VOLUME 10 • NO. 3 • WINTER 2002

Research Links

A Forum for Natural, Cultural and Social Studies

Resolving the Triassic/Jurassic Extinction Event
A Case Study in Fossil Resource Management, Queen Charlotte Islands, BC

James W. Haggart

Approximately 200 million years ago, at the end of the Triassic Period, the Earth's biota experienced one of the most significant events in its history: at that time, as much as 80% of all living plant and animal life went extinct (Sepkoski 1993; Hallam & Wignall 1997). The end-Triassic extinction event was widespread, affected both animals and plants, and occurred both in the ocean and on land. The life forms that evolved subsequently, in the Early Jurassic, expanded into a world that was radically changed from the preceding one. Indeed, so great was the end-Triassic (hereafter referred to as "T/J") extinction that it forms the basis for the division between the Triassic and Jurassic geological time periods.

Although the magnitude of the extinction is not questioned, geologists and Earth historians have seriously debated both the length of time over which it occurred, and the nature of global environmental changes that accompanied it. Some workers have suggested that the extinction was rapid, taking place over tens of thousands of years, while others have maintained that it was a very long-lived event, spread out over several million years. Similarly, differing explanations for the underlying causative mechanism of the extinction have been proposed, including changes in volcanic activity on the Earth, changes in oceanic sea levels, and extraterrestrial impacts.

THE EXTINCTION RECORD ON THE QUEEN CHARLOTTE ISLANDS

Sedimentary rocks exposed on the Queen Charlotte Islands, BC, contain a well-preserved record of the ocean life during the Late Triassic. Two critical areas of exposure, one on the northwest coast of the islands at Kennecott Point, and the other 175 kilometers to the south at Kunga Island, in Gwaii Haanas National Park Reserve (Figure 1), have provided significant insights into changes occurring during the T/J extinction. In fact, the abundance and quality of paleontological data at these sites has led to their proposal as the global boundary stratotype (GSSP) section for the Triassic/Jurassic boundary (Carter & Tipper 1999; Haggart *et al.* submitted).

Paleontological analysis of the rocks from these two stratigraphic sections reveals a Late Triassic ocean teeming with a great variety of large life forms, such as bivalves, ammonites, and reptiles, as well as microscopic life forms such as the conodonts and radiolarians. Conodonts were primitive chordates that suffered final extinction in the end-Triassic event, while radiolarians are unicellular life forms with complex skeletal structures that are still found in today's oceans.

Detailed study of stratigraphic sections spanning the Triassic/Jurassic boundary at
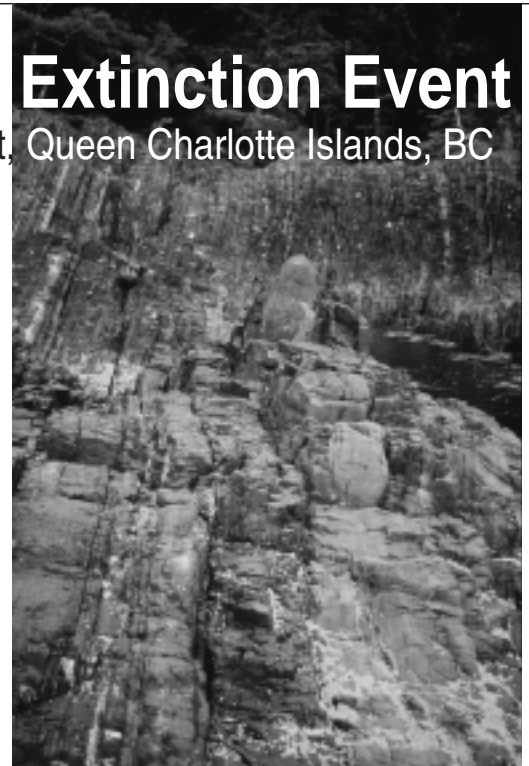
Kunga Island, Gwaii Haanas National Park Reserve, BC

both Kennecott Point and Kunga Island has shown a great diversity of ocean life forms. Through careful paleontological sampling, the precise stratigraphic ranges of various fossil species in the rock sections have been recorded (Figure 2, on p. 5). The geological and paleontological data show dramatically that the T/J extinction event recorded in these rocks is restricted to a very thin section of rock, less than one meter thick (Figure 2: horizontal line (Carter 1993; Tipper *et al.* 1994; Tipper *et al.* 1998; Haggart *et al.* 2001).

Once the precise ranges of fossil taxa were established at Kennecott Point and Kunga

Research Links

Contents

FEATURE ARTICLES

RESEARCH HIGHLIGHTS

DEPARTMENTS

FRANCOPHONES

Le texte de cette publication est offert en français. Vous pouvez l'obtenir en écrivant à l'adresse à la page 24.

RESEARCH LINKS ONLINE

Previous issues of *Research Links* are available online at: **http://parkscanada.pch.gc.ca** under "Library", in the "Download Documents" section.

Editorial

It may be that my perspective is influenced by six-months leave from *Research Links*, but the articles in this issue seem to cover a particularly wide range of research projects and challenges. From archaeological/paleological research, to population ecology, to behavioural studies, each of the authors addresses important questions regarding resource conservation. They also draw our attention to several problems they are facing in a broader context.

The research in this issue spans a timeline that exceeds 200 million years. Haggart's work focuses on biological clues to explain a massive extinction event at the end of the Triassic period. Although his work deals with events of the past, it points to the importance of standards that preserve data and finite resources (i.e. fossils) for future analysis. At the other end of the timeline, we have forecasts for caribou and whitebark pine populations (Flanagan & Rasheed, Stuart-Smith *et al.*, respectively) that show the value of careful predictions and adaptive management.

Behavioural studies in this issue focus on several fauna, ranging from insectivores to carnivores, and deal with cumulative effects. In one article, we see that a previous management decision to merge small water bodies may be having unforeseen effects on waterfowl foraging behaviour (McParland & Paszkowski). In another study, Whittington examines the combined effects of human development on wolf movements in Jasper.

The long-term and interdisciplinary benefits of today's research are often not immediately apparent. However, articles like these suggest that researchers are thinking beyond the scope of their own research questions in an effort to improve the usefulness of their findings to others. Effective communication among and between researchers and managers should not be underrated, and is one of the main objectives behind *Research Links*. We hope this issue and this publication continue to address that need.

Dianne Dickinson is Production Editor of Research Links.

OUR APOLOGY!

We offer a sincere apology to David Gummer, whose image of a Northern Black-tailed Prairie Dog was somewhat distorted in the Summer/ Autumn 2002 issue (*Research Links* 10[2] p. 4).

The image should have appeared like this:

NEW ADDITIONS...

The population boom in the *Research Links* family continues as Dianne Dickinson returns from maternity leave and Sharon Thomson begins her maternity leave. Both Rebecca Dickinson and Grace Thomson are doing very well.

In Sharon's place, we welcome Katharine Kinnear, who works with Cultural Resource Services in the Calgary office of the WCSC, as our cultural representative.

Resolving the Triassic/Jurassic Extinction Event

Island, and the T/J boundary precisely identified at both localities, the geology of the boundary interval was further investigated to look for changes in Earth's global atmospheric and biological systems. Sampling and analysis of stable isotopes of carbon in these rocks show a significant change in global productivity during the Late Triassic. The isotopic record across the boundary interval shows that the end-Triassic extinction was accompanied by a dramatic negative shift in the percentage of organic carbon in the rocks (Ward *et al.* 2001). Such a decline suggests a pronounced global productivity collapse right at the level of the extinction event, an event that likely occurred over 50 000 years or less. Causative mechanisms for such rapid productivity declines are uncertain, but such a scenario is consistent with an impact from an extraterrestrial object. Although further work is required to conclusively demonstrate that an impact event caused the end-Triassic biological catastrophe, this is the first strong evidence obtained for a rapid end-Triassic extinction.

RESOURCE RECOGNITION

Unraveling this complex biological history has required a great deal of paleontological patience and geological sleuthing, spread over many years. To make the necessary fossil collections, many kilograms of rock have been extracted and processed to isolate the microscopic fossils contained within them. Unfortunately, the fossils are limited in occurrence; they are found only in specific rock lithologies conducive for their long-term preservation — these were the rock types sought in each of the critical sections. Fossil materials from each sample were carefully catalogued, so that the precise level of each fossil collection (and there are hundreds) was identified relative to all others, and relative to the succession of strata in the overall rock sequence. Indeed, the sampling required to elucidate the relative succession of fossil species in both the Kennecott Point and Kunga Island sections has been so extensive that

very few examples of appropriate lithology remain at either locality.

By their nature, fossils are a finite resource — they are the sole remains of unique biological species that existed at one time in the ancient past, most of which went extinct and are gone forever. Once fossils are removed from the ground, they can never be regenerated. The only way further examples may be found is to excavate the rock immediately adjacent to the original locality. In the case of both Kennecott Point and Kunga Island, the fossil-containing beds have been thoroughly excavated. Only through further erosion or large-scale excavation will additional fossils be brought to the surface; such processes over natural time scales can take many millennia to accomplish.

THE RESOURCE MANAGEMENT DILEMMA

As in other scientific disciplines, geologists and paleontologists attempt to replicate results of others' work to ensure accuracy of results. However, further sampling of the Kennecott Point and Kunga Island sections is clearly problematic since very little, if any, of the appropriate, microfossil-containing lithologies remain in the outcrop. This dilemma raises a clear management issue: how best to maintain the paleontological record of these sites? Since very few examples of the fossils themselves can or will ever be re-collected from these localities, an effective management plan for the paleontological resources must address the long-term maintenance

Figure 1. Location map of Queen Charlotte Islands, British Columbia, showing Kennecott Point and Kunga Island, sites of Triassic/Jurassic boundary interval sections.

and storage of the existing collections as well as the associated database information.

Fortunately, informal standards exist for the collection and documentation of fossil collections obtained through most scientific programs. Fossil locality data are typically encoded in a computer database for permanent storage. Ideally, the fossil collections themselves are placed in an institution with a mandate for long-term storage of such materials, such as a museum, a university, or a government laboratory. In this way, scientists can gain access to specific paleontological materials that have been studied previously. Unfortunately, legislation governing such long-term storage is quite limited, and numerous instances where uninformed bureaucrats have disposed of fossil collections have been documented.

There are approximately 6,000 known fossil localities on Queen Charlotte Islands, though only a few share the same level of importance as the T/J boundary sections at Kunga Island and Kennecott Point. This may be because the fossils themselves and/ or the stratigraphic level represented by these localities are more widely distributed. As well, the precision of paleontological studies may not necessitate, or even allow
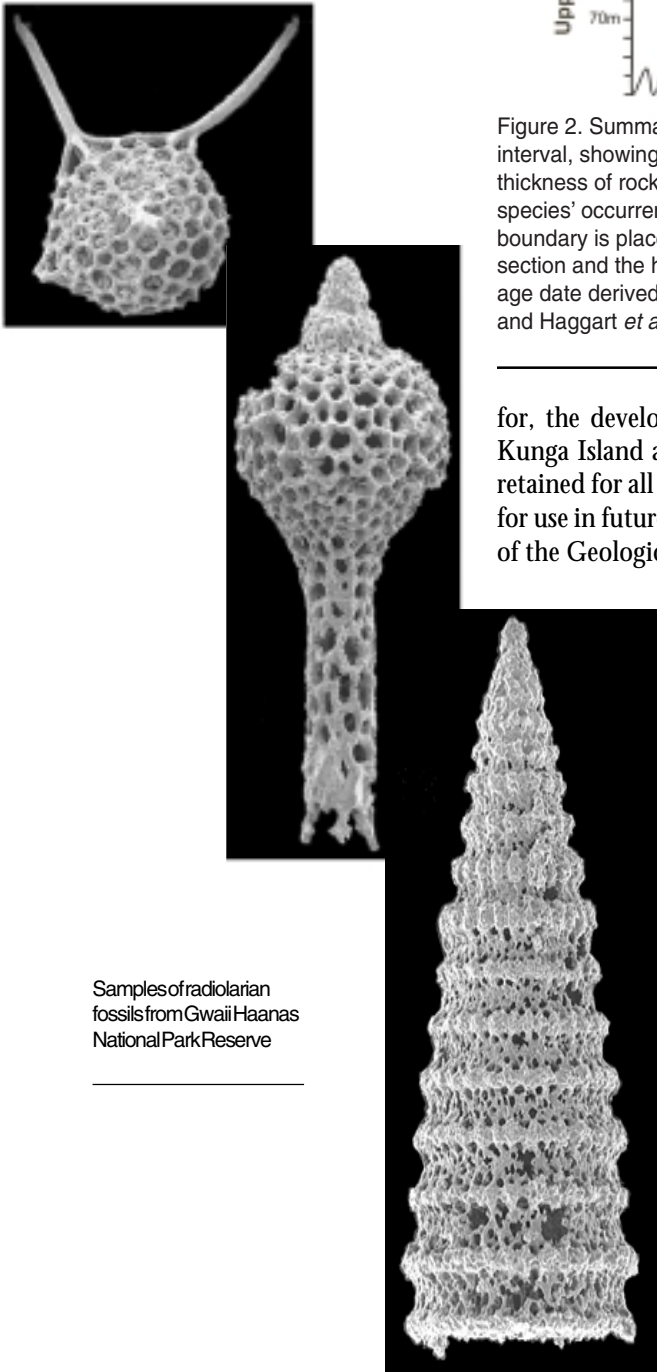
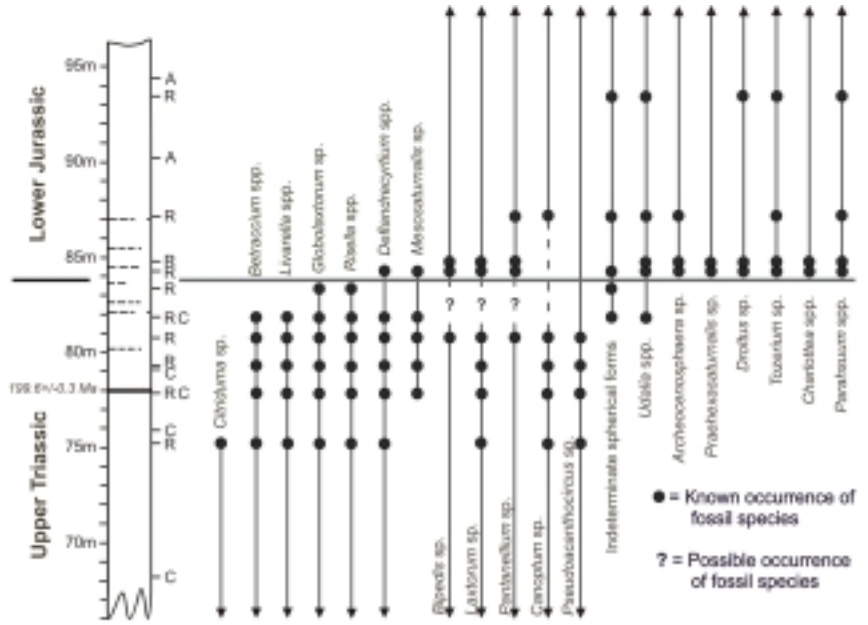Samples of radiolarian fossils from Gwaii Haanas National Park Reserve

Figure 2. Summary schematic stratigraphic section of Kunga Island Triassic/Jurassic boundary interval, showing stratigraphic ranges of important radiolarian microfossil taxa plotted against thickness of rock outcrop section, measured in meters above base. Vertical lines linking a species' occurrences reflect total stratigraphic range of each species. The Triassic/Jurassic boundary is placed at the major radiolarian extinction event found at ca. 84 m in the stratigraphic section and the horizontal line to the right is the level of the end-Triassic extinction. An absolute age date derived from a volcanic tuff is given for the bed at ca. 78 m. Modified after Carter (1993) and Haggart *et al.* (submitted).

for, the development of detailed biostratigraphic and geological records like those of Kunga Island and Kennecott Point. Nonetheless, locality data should be collected and retained for all paleontological collections, and the collections themselves stored securely for use in future studies. As an illustrative example, fossil collections made by field parties of the Geological Survey of Canada on Queen Charlotte Islands in 1872 and 1878 have figured importantly in interpreting the Islands' geological history nearly 130 years later (Haggart 1987). Fortunately, basic locality data for these collections were retained in archival records of the Geological Survey of Canada, allowing them to be used in contemporary research projects; to date, however, accessibility to those data remains limited, and little information is available to resource managers to help design a satisfactory management plan for the Islands' fossil resources.

A MANAGEMENT SOLUTION

As a first step in addressing the issue of fossil resource management on park lands, Gwaii Haanas National Park Reserve (Parks Canada), in partnership with the Geological Survey of Canada (GSC; Natural Resources Canada), initiated a detailed survey of all known fossil localities within the park. Funding has been allocated to review all documentation related to fossil collections made by federal and provincial government field parties over the past 135 years. Locality data for fossil collections have been compiled from examination of original field maps and notebooks archived at the GSC, and then entered into a comprehensive database. The scientific paleontological community has identified specific fossil localities of high scientific importance in the national park reserve, and recommendations

have been proposed for managing and protecting (where necessary) those localities and to guide future scientific collecting.

The fossil resources of all sedimentary rocks of Queen Charlotte Islands, not just those associated with the T/J boundary, have been well-known throughout the international paleontological community for many years, and the islands' fossils have consequently attracted researchers from all parts of the globe. In addition, significant numbers of lay persons are now visiting the islands to examine the more-accessible fossil localities themselves. Given that many localities are found in the intertidal region, it is critical that sites are adequately managed to protect not only the fossil resources themselves, but the associated fauna and flora found at the localities. The

guidelines that have been developed by staff of Parks Canada and the GSC, based on the input of the scientific community, will help to ensure that the limited fossil resources found at these localities are neither exploited nor neglected scientifically. Continued scientific collecting on the islands, including within Gwaii Haanas National Park Reserve, will be encouraged, but within a framework that will ensure that the maximum of scientific information associated with studied localities is retained by both Parks Canada and the GSC.

Through this effort, baseline data on the extent of fossil resources in the National Park Reserve have been compiled and more efficient management of those resources can now be implemented. Perhaps some yet-unstudied stratigraphic horizon in the

reserve will reveal heretofore hidden secrets about the end-Cretaceous extinction of the dinosaurs!

Jim Haggart, Research Scientist, Geological Survey of Canada (Natural Resources Canada), Vancouver, BC. Tel. (604)666-8460; jhaggart@nrcan.gc.ca

REFERENCES CITED

Carter, E.S. 1993. Biochronology and paleontology of uppermost Triassic (Rhaetian) radiolarians, Queen Charlotte Islands, British Columbia, Canada. *Mémoires de Géologie (Lausanne)* **11**, 175 p.

Carter, E.S., and H.W. Tipper. 1999. Candidate Hettangian GSSPs. Kunga Island, Queen Charlotte Islands, British Columbia, Canada. *International Subcommission on Jurassic Stratigraphy, Newsletter* **27**, p. 20

Haggart, J.W. 1987. On the age of the Queen Charlotte Group of British Columbia. *Canadian Journal of Earth Sciences* **24**: 2470-2476.

Haggart, J.W., E.S. Carter, M.J. Beattie, P.S. Bown, R.J. Enkin, D.A. Kring, M.J. Johns, V.J. McNicoll, M.J. Orchard, R. Perry, C.S. Schröder-Adams, P.L. Smith, L.B. Suneby, H.W. Tipper, and P.D. Ward. 2001. Stratigraphy of Triassic/Jurassic boundary strata, Queen Charlotte Islands, British Columbia: potential global system stratotype boundary. *IGCP #458 (Triassic-Jurassic Events), Southwest England Field Workshop, Somerset, 2001, Program and Abstracts*: 10-13.

Haggart, J.W., E.S. Carter, M.J. Beattie, P.S. Bown, R.J. Enkin, D.A. Kring, V.J. McNicoll, M.J. Orchard, R. Perry, C.S. Schröder-Adams, P.L. Smith, L.B. Suneby, H.W. Tipper, and P.D. Ward. [submitted] Stratigraphy of Triassic/Jurassic boundary strata, Queen Charlotte Islands, British Columbia: potential global system stratotype boundary. *Canadian Journal of Earth Sciences*.

Tipper, H.W., E.S. Carter, M.J. Orchard, and E.T. Tozer. 1994. The Triassic-Jurassic (T-J) boundary in Queen Charlotte Islands, British Columbia defined by ammonites, conodonts and radiolarians. *In:* E. Cariou and P. Hantzperque [eds.]. 3rd International Symposium on Jurassic Stratigraphy, Poitiers 1991. *Geobios, Mémoire Spécial* **17**: 485-492.

Tipper, H.W., J.W. Haggart, E.S. Carter, R.L. Hall, G.K. Jakobs, and J. Pálfy. 1998. Field Trip B1: Haida Gwaii (Queen Charlotte Islands). *In:* P.L. Smith [ed.]. *Field Guide for the Fifth International Symposium on the Jurassic System*: 127-229.

Ward, P.D., J.W. Haggart, E.S. Carter, D. Wilbur, H.W. Tipper, and T. Evans. 2001. Sudden productivity collapse associated with the Triassic-Jurassic boundary mass extinction. *Science* **292 (5519)**: 1148-1151.

Movement of wolves (Canis lupus) in response to human development
in Jasper National Park, AB

Jesse Whittington

Human activity may be particularly problematic for wildlife in mountainous regions. These regions are increasingly popular destinations for tourists, outdoor enthusiasts, and new residents. People concentrate their activity in the valley bottoms, which also provide the best habitat for many species (Noss *et al.* 1996). Consequently, human activity has high potential to degrade the quality of limited habitat. Moreover, human activity may impede animal movements across or among valleys where high levels of human activity abut rugged topography. To mitigate habitat degradation and fragmentation effects, it is necessary to determine the movement response of animals to developments such as roads, trails, resorts, and towns (Beier & Noss 1998).

Of the many types of human developments animals face, roads present one of the leading causes of habitat degradation and fragmentation (Trombulak & Frissel 2000, Forman 2000). Several species of large carnivores and ungulates avoid roads or areas of high road density (Mace *et al.* 1996, Mladenoff *et al.* 1995, 1999, Rowland *et al.* 2000, Dyer *et al.* 2001). Wolves in Wisconsin and Michigan, for example, recolonized and persisted in areas with road densities below 0.45 km/km^2 (Mladenoff 1995, 1999). Roads also have high potential to obstruct animal movement across valleys (Clevenger & Waltho 2000, Bélisle & St. Clair 2001). While roads degrade and fragment the habitat of some species, they are used as travel routes by other species. Wolves in remote regions of Alaska and northern Alberta selected areas close to roads and seismic lines (Thurber *et al.* 1994, James & Stuart-Smith 2000). This suggests that the response of wolves to roads may depend on the levels of human activity associated with roads.
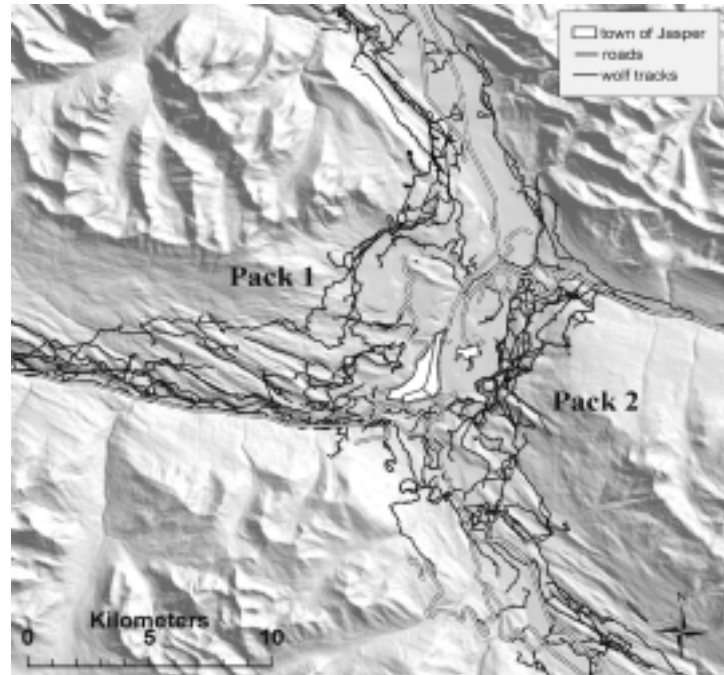

Figure 1. Wolf routes around the town of Jasper from two winters of snow tracking (November 1999 - March 2001). Pack 1 occupies the territory west and north of Jasper, while Pack 2 occupies the territory south and east of Jasper.

The ecological effects of roads are well studied, yet little is known about the cumulative effects of other developments, such as trails, railway lines, and resorts. In fact, only three published studies have quantified the disturbance effects of hikers on animals (Miller & Hobbs 2000, Miller *et al.* 2001, Papouchis *et al* 2001). In one of these studies, bighorn sheep (*Ovis canadensis*) fled 3 times more often from hikers than from vehicles, possibly because hiker's movements are less predictable (Papouchis *et al.* 2001). More research is needed to clarify the cumulative effects of trails, resorts, and railway lines on wildlife movement and habitat quality.

The town of Jasper (population 4,500) in Jasper National Park is a popular destination for tourists and outdoor enthusiasts. Like many communities within the Rocky Mountains, Jasper lies at the confluence of several valleys. Therefore, wary and wide-ranging species such as wolves must circumvent the town to travel among valleys. Parks Canada is concerned that present or future levels of human activity surrounding the town may obstruct wolf movement across valleys. To address this conservation issue I asked the following three questions:

(1) Do wolves avoid or select areas near roads, trails, railway lines, and resorts? If so, does the response of wolves depend on the level of human activity or habitat quality?

Movement of wolves in response to human development

(2) Do wolves travel in more circuitous (tortuous) routes near human developments?

(3) Do wolves avoid crossing roads, trails, and railway lines?

METHODS

This study focused on the winter movements of two wolf packs within 25 km of Jasper for two winters (1999-2000, 2000-2001). To record wolf movements, we conducted valley transects and road surveys to find wolf tracks in the snow. Once we found wolf tracks, we followed the wolves' travel routes and simultaneously recorded our location with a hand-held GPS every 25 m. Fresh tracks were backtracked so as not to interfere with natural movement patterns. Single wolves may respond differently to roads and trails than wolf packs, so tracks of single wolves were excluded from the analysis. Pack 1 consisted of 7 to 9 wolves and traveled north and west of town while Pack 2 consisted of 2 to 3 wolves and traveled south and east of town (Figure 1). The study area included 292 km of roads and 759 km of trails. Roads and trails were classified into high and low-use categories. High-use roads received over an estimated 10,000 vehicles per month in February while low-use roads received less than 10,000 vehicles per month. High-use trails received foot traffic on a daily basis whereas low-use trails received infrequent to no foot traffic.

I used the following approaches to answer my three research questions.

(1) To determine whether wolves avoid or select areas near human developments, I first simplified the wolf tracks into a series of locations separated by 1 km. I then compared the habitat characteristics of each wolf location to ten random locations that defined where the wolves could have traveled given their previous location (Figure 2). For the analysis I used match case-control logistic regression; this pairs wolf and random locations and simultaneously examines the effects of all habitat and human-use variables (Hosmer & Lemeshow 2000).

(2) To identify how human developments affect the tortuosity of wolf paths, I first separated the wolf paths into path segments of three lengths: 0.5, 1, and 5 km. I then measured tortuosity as the ratio of net-displacement to path length for each segment (Turchin 1998). Finally, I examined the effects of habitat and human-use variables on path tortuosity using multiple linear regression.

(3) To test whether wolves avoided crossing linear features, I created 100 random paths for each wolf path and compared the frequency with which wolf and random paths crossed each type of linear feature. Random paths were generated using a biased-correlated random walk (e.g., Shultz & Crone 2001), and were paired with wolf paths in terms of path length, start location, and general path direction. I first tested whether wolves generally avoided crossing all linear features by calculating the proportion of wolf and random paths that crossed each feature type and then running a t-test (paired by feature type) on the differences between the wolf and random proportions. Next, I examined differences among feature types using logistic regression where the dependent variable was "wolf-cross" (Yes/No) and the independent variables were "proportion of random paths that crossed" and "feature type".
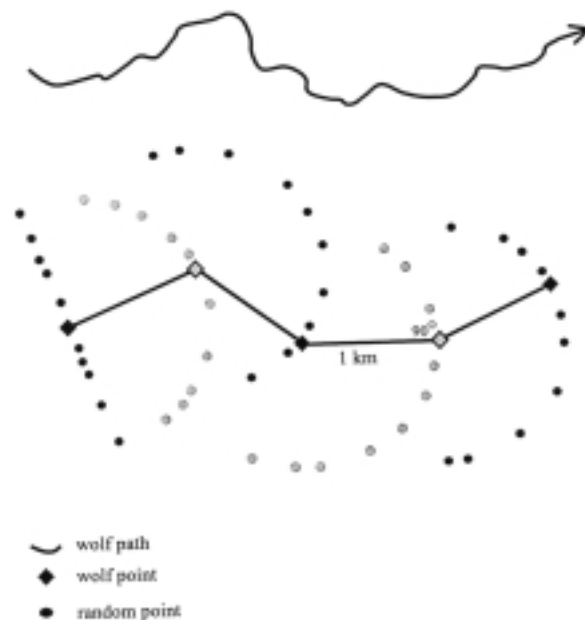

Figure 2. To define habitat available to wolves, the wolf paths were first simplified into a series of locations separated by 1 km. Random locations were then created by adding random turning angles (±90 degrees) to the previous direction of travel.

RESULTS

Over the course of two winters, I snow tracked the two wolf packs 1,390 km. The length of tracking sessions ranged from 0.5 to 30 km with a median length of 5.6 km. The wolves traveled on roads, trails, and railway lines 16% of the time and traveled through the forests, rivers, and meadows the other 84% of the time.

The wolves selected travel routes with low elevations, shallow slopes, and southwest aspects (Figure 3). These are areas where people also concentrate their activity, yet wolves strongly avoided areas of high trail density and high road density. When trail density was analyzed separately, wolves varied their response to trail densities between 0.75 and 2.5 km/km² and strongly avoided trail densities greater than 2.5 km/km². Conversely, wolves strongly selected low-use trails as travel routes. They varied in their response to other linear features but generally avoided resorts, high-use roads, and high-use trails, but selected for low-use roads and railway lines.

The tortuosity of wolf paths increased near predation sites and in areas with high variability in elevation and aspect (Figure 4). Path tortuosity also increased near high-use trails, in areas of high trail density, and in areas of high road density. Road density and aspect were important predictors for 5 km paths only. The strength and predictability of the models increased substantially for the longest path segments (correlation coefficients: r^2 = 0.03, 0.14, 0.51 and n = 667, 321, 54 for path segments of 0.5, 1, and 5 km respectively).

The wolves avoided crossing all linear features in general (t-test paired by feature type: t-value = 7.25, df = 4, p = 0.002), yet none of the features were absolute barriers to wolf movement (Figure 5). Overall, 24.3% of wolf paths and 34.0% of random paths crossed linear features. Wolves were much more likely to cross low-use roads and trails compared to high-use roads (Figure 6). Pack 1 was more likely to cross railway lines and less likely to cross low-use roads than Pack 2.

DISCUSSION

Wolves in this study strongly selected areas with low elevations, shallow slopes, and southwest aspects. These areas are typically associated with shallower snow and higher prey abundance (Telfer & Kelsall 1984,
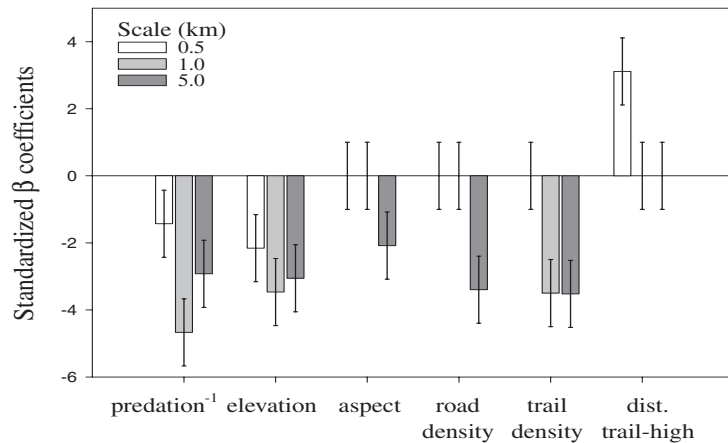
Figure 4. Statndardized β coeficients ±1 standard error for linear regression models predicting path tortuosity. Independent variables are inverse distance to predation site (predation), variation in elevation (elevation), variation in cosine of aspect (aspect), road density, trail density, and distance to high-use trail (trail-high). Models were created for path segments of 0.5, 1 and 5 km.

Huggard 1993), but are also the same areas where people concentrate their activities. Because the wolves avoided areas with high road/trail density and high-use roads/trails, high levels and densities of human activity appear to degrade the quality of wolf habitat. Wolves likely avoided these areas to minimize the probability of encountering people. High-density and high-use features also affected wolf movement behaviour. The tortuosity of wolf paths increased near these features, but the reason why tortuosity increased is unclear. When traveling through these areas, the wolves may have avoided encountering people by selecting vegetative or terrain cover while avoiding trail junctions. Alternatively, when approaching these areas, they may have changed their direction of travel and traveled around rather than through these areas. The second explanation is more likely given that wolves avoided

crossing all linear features and particularly high-use roads and trails. The wolves avoided high-use features, but strongly selected areas near low-use trails. Therefore, the response of wolves to linear features depended on the number of features in an area and the levels of use on those features.

The results from this study may be conservative for three reasons. First, many ungulates concentrate their movements along roads and even within the town limits. If ungulates were not concentrated in these areas, wolves might more strongly avoid human developments. Second, this study occurred in winter when human traffic on trails and roads is very low (<25%) compared to summer months. Wolves may show much greater avoidance during summer when contact with people is much more likely. Third, this study did not examine the diurnal effects of human activity on wolf movement behaviour. For instance, while wolves in this study sometimes preferred areas close to roads and trails, they may only travel in these areas at night when levels of human-use are low. While the results of this study were likely conservative, one limitation of the
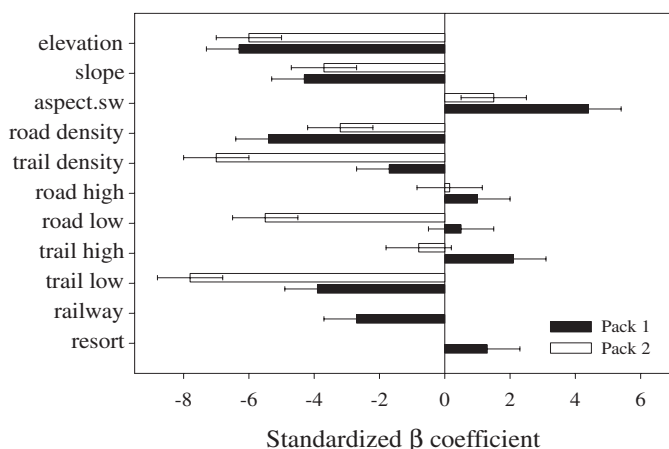
Figure 3. Standardized β coefficients ±1 standard error for a match case-control logistic model predicting wolf occurrence. Positive β coefficients indicate wolves selected for high values of that variable, negative β coefficients indicate wolves selected for low values of that variable.
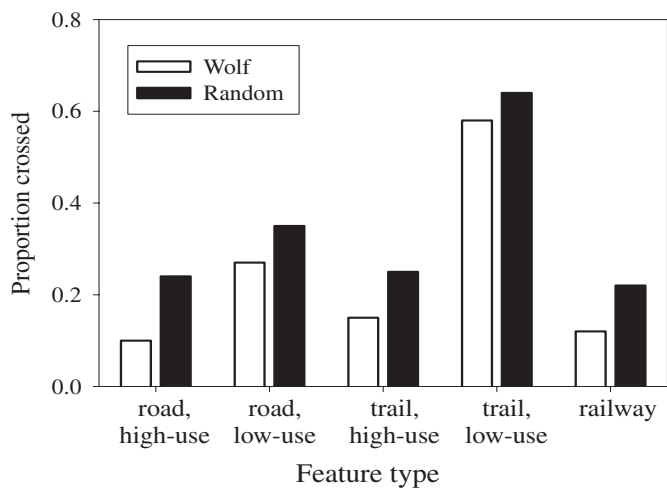
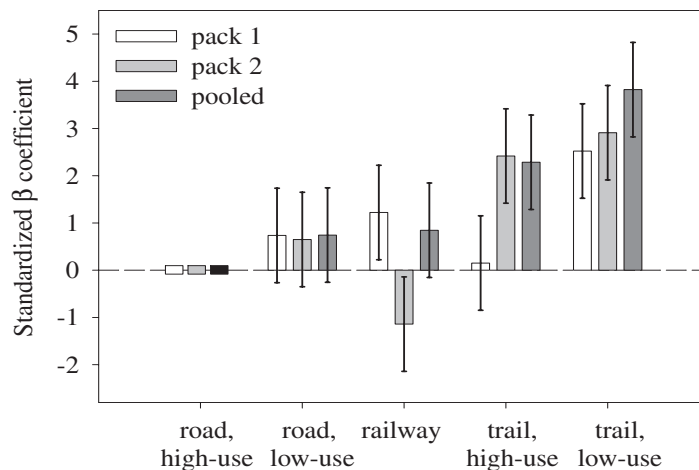Figure 5. Proportion of wolf and random paths that crossed linear features.

Figure 6. Standardized β coefficients ±1 standard error of dummy variables within feature type. High-use roads were the reference category. Logistic regression models were created for each pack and packs were pooled together.

study is that it consisted of a sample size of two wolf packs. This small sample size could not capture the variability of the larger wolf population and thus limits the confidence with which the results can be applied to other regions.

The most surprising result of this study was that trails had a strong effect on wolf movement in all analyses. Moreover, trails had a similar and sometimes stronger effect on model performance compared to roads even though roads receive well over 100 times the daily traffic and present the wolves with a mortality risk. This strong trail effect, also demonstrated by bighorn sheep (Papouchis *et al.* 2001), suggests there is a fundamental difference in how wolves perceive vehicles versus pedestrians. The importance of trails to wolf movement has important conservation implications for Jasper. People in Jasper are creating a rapidly expanding network of unofficial bicycling and hiking trails (G. Mercer, unpublished data). If trails continue to expand to the base of steep-sided mountains and within narrow movement corridors, then wolves may no longer be able to travel between valleys and will therefore be unable to access large tracks of habitat and its prey resources. The effects of

habitat loss, degradation, and fragmentation associated with trails, roads, and other developments are not limited to wolves and would undoubtedly affect complex ecological interactions among trophic levels.

Jesse Whittington is a wildlife research assistant in Jasper National Park. Tel: (780) 852-6187; jesse.whittington@pc.gc.ca. This research was generated for his M.Sc. Thesis in the Department of Biological Sciences, University of Alberta.

REFERENCES CITED

Beier, P., and R.F. Noss. 1998. Do habitat corridors provide connectivity? *Conservation Biology* **12**: 1231-1252.

Bélisle, M., and C.C. St. Clair. 2001. Cumulative effects of barriers on the movement of forest birds. *Conservation Ecology* **5**: 9 [online] URL:http://www.consecol.org/vol5/iss2/art9

Clevenger, A.P., and N. Waltho. 2000. Factors influencing the effectiveness of wildlife underpasses in Banff National Park, Alberta, Canada. *Conservation Biology* **14**: 47-55.

Dyer, S.J., J.P. O'Neill, S.M. Wasel, and S. Boutin. 2001. Avoidance of industrial development by woodland caribou. *Journal of Wildlife Management* **65**: 531-542.

Forman, R.T. 2000. Estimate of the area affected ecologically by the road system in the United States. *Conservation Biology* **14**: 31-35.

Hosmer, D.W., and S. Lemeshow. 2000. *Applied logistic regression*, 2nd edition. John Wiley & Sons, New York, USA. 375pp.

Huggard, D.J. 1993. Effect of snow depth on predation and scavenging by gray wolves. *Journal of Wildlife Management* **57**: 382-388.

James, A.R.C., and A.K. Stuart-Smith. 2000. Distribution of caribou and wolves in relation to linear corridors. *Journal of Wildlife Management* **64**: 154-159.

CONSERVING WHITEBARK PINE
in the Canadian Rockies

Jon Stuart-Smith, Brendan Wilson, Rob Walker and Ellen Macdonald

Whitebark pine (*Pinus albicaulis*) is a key component of the subalpine ecosystem of the Canadian Rocky Mountain Parks (see photo). The range of whitebark pine consists of two major areas throughout the mountains of western North America (Arno & Hoff 1989). This ranges includes the area covered by Waterton Lakes, Banff, Yoho, Kootenay Jasper, Mt. Revelstoke and Glacier National Parks. Whitebark pine is often found at treeline in small, isolated populations on exposed ridges and rocky talus slopes at the northern-most extent of its range in the Canadian Rockies (Ogilvie 1990). Despite this extreme environment, whitebark pine plays a vital role in the ecology of the subalpine ecosystem. Many other species, including birds, squirrels, and bears, feed on whitebark pine seeds (Kendall 1983, Mattson *et al.* 1992, Lanner 1996).

One species of bird in particular, Clark's Nutcracker *(Nucifraga columbiana)*, has developed a very close relationship with whitebark pine, to the point where the two species depend on each other for their survival (Tomback & Linhart 1990). Clark's nutcrackers have a sublingual pouch in which they can hold whitebark pine seeds (Bock *et al.* 1973). With a full pouch, nutcrackers fly to a suitable site, make small caches just below the soil surface that are retrieved at a later time (Tomback 1982).

Whitebark pine also has adaptations that accommodate seed dispersal by Clark's nutcrackers. Unlike serotinous species such as lodgepole pine, the cones of whitebark pine do not require fire to open the cones to allow dispersal of the seeds (Arno & Hoff 1989). In fact, whitebark pine cones never open unless they are ripped open by animals. Lacking even a rudimentary wing, whitebark pine seeds cannot be carried by the wind and depend on Clark's

nutcrackers for dispersal (Tomback 1982). Although Clark's nutcrackers depend on whitebark pine seeds as a food source, the relationship between whitebark pine and this bird species is mutualistic (Tomback & Linhart 1990). Without seed caching by nutcrackers, no whitebark pine regeneration would occur. Clark's nutcrackers prefer to cache seeds in open areas such as burns so that they are easier to retrieve (Tomback 1982). This behaviour accommodates the need that whitebark pine seedlings have for high levels of sunlight.

Because whitebark pine is not a commercialy important species in the forest industry, very little attention has been paid to the species. Over the last decade research in the United States has indicated that whitebark pine is in trouble and that the important role it plays may be in jeopardy. Keane & Arno (1993) point to three causes in the decline of whitebark pine: white pine blister rust, mountain pine beetle and fire suppression.

White pine blister rust is an fungus from Eurasia that was accidentally introduced to North America around 1900 (McDonald & Hoff 2001). Since then, it has spread throughout the native five-needled pines causing extensive mortality. A number of studies in the United States had shown high levels of blister rust infection on whitebark pine (Keane & Arno 1993) but in 1996, still no data on the level and extent of blister rust infection in Canada was available. This study was initiated by Parks Canada to determine if a blister rust epidemic similar to what was being documented in the United States was happening in the Canadian Rockies. This article describes data collected from this initial project (Stuart-Smith 1998) and discusses some of the work that has occurred since then on the conservation of whitebark pine in the Canadian Rocky Mountain Parks.

METHODS

Twenty-nine whitebark pine populations throughout the Canadian Rockies were sampled during the summer of 1996. Within a sample plot of 100 by 200 meters, 25 whitebark pine were randomly chosen and examined for evidence of blister rust infection. The number of cankers on each tree was also recorded so that the intensity of infection could be examined in more detail. Site characteristics such as slope, aspect and elevation were also recorded for each site.

RESULTS AND DISCUSSION

The level of blister rust infection ranged from 0% of the 25 trees sampled showing signs of infection to 76% of individuals with

RESEARCH L

ECOLOGICAL LAND CLASSIFICATION
OF WATERTON LAKES NATIONAL PARK

View of Blackiston

Cyndi Smith and Peter Achuff

Waterton Lakes National Park occupies about 525 sq km in the Rocky Mountains of extreme southwestern Alberta. The Ecological Land Classification of the park is an integrated resource inventory of landform, soils, vegetation and wildlife information presented in both report and 1:20,000 map format (Achuff *et al.* 2002a, b); field work was conducted between 1994 and 1997. The map information is in both hard copy and computerised Geographic Information System (GIS) form.

Four Ecoregions were recognised: Foothills Parkland, Montane, Subalpine, and Alpine. The Subalpine Ecoregion is divided into Lower Subalpine and Upper Subalpine based on vegetational characteristics that reflect macroclimatic differences. WLNP is the only national park in Canada containing a portion of the Foothills Parkland Ecoregion.

The vegetation type and ELC systems for WLNP are part of a broader framework encompassing Jasper, Banff, Yoho, Kootenay, Mt Revelstoke and Glacier national parks. While the four more northerly Rocky Mountain parks have many vegetation types and land classification units in common, the distinctness of WLNP is indicated by a low

degree of overlap in both vegetation types (4%) and Ecosites (0%) between WLNP and the other parks.

A total of 971 vascular plant species were identified, including 82 non-natives. Twenty species new to the park (nine native, 11 non-native) were found during this study, including two species new to Alberta and one new to Canada.

Six species of amphibians and four species of reptiles were recorded in Waterton Lakes National Park (Wallis *et al.* 2002). The avifauna of WLNP includes at least 256 species, including 149 that nest, or are suspected of nesting, in the park. Of the 149 nesting species, 112 are summer residents and 37 are permanent residents or probable permanent residents. The small mammal fauna (bats, shrews, mice, voles, squirrel, chipmunks, marmot, pocket gopher, ground squirrels, beaver, muskrat, pika, hare) of WLNP includes 35 known species.

Since the completion of the ELC the data has been used in a number of different ways,

including to identify special plant and landscape features, and to correlate wolverine sightings and tracks with ecosite characteristics.

The ELC data is being used in an ambitious project, initiated in 1998, to classify, describe and map vegetation communities in Waterton-Glacier International Peace Park (IPP), using a common vegetation classification system and mapping protocol. The proposed products (digital files of vegetation maps, keys to the vegetation classes, plot data, vegetation and geospatial metadata, photos of vegetation types, interpreted aerial photos, hard copy maps, and map accuracy reports) were seen as a valuable data resource to help support a wide variety of IPP management needs including a broader ecosystem approach to park planning and management. In the early stages park managers identified a number of ways they anticipated using the products; two examples being to model habitat available for carnivores such as grizzly bears, and to delineate whitebark and limber pine mortality.
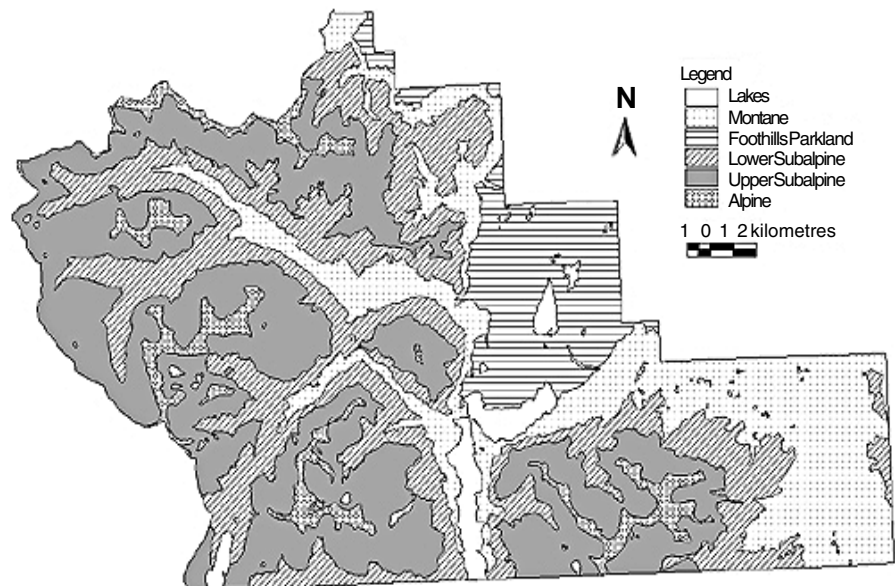
The IPP vegetation classification will be derived from qualitative and quantitative analysis of data for plots sampled within the entire IPP, including 276 vegetation plots sampled in WLNP as part of the ELC, 54 additional WLNP plots sampled in 1999 to address gaps from the ELC, and over 900 plots sampled within Glacier National Park, beginning in 1999. Plot data has been incorporated into the PLOTS database of the U. S. National Vegetation Classification System (USNVC). Botanists involved in the project have jointly defined a preliminary vegetation classification for WLNP and the east side of GNP using qualitative analysis of available plot data. The process relied heavily on WLNP's vegetation classification done as part of the ELC.

The preliminary vegetation classification for WLNP and GNP (east side) was needed to define map classes so that mapping could begin. Thirty-six map classes, which are recognizable on aerial photographs, have been agreed upon by classifiers and mappers to represent the key vegetation types. Interpretation of 1:15,840 air photos for WLNP (~180 photos) and the east side of GNP (~600 photos) currently is underway. Minimum map unit is 0.5 hectares. Once interpretation is complete the interpreted photo overlays will be scanned into digital format, rectified to an ortho map base and mosaiced together. Accuracy assessment of the vegetation classification and mapping for WLNP and the east side of GNP will be conducted in the field in 2003/04. Accuracy requirements for the project specify 80% accuracy for each map class.

The major participants for the IPP project are: Glacier National Park, Waterton Lakes National Park, Montana Natural Heritage Program, The Nature Conservancy (now NatureServe), and the U.S. Geological Survey's Upper Midwest Environmental Sciences Center (UMESC). Completion of this project is expected by 2006.

Cyndi Smith, Conservation Biologist, cyndi.smith@pc.gc.ca

Peter Achuff, Species-at-Risk Botanist, peter.achuff@pc.gc.ca

REFERENCES CITED

Achuff, P.L., R.L. McNeil, M.L. Coleman, C. Wallis, and C. Wershler. 2002a. *Ecological land classification of Waterton Lakes National Park, Alberta*. Vol. I: Integrated Resource Description. Parks Canada, Waterton Park, Alberta. 226 pp.

Achuff, P.L., R.L. McNeil, M.L. Coleman, C. Wallis, C. Wershler, and R. Riddell. 2002b. *Ecological land classification maps of Waterton Lakes National Park, Alberta*. Prepared by Terrain Resources Ltd., Lethbridge, Alberta, for Parks Canada, Waterton Park, Alberta. 5 maps & legend.

Wallis, C., C. Wershler, and R. Riddell. 2002. *Ecological land classification of Waterton Lakes National Park, Alberta*. Vol. II: Wildlife Resources. Parks Canada, Waterton Park, Alberta. 258 pp.

2002 WCSC Archaeological Field Activities
in BC Coastal Field Unit

Ian Sumpter

This past summer, Cultural Resource Services staff (WCSC) in Victoria assisted the Huu-ay-aht First Nation in field investigations at their Kiix?in NHS, near Bamfield on the west side of Vancouver Island. Kiix?in NHS is a unique Nuu-chah-nulth village site in that it boasts still-standing longhouse remains and adjoining defensive fort. The fieldwork, brought about through the site's Commemorative Intent Statement, focussed on two tasks. These included: detailed mapping of architectural, archaeological, and topographic features in the

Pacific Rim park CRM Officer Arlene Suski escorted over mud-flats in Bamfield Inlet by Denis St. Claire.

village coupled with the acquisition of samples from cedar house posts, roof beams, and underlying cultural sediments for dendrochronological and radiometric dating. Mapping, dating, and analytical results are expected before January 2003.

The fieldwork at Kiix?in was successful and the experience rewarding because of an integrated team approach. Team participants included Kiix?in band members (Stella Peters, Lonnie Nookemus, Karen Haugen), Parks Canada staff (Ian Sumpter, Arlene Suski), Department of Public Works (surveyor Pat McFadden), University of Victoria's Dendrochronology Lab (led by Dr. Dan Smith), and volunteer archaeologists (Al Mackie, Denis St. Claire, Alex Clarke).

Archaeological field activities are anticipated this fall at Fort Rodd Hill/Fisgard Lighthouse National Historic Sites. Proposed wharf improvements in Esquimalt Harbour will involve an underwater archaeological assessment study by ARCAS Consulting Archaeologists, Vancouver. An assessment of a new washroom facility will be carried out by WCSC archaeologists Ian Sumpter and Daryl Fedje, with assistance from Songhees and Esquimalt heritage specialists.

Ian Sumpter, Assistant Archaeologist, ian.sumpter@pc.gc.ca

cankers (Figure 1). The average level of infection over all 29 sites was 30%. The intensity of infection was strongly correlated with latitude (r^2=0.49, $p \leq 0.01$) with the highest rates of infection south of the Crowsnest past. The average number of cankers per tree was also much greater in the southern populations. The average number of cankers per tree across all populations was 0.61, and of those trees that were infected there was an average of 2.02 cankers per tree. This shows that infected trees were likely to have more than one canker. Although there were a number of sites where none of the sampled individuals showed signs of infection, blister rust cankers were found on unsampled trees at all 29 sites. Over the course of the study and subsequent investigations throughout the Mountain Parks, no sites have been found to be free from blister rust although many have very low levels of infection.

Figure 1. Map showing the level of blister rust infection.

pine because more shade tolerant species may be out competing whitebark pine (Keane & Arno 1993). The total area burned annually by fire in the mountain parks has declined over the past century (Parks Canada unpublished data), which has reduced the area available for whitebark pine regeneration. By using low intensity fire to remove competing species such as subalpine fir and Engelmann spruce, regeneration of whitebark pine should increase, and the importance of whitebark pine should be maintained on the landscape. One site in Banff National Park was burned in 1998. Three other study locations, including a site in Waterton Lakes National Park, have been surveyed and when the right conditions arise they will also be burned.

Data collected during the prescribed burn program have added further details to the picture of whitebark pine decline. Mortality rates, data that were not collected during the initial study, have been documented at the four prescribed burn locations. