harvest of forest in the boreal transition zone. In 1990, average prices for hardwood pulp delivered to mills in our study area were about \$18.82/m^3. Economic models for the study area suggest that a traditional grain farmer would require current pulpwood prices to rise to \$33/m^3 to achieve a return to labor of \$11–12 an hour (FWAS 1991). Given the disparity between these prices and the agricultural subsidies that encourage land clearing, the most economical strategy for private landowners in our study area would be to clearcut forested areas and use the proceeds to finance the costs of preparing the land for grain or cattle production. The forest industry has capitalized on this situation, and at least 9% of all wood harvested in Saskatchewan comes from private landowners (FWAS 1991). What portion of this wood is coming from woodlots that are harvested sustainably is unknown but likely insignificant.

The rapid deforestation of the boreal forest in western Canada (Environment Canada 1991; Alberta Environmental Protection 1999) has important implications for Canada's climate-change strategy. Significant releases of carbon are associated with deforestation, particularly when land is permanently converted to agriculture (Brovkin et al. 1999; Mahli et al. 1999). Limiting deforestation of the boreal mixed woods may be one of the less expensive options for reducing the effects of global climate change (Cairns & Meganck 1994; Guy & Benowicz 1998; Peterson et al. 1999). Policies promoting reforestation of marginal agricultural lands would aid programs designed to mitigate the effects of increased carbon dioxide in the atmosphere (Canadian Forest Service 1998). Current carbon-credit initiatives by Saskatchewan Environment and Resource Management (SERM) and SaskPower will plant approximately 5 million trees (3000 ha) in areas in the commercial forest with insufficient regeneration after fire or harvesting (SERM 2000; T. Baumgartner, personal communication). Although this program represents a positive step, reforestation of agricultural lands would provide greater benefits as forests sequester between 20 and 100 times more carbon than agricultural crops and secure carbon for longer periods (Cairns & Meganck 1994; van Kooten et al. 1999).

Although we only examined factors influencing deforestation in Saskatchewan, we believe these patterns to be typical of other provincial jurisdictions in the western boreal forest. Based on the location of forest-management zones in each province, 25% of the boreal plains ecozone in Alberta, 31% in Saskatchewan, and 48% in Manitoba is at risk of deforestation from agriculture (Acton et al. 1998; Alberta Environmental Protection 1999). In addition, much of the remaining boreal plains has been leased to industrial forestry companies through long-term forest-management agreements (Cummings et al. 1994; Stelfox 1995). Land managers need to recognize the importance of this ecozone to wildlife. Much of the boreal forest in North America—the boreal shield—is dominated by largely monospecific stands of conifer that support relatively few wildlife species compared with forests in the southern boreal plains. As a result of high landscape diversity, the boreal plains attract more species of breeding birds than virtually any other forest ecosystem in North America (Smith 1993; Hobson & Bayne 2000*b*). Because current land-use practices are reducing the diversity of these forested landscapes, long-term negative effects on the avifauna of this region are expected (Hobson & Bayne 2000*a*, 2000*b*). Because parks in which logging and grazing are not permitted make up <4.3% of the boreal plains ecozone in Saskatchewan, effective conservation must be achieved outside these protected areas.

Even if dramatic changes in land-use policy occur in this region, the imprint of the current land-ownership patterns will remain far into the future (Wallin et al. 1994). Given the importance of existing conditions to the design of future landscape patterns, it is important that a land-use plan for the long term be developed for this region. Many of the rural municipalities in the southern portion of our study area are highly suitable for agriculture and have been so extensively cleared that they should be maintained as agricultural lands. Much of the agricultural land in the northern portion of our study area is of lower quality, however, and may be better suited to other land uses. Policies that encourage private landowners to retain forest cover on their land and stop direct conversion of forested land to agriculture on public land would be important first steps in protecting biodiversity in this region. The use of plantations in the creative reforestation of regions to connect isolated patches will likely be a considerable improvement over the highly fragmented forested landscape that exists today. For small-scale sustainable forestry in the agricultural zone to be effective, however, changes to the way subsidies are paid to industrial forest users in the commercial forestry zone or a more equitable pricing policy for private landowners will be necessary.

