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## Effect of agriculture and presence of American beaver *Castor canadensis* on winter biodiversity of mammals

Tim B. Nelner & Glynnis A. Hood

Several metrics of biodiversity are used to assess the variability and abundance of multiple species across various ecological levels. Previous research suggests that biodiversity in agricultural areas is diminished due to habitat fragmentation and alterations to natural vegetation. Contrary to the effects of agriculture, the American beaver *Castor canadensis* has been shown to have a significant contribution to increased biodiversity. Our study focuses on the biodiversity of mammals in wetland areas of the southern mixed-wood boreal forest natural region of east-central Alberta, Canada. We compared various measures of biodiversity levels of mammals in the winter months between wetlands on agricultural land and wetlands in Miquelon Lake Provincial Park (MLPP). Similarly, we compared wetlands with active and inactive beaver colonies to determine differences in winter biodiversity of mammals and the amount of water coverage within a pond. We collected data using winter tracking surveys and analyzed the data using geographic information systems (GIS) techniques. We found that winter biodiversity of mammals was higher at the sites within the protected area (MLPP) than at those on agricultural lands. However, our data also suggest that forested areas surrounding agricultural wetlands may play an important role in maintaining biodiversity within agricultural areas. The presence of beavers alone was not a significant factor in relation to winter biodiversity, but the presence of the species was important for maintaining water levels of agricultural wetlands, which in turn plays a role in resource heterogeneity.

*Key words:* agriculture, American beaver, biodiversity, *Castor canadensis*, mixed-wood boreal forest, winter tracking

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Agriculture has been identified as the greatest factor contributing to loss of biodiversity worldwide (McLaughlin & Mineau 1995). Impacts on biodiversity stems from the high degree of alteration of natural habitats within agricultural areas (Rosenblatt et al. 1999, Crooks 2002). These habitat changes can reduce the amount of habitat available for certain species and increase the distance between habitat patches (Mackenzie et al. 1998). In addition to habitat fragmentation, agricultural practices can increase nutrient and sediment levels in wetlands, which affect the ecological process therein (Detenbeck et al. 2002).

The complexity of the vegetation community

appears to be related to the abundance and diversity of small mammals (Maisonneuve & Rioux 2001, Moro & Gadal 2007). Vegetation provides ecological functions such as provision of food resources and protection from predation for these mammals (Moser & Witmer 2000, Moro & Gadal 2007). However, grazing by cattle has the potential to remove or alter this vegetative cover, thereby leading to lower biodiversity of small mammals (Moser & Witmer 2000). Also, increased nutrient and sediment run-off as a result of tillage can reduce the complexity of the vegetation community in adjacent agricultural areas (Houlahan et al. 2006). Lower plant diversity has been linked, in turn, to a decrease in diversity of small

mammals (Maisonneuve & Rioux 2001, Moro & Gadal 2007).

Large mammals (i.e. > 5 kg) occupy larger home ranges than small mammals (i.e. < 0.3 kg), which makes them less reliant on local habitat and more dependent on the condition of the overall landscape (Gehring & Swihart 2003). Therefore, the primary impact that agriculture has on these mammals is fragmentation of habitat (Mackenzie et al. 1998). Research on biodiversity in wetlands has often focused on invertebrate, bird, fish and vegetation communities (Mensing et al. 1998, Pollack et al. 1998). However, we had difficulty finding any empirical studies that examined the biodiversity of large mammals in wetlands, especially during the winter months. This lack of research surprised us because, while individual wetlands may not provide sufficient habitat for large mammals, they may help to serve as 'stepping stones' for wildlife movement (Hickey & Doran 2004), similar to those discussed in other studies relative to landscape connectivity in fragmented landscapes (e.g. Noss 1990). These 'stepping stones' are even more accessible during the winter months when ponds are frozen and facilitate travel.

American beaver *Castor canadensis* are abundant throughout boreal regions of North America (Wright et al. 2002), which includes our study area. Beaver significantly influence the ecology of a region by building and maintaining wetlands, which alters the water chemistry, availability and the structure of riparian vegetation (Naiman et al. 1994, Hood & Bayley 2008). The alterations are so significant that up to 25% of riparian vegetation may be linked to the presence of beaver (Wright et al. 2002). Furthermore, as beaver move through a landscape, they create a variety of habitat patches of varying sizes, ages and successional states (Naiman et al. 1994). This variety of habitat types across a region increases the biodiversity at the landscape level (Naiman et al. 1994).

Some studies suggest that agriculture reduces the biodiversity of a region (Maisonneuve & Rioux 2001, Gehring & Swihart 2003, Moro & Gadal 2007). To measure the biodiversity of wetlands in agricultural areas, most studies assess biodiversity of plants (Davis et al. 2007), macro-invertebrates (Thiere et al. 2009) or birds (Whited et al. 2000). Mensing et al. (1998) found that as cultivation of agricultural crops increased, shrub-carr vegetation communities and bird and fish abundance decreased. However, very few studies focus on the

presence of various taxa of large mammals within wetlands. We consider our project unique, because we attempted to assess the biodiversity of both large and small mammals in adjacent agricultural and protected landscapes.

The objective of our study was to determine which effects, if any, agriculture and the presence of beaver have on winter biodiversity of mammals in wetland areas. We hypothesized that: 1) winter biodiversity of mammals will be higher in the protected areas (MLPP) than in the agricultural lands, and 2) the same wetlands in which beaver are present, will have higher levels of winter biodiversity of mammals than wetlands without beavers.

## Material and methods

### Study area

Our research was conducted in Miquelon Lake Provincial Park (MLPP) and the surrounding area (Fig. 1). The study area is located at the southern end of the Cooking Lake Moraine and is part of the mixed-wood boreal forest region of east-central Alberta, Canada (Achuff 1994). This region is dominated by aspen *Populus* spp. forest and has an abundance of wetlands due to the knob and kettle terrain (Hood & Bayley 2008). Much of the area surrounding the park has been converted into agriculture, but forest cover is still present in some areas.

### Study design

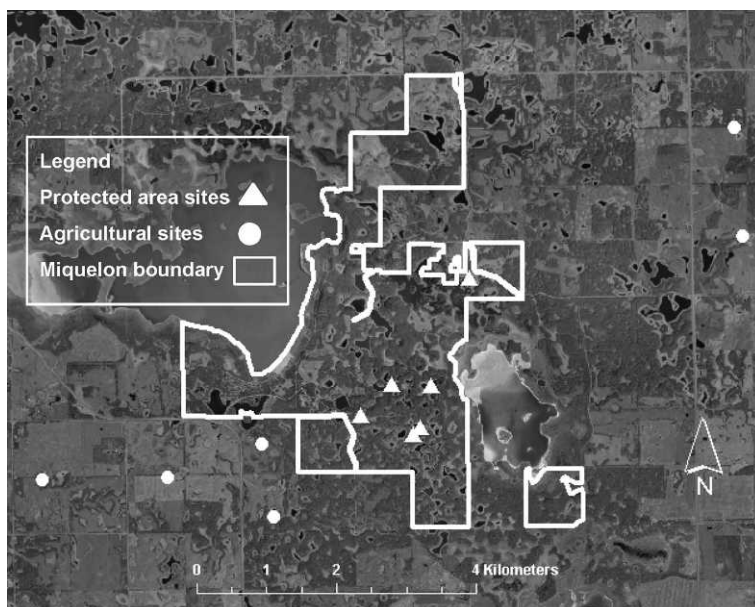
To examine the effects of agriculture and the presence of beaver on biodiversity, we used a stratified random sampling design to select three wetlands from each of the following four wetland categories: 1) agricultural with beaver present, 2) agricultural with no beavers present, 3) protected area with beaver present and 4) protected area with no beavers present.

### Data collection

#### *Beaver presence, land use and site selection*

During a previous study in January 2008 (Bromley & Hood, unpubl. data), all beaver lodges in MLPP were mapped and categorized relative to the presence of active beaver colonies. Lodges were classified as 'active' if a winter food cache was present, and 'inactive' if there was no food cache or other sign of activity (e.g. frost in the vent hole of the lodge or fresh cuttings). These activity data were confirmed during the course of our research.

Figure 1. Study sites in protected (within Miquelon Lake Provincial Park) and agricultural landscapes in the southern mixed-wood boreal forest in east-central Alberta, Canada.



We selected the potential wetlands for our study from a 2007 orthophoto (resolution < 1 m on the ground) and previously gathered data documenting beaver presence in wetlands in the MLPP (Bromley & Hood, unpubl. data). We limited potential study sites to a similar size of 4-10 ha. This process yielded an abundance of potential sites from which we identified a random set of study sites (see Fig. 1). The availability of study sites outside the MLPP was dependant on access from land owners, which limited the set of potential agricultural sites. We surveyed beaver lodges on private lands in January 2009 for activity (with the owner's permission), following the protocol of Hood et al. (2007) and Bromley & Hood (unpubl. data).

Agricultural sites (N = 6) tended to be concentrated to the southwest of the park, a result of final permissions for access to private lands. The park sites (N = 6) were also somewhat concentrated, but this trend was the result of a random selection. We acknowledge that 12 sites is a relatively small sample size. However, given that logistical considerations (e.g. access and seasonal trends) resulted in a short field season (i.e. four months) and adverse weather conditions limited the opportunity for field work, gathering data of this sample size was what was achievable to us.

#### ***Type of land use and track counts***

At each site we used a modified line-transect method to conduct winter tracking surveys. We conducted

surveys two days after a snowfall. At each pond, we surveyed four transects (at each of the four cardinal directions). These transects were 4-meters wide and extended 100 meters from the estimated center of each pond. The location of each mammal track encountered on the transect was recorded using a Garmin 60X global positioning system (GPS) unit, and the information was later entered into a GIS database. Other recorded variables included distance along the transect at which the track was found, habitat type (pond, riparian or upland), apparent habitat use of the animal and species of wildlife. We were unable to differentiate deer *Odocoileus* spp. tracks to species; therefore, we identified these tracks as 'deer'. Due to sexual dimorphism, we were also unable to differentiate the weasel *Mustela* spp. tracks to species, so these tracks were identified as 'weasel'. Occasionally snow conditions and age of the track also limited our ability to identify the track to species.

For data analysis, we categorized tracks into three groups: 1) total tracks identified for each species (identified tracks), 2) number of species and 3) total number of tracks (species identified and unidentified). We analyzed the tracks identified to species and the number of species observed to assess richness and abundance of particular species and the total number of tracks to assess gross habitat use by mammals.

Due to the time constraints and an unusually dry January that limited snowfall events, we were only able to visit each site once in that month. Conse-

quently, the data that we collected only represent relative biodiversity on a particular day. However, we surveyed eight sites on the same day and the remaining four sites were surveyed within the next 11 days. This survey approach ensured that observations occurred during similar weather conditions and time of year, which mitigated the potential error from a small sample size.

### **Beaver status and track counts**

Because we used the presence or absence of beaver as an explanatory variable in our experimental design, beavers were not included as a species in the assessment of species richness and abundance. We categorized track data relative to pond type (i.e. active or inactive and protected or agricultural).

### **Wetland characteristics, land use and beaver status**

All wetlands were open water ponds, often with a marsh edge. Although active beaver colonies were not present at all sites, given the presence of old beaver cuttings and the natural history of the area, we assumed that all ponds had some level of beaver activity in the past. Wetlands were classified in the field according to visible ice cover, which we then used to estimate the area of open water prior to freeze-up. This classification system is based on a four class scale (class 1, 2, 3 and 4). A Class 1 pond is one that is completely full of water to the vegetated edge, whereas a Class 4 pond is completely dry (although as a true wetland, it could refill with an increase in precipitation). A Class 2 pond has some water in the pond body, but has exposed soil on the shore due to some loss of water. A Class 3 pond has

water only in the channels (generally only ponds with a current or recent (< 5 years) beaver activity had numerous channels along the pond bottom; see Hood & Bayley 2008).

We also recorded the distance along the transects where habitat changed from pond to riparian to upland. We defined these habitats by vegetation type (i.e. 'pond' = ice-covered pond surface, 'riparian' = emergent and wetland facultative species and 'upland' = non-wetland facultative shrub and trees). Along with separating habitat types, we used these data to determine the width (and approximate area) of each 'wetland zone'. If the transect ran immediately into agricultural croplands or pasture, we defined the values for the three zones to be zero.

We determined pond size (Table 1) and riparian area using a geographic information system (ArcMap 9.2; ESRI 2009). We digitized the study sites in the GIS, and we conducted a field calculation to determine the pond area. We calculated the riparian areas in the same way. We calculated these pond attributes using a 2007 orthophoto taken prior to the late summer drought conditions of 2008.

Finally, we determined how these pond-specific metrics (e.g. riparian width and pond type) related to the presence or absence of beavers. For these analyses, beaver status and land use were independent variables, and pond size, pond type and riparian width were dependent variables.

### **Data analysis**

After testing the data for normality using a Shapiro-Wilk's *W* test and homogeneity of variance (Le-

Table 1. Site information relative to land type, size, transect type, pond-class (1 = water to edges, 2 = exposed shorelines, 3 = water only in channels and 4 = dry) and the presence (+) or absence (÷) of beavers.

Pond ID	Land type	Number of		Size (m <sup>2</sup> )	Pond class	Beaver presence
		Forested transects	Agricultural transects			
1	Agricultural	2	2	8487	3	+
2	Agricultural	1	3	2367	4	÷
7	Agricultural	4	0	1426	4	÷
18	Agricultural	3	1	10625	2	+
20	Agricultural	3	1	2880	3	+
22	Agricultural	1	3	5740	3	÷
23	Forest	4	0	6485	2	+
32	Forest	4	0	13022	2	+
37	Forest	4	0	7415	2	÷
42	Forest	4	0	1387	2	÷
43	Forest	4	0	2577	3	+
65	Forest	4	0	11311	3	÷

vene's test), we used a nested analyses of variance (ANOVA) to assess differences in biodiversity while controlling for any within-pond site variability. We found no significant differences between ponds in a specific landscape type ('pond' was nested within 'landscape type') for any of the measured variables. Therefore, we felt comfortable using a number of one-way ANOVA tests to further analyze our data. The measures of biodiversity analyzed were 1) number of identified tracks (tracks that could be attributed to a species), 2) total number of tracks (includes tracks that could not be attributed to a species), 3) number of species and 4) Simpson's Index of Diversity (D). These four biodiversity measures allowed us to examine any combination of the measures to show either strict relative abundance of mammals or relative abundance and diversity. To accommodate for differences in vegetation cover relative to transect location (particularly at agricultural sites with some forested transects), we analyzed transect specific data as opposed to grouping the total data collected from a particular pond. As

such, for agricultural sites we further categorized the data as 1) 'agricultural-agricultural' (i.e. an agricultural transect at an agricultural site) and 2) 'forested-agricultural' (i.e. a forested transect at an agricultural site). All transects within MLPP were forested due to the protected status of the landscape, so they were all 'forested-protected' (see Table 1). All values were considered significant at  $\alpha=0.05$ .

When the data did not meet normality or homogeneity of variance assumptions (e.g. species-specific habitat preferences), we used the Kruskal-Wallis ANOVA. We analyzed all data using STATISTICA version 8 (StatSoft Inc. 2008) and SIGMAPLOT version 11 (Systat Software Inc. 2009).

## Results

### Type of land use and track counts

The total number of tracks identified to species per transect was 1.9 times higher in ponds in the

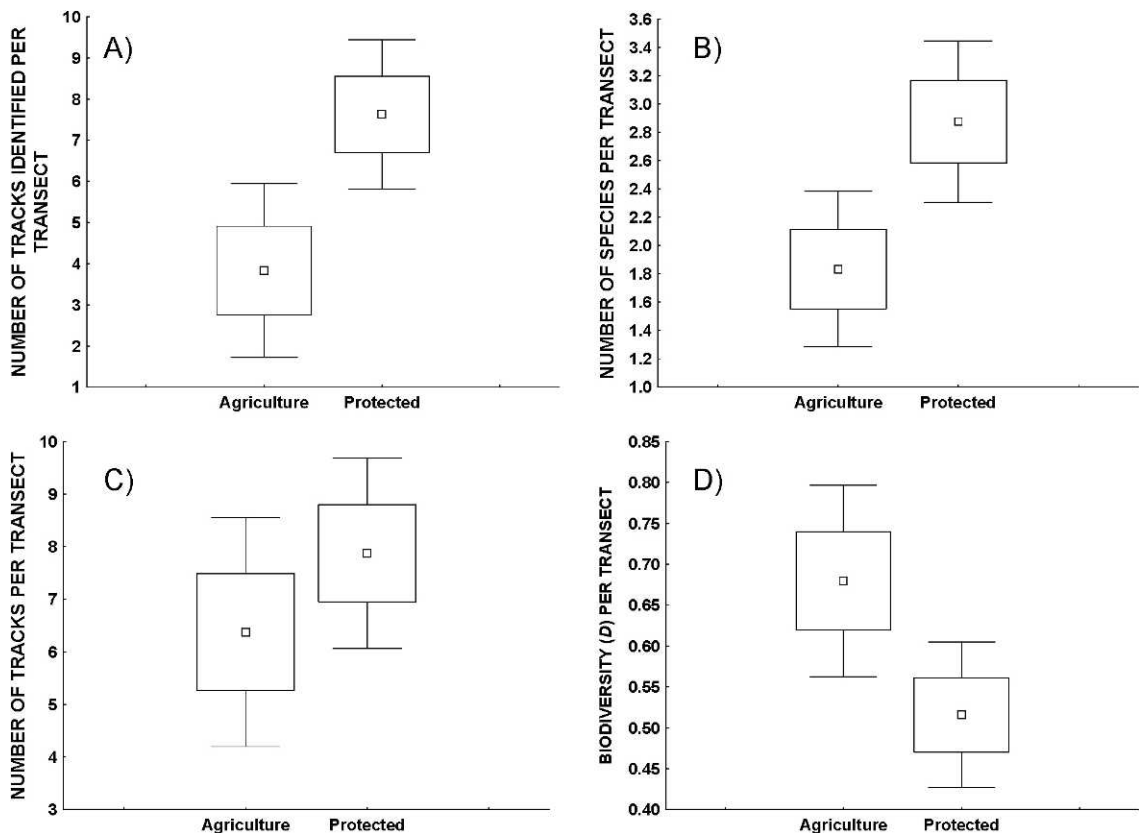


Figure 2. Number of tracks identified to species per transect (A), number of species per transect (B), number of identified and unidentified tracks per transect (C) and biodiversity, as measured by the Simpson's Diversity Index (D) per transect at agricultural and protected sites. Values given are means  $\pm$  standard error.

protected area ( $\bar{x} = 7.6$  tracks/transect) than in ponds in agricultural lands ( $\bar{x} = 3.8$  tracks/transect,  $F_{1,46} = 7.143$ ,  $P = 0.01$ ; Fig. 2A). In addition to the number of identified tracks, the number of species per transect was higher at ponds in protected areas ( $\bar{x} = 2.8$  species/transect) than at agricultural sites ( $\bar{x} = 1.8$  species/transect,  $F_{1,46} = 6.739$ ,  $P = 0.013$ ; see Fig. 2B). The total number of tracks per transect showed a trend of more tracks in protected area ponds ( $\bar{x} = 7.9$  tracks/transect) than in agricultural ponds ( $\bar{x} = 6.4$  tracks/transect,  $F_{1,46} = 1.08$ ,  $P = 0.3$ ; see Fig. 2C), but this trend was not statistically significant. Biodiversity (as calculated using the Simpson's Index) was lower in agricultural ponds ( $D = 0.68$ /transect) than in protected area ponds ( $D = 0.52$ /transect,  $F_{1,46} = 4.272$ ,  $P = 0.044$ ; see Fig. 2D). Note that a higher value for  $D$  indicates lower biodiversity. We identified 10 species or species groups (e.g. deer) along the transects. They included: moose *Alces alces*, deer, coyote *Canis latrans*, weasel, pine marten *Martes americana*, snowshoe hare *Lepus americanus*, red squirrel *Tamiasciurus hudsonicus*, red-back vole *Myodes gapperi*, shrew *Sorex* spp. and 'subnivian' (as determined by vent or access holes in snow).

We found that for transect-specific data, forested transects (in both agricultural and protected areas) had 2.8 times the number of identified tracks ( $\bar{x} = 6.6$  tracks/transect) than agricultural-agricultural transects ( $\bar{x} = 2.3$  tracks/transect,  $F_{1,46} = 6.02$ ,  $P = 0.018$ ; Fig. 3A). The number of species was also higher for forested transects, regardless of land type ( $\bar{x} = 2.5$  species/transect), than on agricultural-agricultural transects ( $\bar{x} = 1.4$  species/transect,  $F_{1,46} = 5.78$ ,  $P = 0.02$ ; see Fig. 3B). The total number of tracks was 2.2 times higher on forest transects ( $\bar{x} = 8.3$  tracks/transect) than on agricultural transects ( $\bar{x} = 3.7$  tracks/transect; see Fig. 3C), but this trend was not statistically significant ( $F_{1,46} = 6.624$ ,  $P = 0.13$ ). Biodiversity ( $D$ ) did not vary between transect types ( $\bar{x}_{\text{forest}} = 0.4$ ,  $\bar{x}_{\text{agricultural}} = 0.305$ ,  $F_{1,46} = 1.552$ ,  $P = 0.22$ ).

Note that we conducted this analysis using both ponds in the protected area and ponds in agricultural lands. When we limited our analysis to forested-agricultural and agricultural-agricultural transects only, the number of tracks identified to species was 2.1 times higher for forested transects than agricultural transects, but this difference was not statistically significant ( $\bar{x}_{\text{forested-all}} = 4.9$  tracks/transect,  $\bar{x}_{\text{agricultural-agricultural}} = 2.3$  tracks/transect,  $F_{1,22} = 1.48$ ,  $P = 0.24$ ). However, the overall number

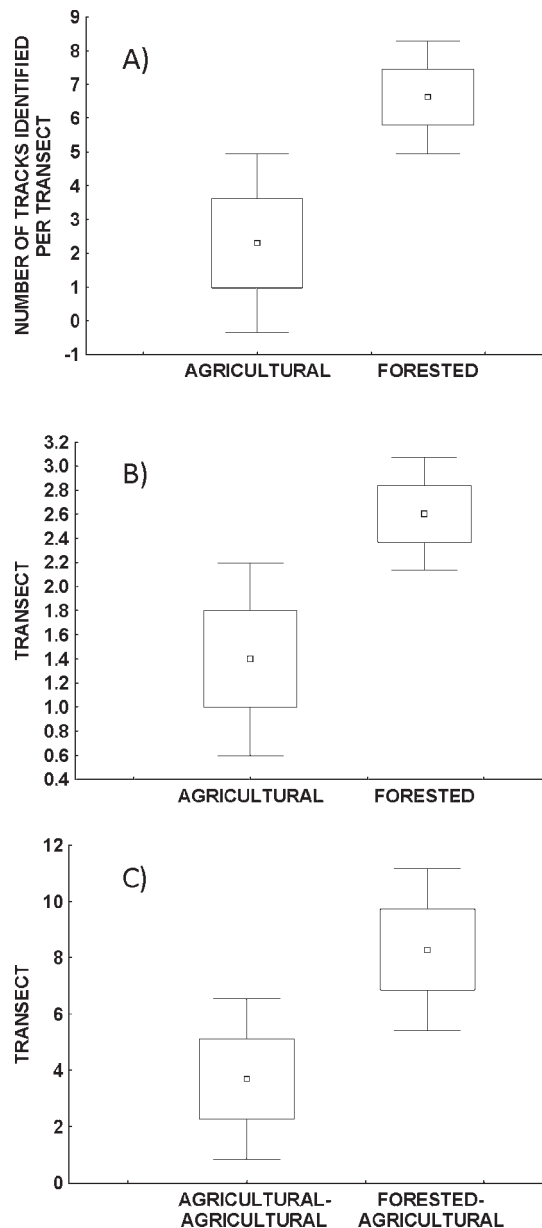


Figure 3. Number of tracks identified to species on forested and agricultural transects regardless of land type (A); number of species on forested and agricultural transects, regardless of land type (B); and number of tracks on transects leading directly onto croplands or grazing lands (agricultural-agricultural) and number of tracks on transects leading onto forested dominated transects on agricultural lands (forested-agricultural; C). Values given are means  $\pm$  standard error.

of tracks (including tracks unidentified to species) on 'forested-agricultural' transects was significantly higher.

There were 2.2 times as many of these tracks on the forest transects than on the agricultural

transects ( $\bar{x}_{\text{forested-agricultural}} = 8.3$  tracks/transect,  $\bar{x}_{\text{agricultural-agricultural}} = 3.7$  tracks/transect,  $F_{1,22} = 4.845$ ,  $P = 0.039$ ; see Fig. 3C).

### Beaver status and track counts

Within ponds in protected areas, there was no significant trend for winter biodiversity between ponds with active and inactive beaver colonies. This trend (or lack thereof) was consistent for all measures of wildlife presence and biodiversity in our study.

Within agricultural ponds there were 2.5 times as many identified tracks in ponds with active beaver colonies ( $\bar{x} = 5.5$  tracks/transect) than in ponds without active colonies ( $\bar{x} = 2.1667$  tracks/transect,  $F_{1,22} = 2.561$ ,  $P = 0.12$ ). While this was a very strong trend, it was not statistically significant.

### Wetland characteristics, land use and beaver status

We found that ponds in the protected area had a higher pond class ( $\bar{x} = 2.3$  pond class) than ponds in agricultural areas ( $\bar{x} = 3.2$  pond class,  $F_{1,10} = 5.0000$ ,  $P = 0.049$ ). As previously indicated, lower values for pond classes (e.g. classes 1 and 2) indicate ponds with more water than ponds with higher values (e.g. classes 3 and 4). There was no difference in pond size between protected areas ( $\bar{x} = 7,032.8$  m<sup>2</sup>) and agricultural areas ( $\bar{x} = 5,254.2$  m<sup>2</sup>,  $F_{1,22} = 0.544$ ,  $P = 0.48$ ). Similarly, the area of riparian vegetation surrounding the pond sites tended to be larger in protected area ponds ( $\bar{x} = 13,721.5$  m<sup>2</sup>) than in

agricultural ponds ( $\bar{x} = 9,380.333$  m<sup>2</sup>), but this trend was also not significant ( $F_{1,10} = 1.444$ ,  $P = 0.26$ ).

Habitat type (pond, riparian and upland) was a strong determinant of species presence (Fig. 4). The species most associated with upland habitat type were deer ( $\chi^2 = 17.841$ ,  $df = 2$ ,  $P < 0.0001$ ), weasel ( $\chi^2 = 10.369$ ,  $df = 2$ ,  $P = 0.0056$ ) and snowshoe hare ( $\chi^2 = 28.58$ ,  $df = 2$ ,  $P = 0.0001$ ). Subnivian species showed a significant preference for sedge or *Typha latifolia* dominated riparian habitats ( $\chi^2 = 15.631$ ,  $df = 2$ ,  $P = 0.0004$ ).

When we grouped agricultural and protected area sites, differences in pond class for ponds with beaver ( $\bar{x} = 2.5$  pond class) compared to those without ( $\bar{x} = 3.0$  pond class) were not significant ( $F_{1,10} = 1.364$ ,  $P = 0.27$ ). However, when we only examined agriculture sites, the presence of beaver was a significant factor relative to pond class. Agricultural ponds with active beaver colonies had a higher pond class ( $\bar{x} = 2.7$  pond class) than those without active beaver colonies ( $\bar{x} = 3.7$ ,  $F_{1,22} = 24.75$ ,  $P < 0.001$ ). We did not find a similar trend relative to beaver status within protected sites. Average pond class was equal between ponds with beaver and those without ( $\bar{x} = 2.3$ ,  $F_{1,22} = 0.000$ ,  $P = 1.0$ ).

## Discussion

In many cases, we found that land use and the presence of beaver were good determinants of winter

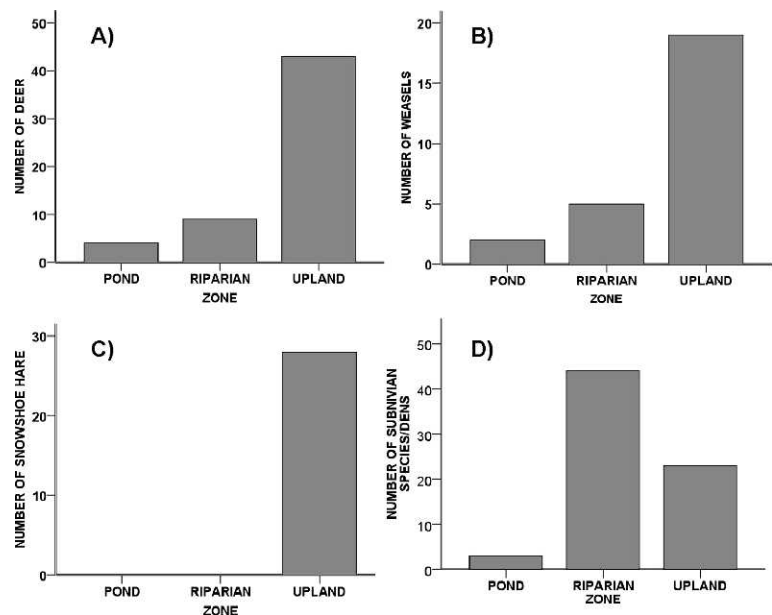


Figure 4. Use of pond (ice and snow on pond surface), riparian (wetland obligate and facultative vegetation species) and upland habitats (non-facultative vegetation species) habitats by deer (A), weasels (B), snowshoe hare (C) and subnivian species (D). Subnivian species were identified by entrance and exit holes in the snow.



wildlife use of wetlands. Protected areas provide important habitats for many species; however, we were also determined that not all agricultural areas are void of high levels of biodiversity. Despite these findings, our small sample size ( $N=12$ ) suggests that only the most robust effects might be detected. A larger sample size for similar studies would aid in a more in-depth analysis of the relationship between land use and biodiversity.

### **Type of land use and track counts**

Our research determined that the total number of mammal species was almost twice as high in wetland and adjacent upland habitats in protected areas than in areas dominated by agriculture. In turn, biodiversity values (as determined by D) and the number of species of mammals were also higher in protected area wetlands than in agricultural sites. These results are consistent with other studies (McLaughlin & Mineau 1995, Moser & Witmer 2000) that also noted reduced biodiversity in agricultural areas.

Many studies discuss the importance of forests for connectivity (Noss 1990, Rosenblatt et al. 1999) as wildlife corridors and useful habitat. We found that all measures of track abundance (i.e. number of tracks identified to species/transect, number of species/transect and total number of tracks/transect) were between 2 and 3 times higher on forested transects regardless of land type (agricultural or protected). In addition, an important finding in our research was that not all agricultural transects offered equivalent habitat quality, despite being within a similar matrix of large tracts of agricultural land. We found a two-fold increase in the total number of tracks on forested transects on agricultural lands compared to non-forested agricultural transects. Adjacent upland habitats were also important for many species using the agricultural transects.

Within our study area, all wetlands surveyed had at least one transect leading into forest cover and only 10 of 24 transects (42%) on agricultural wetlands crossed directly from wetland into crop or pasture areas. Some wetlands had only a small buffer strip of emergent vegetation or forested habitat (Fig. 5A), whereas others had extensive forested areas beyond this buffer zone (see Fig. 5B). Therefore, all agricultural wetlands cannot be considered as identical, as is sometimes the case with current land management. Our findings suggest that retention of some forested areas adjacent to agri-

cultural wetlands provides important habitat for mammals during winter months. It is possible that these forest transects in agricultural ponds are creating a refuge effect, where animals choose these areas and avoid areas with a more distinct agricultural interface.

### **Beaver status and track counts**

Previous studies suggest that beavers play a significant role in creating and maintaining biodiversity (Naiman et al. 1994, Wright et al. 2002, Hood & Bayley 2008). Our study suggests that within protected areas, beavers have little or no role in the maintenance of biodiversity and wildlife use during the winter months. It is the wetlands and adjacent habitats, regardless of beaver presence, that appear to be used as travel corridors (Hickey & Doran 2004) and winter refuges. In addition, the adjacent forage resources (Hood & Bayley 2009) and thermal cover are critical for over-winter survival of wildlife that remain active during the winter months.

Within agricultural lands, active beaver sites had a greater number of identified tracks, but this was not statistically significant and other measures of biodiversity did not show similar trends. The discrepancy between our results and those of previous research may be a result of our study being conducted during the winter when many species of wildlife are not active, and our sample sizes being smaller.

### **Wetland characteristics, land use and beaver status**

Ponds had larger areas within the protected landscape than within the agricultural landscape, although this trend could be a result of potential bias due to restricted access to ponds outside the park, or an effect of agricultural land use. In addition to increased pond areas, riparian and upland vegetation was more extensive at ponds within the protected area. Increased vegetation results in increased habitat for wildlife. These habitats are promoted through direct protection of natural landscapes and are likely important for maintaining landscape connectivity (Noss 1990).

Within the agricultural habitats deer, weasel and snowshoe hare showed a strong preference for upland habitat. The abundance of these species in these habitats suggests that key ecological functions are being provided at multiple scales. Whited et al. (2000) determined that landscape connectivity and road density were important predictors for bird

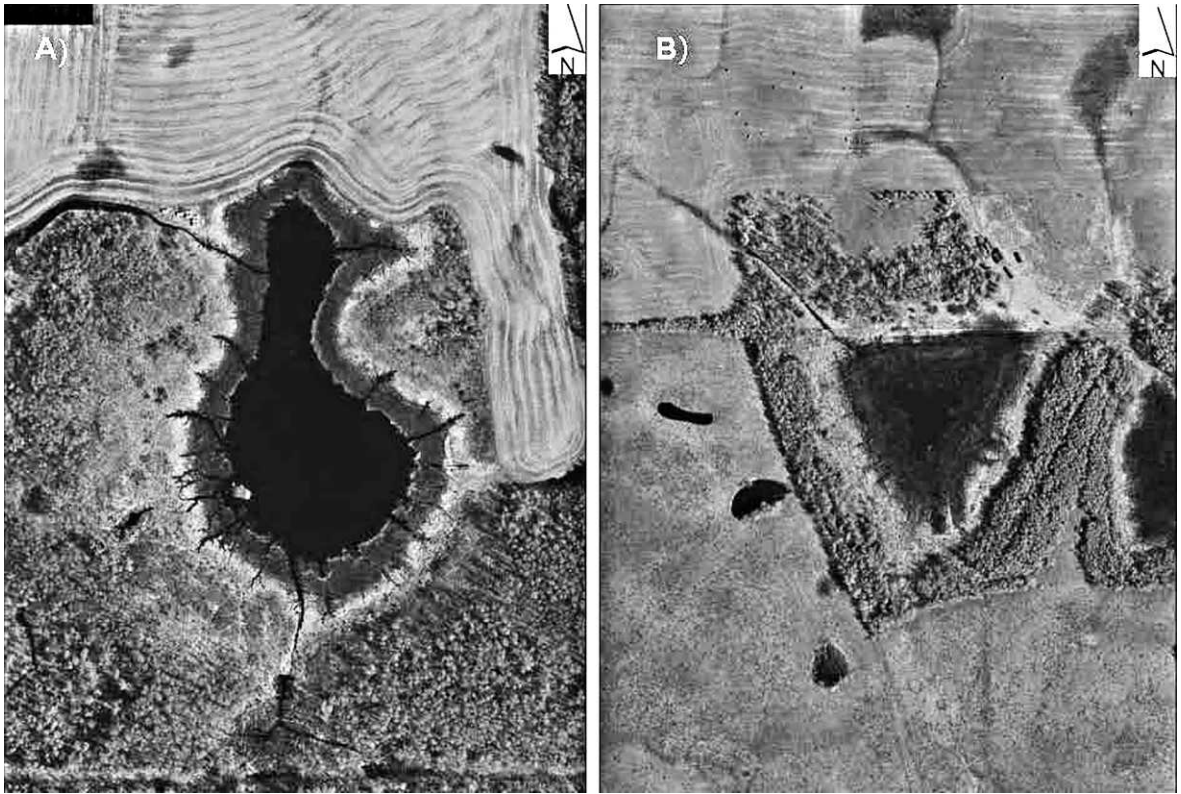


Figure 5. Agricultural study site surrounded by a considerable forest cover (A) and an agricultural study site with a small buffer of forest area (B).

community composition, with connectivity being more pronounced at larger scales, and road effects at smaller scales. As with birds, mammals have varied habitat requirements that function at many spatial and temporal scales. Connectivity between important habitats (e.g. upland habitats in the case of deer, weasel and hare) often provides safer access for dispersal and seasonal migration (Noss 1990).

The significant trend in lower over-winter water levels in agricultural ponds is one that requires further study. We found that protected areas had more ponds within the Class 1 (full to the edge with water) and Class 2 (full with muddy shorelines that indicate drawdown) categories than ponds on adjacent agricultural areas had. On agricultural lands, we found that the presence of beaver was a significant determinant of higher water levels. This finding is likely due to the ability of beaver to build and maintain wetlands (Naiman et al. 1994, Hood & Bayley 2008). The fact that beaver influenced water levels in agricultural lands, but not in the protected area is interesting. This pattern suggests that beaver may have a greater importance in maintaining water levels in agricultural areas than in pristine habitats;

however, a larger sample size is required to fully assess this idea. Also, within MLPP, although a pond might be classified as 'inactive', almost every pond in the park has been inhabited or modified by beaver at some point (as seen from cuttings and old lodges), whereas the same was not always apparent for agricultural ponds. It is important to note that our study was conducted during a drought that began in the summer of 2008 and extended throughout 2009. In a previous study, Hood & Bayley (2008) found that beavers play a critical role in mitigating drought in this region of the mixed-wood boreal forest. The effect of beaver on water availability may differ in the protected and the agricultural landscapes.

Due to weather conditions and other logistical constraints, our study had a relatively small sample of wetlands ( $N=12$ ). Also, field surveys were limited to periods immediately following snowfall events and so only a snapshot of winter use by mammals was possible. To validate the results of this study, a similar study is required in which more sites are selected and the sites are visited more frequently over the winter.

Additional research is also needed to address the effect of beaver on agricultural lands. The overall trend of increased water coverage on agriculture lands, when beaver are present, could be explored more extensively. Beaver create and maintain wetlands (Naiman et al. 1994, Hood & Bayley 2008), but the mechanism that makes this trend more significant in agriculture lands than in the protected area is unclear, especially given that the topography throughout our study area was comparable.

Finally, the ability of forested areas within agricultural lands to maintain biodiversity is important. Future research could focus specifically on this factor within agricultural wetlands to consider the functional differences between forest transects that traverse into a short buffer zone and transects that extend into a significant area of forest. Additionally, comparison of forest buffer versus the effect of beavers on biodiversity could help determine the relative significance of each factor. The importance of these habitats within agricultural lands cannot be overstated and additional research would be enlightening.

## Conclusions

In common rhetoric and even in the literature, there often is a perception that agricultural wetlands have little to no naturally vegetated buffer area and thus, lower habitat value. Agricultural wetlands are also often depicted as being disconnected from other wetlands by an agricultural matrix instead of natural habitat. Such assumptions need to be challenged.

Many agricultural wetlands within the southern Cooking Lake Moraine still maintain some forested buffer and the possibility for conservation management is present. Our research suggests that these forested areas (upland habitat) provide important winter resources for many mammal species (e.g. deer, weasel and snowshoe hare). If riparian and upland areas can be maintained within agricultural areas now and in the future, they may provide important habitat for many species and aid in the conservation of biodiversity. Maintaining these upland areas may also help to ensure connectivity between wetlands. Such conservation measures would maintain the connectivity of the landscape for large mammals while also conserving important riparian habitat for smaller mammals.

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