Acknowledgments

We thank G. Weiss, G. Babish, and B. Yee of Environment Canada (Regina) for providing data, assistance, and direction with geographic information system (GIS) analysis. R. Bayne and G. Bayne provided additional assistance with GIS analysis. We thank M. Fitzsimmons for providing digitized National Topographic Series maps for our analyses. This paper was greatly improved by comments from B. McComb, S. Cushman, and one anonymous reviewer. Funding was provided by a Canadian Wildlife Service (CWS) operating grant to K.A.H. We thank B. Gummer of the CWS for supporting and encouraging our research in this area.

Literature Cited

Acton, D. F., G. A. Padbury, and C. T. Stushnoff. 1998. The ecoregions of Saskatchewan. Hignell Press, Winnipeg, Manitoba, Canada.

Alberta Environmental Protection. 1999. The final frontier: protecting landscape and biological diversity within Alberta's boreal forest natural region. Protected area report 13. Alberta Environment Protection, Natural Heritage Planning and Evaluation Branch, Edmonton, Canada.

Askins, R. A., J. F. Lynch, and R. Greenberg. 1990. Population declines in migratory birds in eastern North America. Current Ornithology **7**:1–57.

Baker, W. L. 1989. A review of models of landscape change. Landscape Ecology **2**:111–133.

Bayne, E. M. 2000. Effects of forest fragmentation on the demography of Ovenbirds (*Seiurus aurocapillus*) in the boreal forest. Ph.D. dissertation. University of Saskatchewan, Saskatoon, Canada.

Bohning-Gaese, K., M. L. Taper, and J. H. Brown. 1993. Are declines in North American insectivorous songbirds due to causes on the breeding range? Conservation Biology **7**:76–86.

Brovkin, V., A. Ganopolski, M. Claussen, C. Kubatzki, and V. Petoukhov. 1999. Modelling climate response to historical land cover change. Global Ecology and Biogeography **8**:509–517.

Cairns, M. A., and R. A. Meganck. 1994. Carbon sequestration, biological diversity, and sustainable development: integrated forest management. Environmental Management **18**:13–22.

Canadian Council of Forest Ministers. 2000. National forestry database program, compendium of forestry statistics. Canadian Council of Forest Ministers, Ottawa. Available from http://nfdp.ccfm.org/ (accessed November 2001).

Canadian Forest Service (CFS). 1998. Forest sector table foundation paper. Mimeograph, September 28. CFS, National Climate Change Process, Ottawa.

Cumming, E. E., K. A. Hobson, and S. L. Van Wilgenburg. 2001. Breeding bird declines in the boreal forest fringe of western Canada: insights from long-term BBS routes. Canadian Field-Naturalist **115**:425–435.

Cummings, S. G., P. J. Burton, S. Prahacs, and M. R. Garland. 1994. Potential conflicts between timber supply and habitat protection in the boreal mixedwood of Alberta, Canada: a simulation study. Forest Ecology and Management **68**:281–302.

Cushman, S. A., and D. O. Wallin. 2000. Rates and patterns of landscape change in the central Sikhote-alin Mountains, Russian Far East. Landscape Ecology **15**:643–659.

Coker, D. R., and D. E. Capen. 1995. Landscape-level habitat use by Brown-headed Cowbirds in Vermont. Journal of Wildlife Management **59**:631–637.

Dale, V. H., R. V. O'Neill, M. Pedlowski, and F. Southworth. 1993. Causes and effects of land-use change in central Rodonia, Brazil. Photogrammetric Engineering and Remote Sensing **59**:997–1005.

Davidson, A. T. 1952. Forest versus agricultural land along the pioneer fringe in Saskatchewan. M.A. thesis. University of Toronto, Toronto.

Dirzo, R., and M. C. Garcia. 1992. Rates of deforestation in Los Tuxtlas, a Neotropical area in Southeast Mexico. Conservation Biology **6**:84–90.

Donald, P. F., R. J. Fuller, A. D. Evans, and S. J. Gough. 1998. Effects of forest management and grazing on breeding bird communities in plantations of broad-leaved and coniferous trees of western England. Biological Conservation **85**:183–197.

Environment Canada (EC). 1991. The state of Canada's environment—1991. EC, Ottawa.

Food and Agriculture Organization (FAO). 1999. The state of the World's forests. FAO, Rome.

Fox, M. F., and S. L. Macenko. 1985. Agriculture-forest interface: an overview of landuse change. Working paper 38. Canada Department of the Environment, Lands Directorate, Ottawa.

Farm Woodlot Association of Saskatchewan (FWAS). 1991. A current evaluation of Saskatchewan's private forest resources. FWAS, Regina, Saskatchewan, Canada.

Fitzsimmons, M. 2002. Estimated rates of deforestation in two boreal landscapes in central Saskatchewan, Canada. Canadian Journal of Forest Research **32**:843–851.

Fox, J., J. Krummel, S. Yarnasarn, M. Ekasingh, and N. Podger. 1995. Land use and landscape dynamics in northern Thailand: assessing change in three upland watersheds. Ambio **24**:328–334.

Guy, R. D., and A. Benowicz. 1998. Can afforestation contribute to a reduction in Canada's net CO_2 emissions? Canadian Pulp and Paper Association, Montreal, Quebec.

Harris, L. D. 1984. The fragmented forest: island biogeography theory and the preservation of biotic diversity. University of Chicago Press, Chicago.

Hobson, K. A., and E. M. Bayne. 2000*a*. Effects of forest fragmentation by agriculture on avian communities in the southern boreal mixedwoods of western Canada. Wilson Bulletin **112:**373–387.

Hobson, K. A., and E. M. Bayne. 2000*b*. Breeding bird communities in boreal forests of western Canada: consequences of unmixing the mixedwoods. Condor **102:**759–769.

Hunter, M. L. 1990. Wildlife, forests, and forestry: principles of managing forests for biological diversity. Prentice-Hall, Englewood Cliffs, New Jersey.

Jenkins, M. A., and G. R. Parker. 2000. Changes in the forest landscape of the Charles C. Dean wilderness, Southern Indiana, 1939–1990. Natural Areas Journal **20:**46–55.

Kabzems, A., A. L. Kosowan, and W. C. Harris. 1986. Mixedwood section in an ecological perspective. Technical bulletin 8. Saskatchewan Parks and Renewable Resources, Regina, Canada.

Kress, M. R., M. R. Graves, and S. G. Bourne. 1996. Loss of bottomland hardwood forests and forested wetlands in the Cache River basin, Arkansas. Wetlands **16:**258–263.

Lee, R. G., R. O. Flamm, M. G. Turner, C. Bledsoe, P. Chandler, C. DeFerrari, R. Gottfried, R. J. Naiman, N. Schumaker, and D. Wear. 1992. Integrating sustainable development and environmental vitality. Pages 499–521 in R. J. Naiman, editor. New perspectives in watershed management. Springer-Verlag, New York.

MacTavish, P. 1995. Saskatchewan digital landcover mapping. Report I-4900-15-B-95. Saskatchewan Research Council, Saskatoon, Canada.

Mahli, Y., D. D. Baldocchi, and P. G. Jarvis. 1999. The carbon balance of tropical, temperate and boreal forests. Plant Cell and Environment **22:**715–740.

McClellan, J. B., L. Jersak, and C. L. A. Hutton. 1967. A guide to the classification of land use for the Canada Land Inventory. Geographical Branch, Department of Energy, Mines and Resources, Ottawa.

Myers, N. 1996. The world's forests: problems and potentials. Environmental Conservation **21:**156–168.

Pace, R. K., and R. Barry. 1997. Quick computation of regressions with a spatially autoregressive dependent variable. Geographical Analysis **29:**232–247.

Padbury, G. A., and D. F. Acton. 1994. Ecoregions of Saskatchewan. Minister of Supply and Services Canada and Saskatchewan Property Management Corporation, Regina, Canada.

Peterson, E. B., G. M. Bonnor, G. C. Robinson, and N. M. Peterson. 1999. Carbon sequestration aspects of an afforestation program in Canada's prairie provinces. Nawitka Renewable Resource Consultants, Victoria, British Columbia, Canada.

Price, J., S. Droege, and A. Price. 1995. The summer atlas of North American birds. Academic Press, London.

Robbins, C. S., D. Bystrak, and P. H. Geissler. 1986. The breeding bird survey: its first fifteen years. Research publication 157. U.S. Fish and Wildlife Service, Washington, D.C.

Saskatchewan Environment and Resource Management (SERM). 1995. State of the environment report - 1995. SERM, Regina, Canada. Available from http://www.serm.gov.sk.ca/publications.htm (accessed November 2001).

Saskatchewan Environment and Resource Management (SERM). 2000. Saskatchewan government initiatives on climate change. Regina, Canada. Available from http://www.serm.gov.sk.ca/environment/climatechange/cc-initiatives.htm (accessed November 2001).

Smith, A. R. 1993. Ecological profiles of birds in the boreal forest of western Canada. Pages 14–25 in D. H. Kuhnke, editor. Birds in the boreal forest. Northern Forestry Centre, Forestry Canada, Northwest Region, Edmonton, Alberta, Canada.

Spies, T. A., W. J. Ripple, and G. A. Bradshaw. 1994. Dynamics and pattern of a managed coniferous forest landscape in Oregon. Ecological Applications **4:**555–568.

Statistics Canada. 1993. 1991 census of agriculture: selected data for Saskatchewan rural municipalities. Statistics, Canada, Ottawa.

Stelfox, B. 1995. Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. Alberta Environmental Centre and Canadian Forest Service, Edmonton, Canada.

Stoorvogel, J. J., and L. O. Fresco. 1996. Quantification of land-use dynamics: an illustration from Costa Rica. Land Degradation and Development **7:**121–131.

Thorpe, J., and B. Godwin. 1999. Threats to biodiversity in Saskatchewan. Saskatchewan Environment and Resource Management, Regina, Canada. Available from http://www.serm.gov.sk.ca/ecosystem/biodiversity/index.htm (accessed 14 November 2001).

Turner, M. G. 1987. Landscape heterogeneity and disturbance. Springer-Verlag, New York.

Turner, M. G., D. N. Wear, and R. O. Flamm. 1996. Land ownership and land-cover change in the southern Appalachian highlands and the Olympic peninsula. Ecological Applications **6:**1150–1172.

van Kooten, G. C., E. Krcmar-Nozic, B. Stennes, and R. van Gorkum. 1999. Economics of fossil fuel substitution and wood product sinks when trees are planted to sequester carbon on agricultural lands in western Canada. Canadian Journal of Forest Research **29:**1669–1678.

Wallin, D. O., F. J. Swanson, and B. Marks. 1994. Landscape pattern response to changes in pattern generation rules: land-use legacies in forestry. Ecological Applications **4:**569–580.

Wear, D. N., and R. O. Flamm. 1993. Public and private disturbance regimes in the southern Appalachian watershed. Ecological Applications **6:**1173–1188.

Weir, J. M. H., and E. A. Johnson. 1998. Effects of escaped settlement fires and logging on forest composition in the mixedwood boreal forest. Canadian Journal of Forest Research **28**:459–467.

Weir, J. M. H., E. A. Johnson, and K. Miyanishi. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. Ecological Applications **10:**1162–1177.

Whitney, G. G., and W. J. Somerlot. 1985. A case study of woodland continuity and change in the American Midwest. Biological Conservation **31:**265–287.

Wickham, J. D., R. V. O'Neill, and K. B. Jones. 2000. Forest fragmentation as an economic indicator. Landscape Ecology **15:**171–179.

Zheng, D., D. O. Wallin, and Z. Hao. 1997. Rates and patterns of landscape change between 1972 and 1988 in the Changbai Mountain area of China and North Korea. Landscape Ecology **12:**241–254.

Zipperer, W. C. 1993. Deforestation patterns and their effects on patches. Landscape Ecology **8:**177–184.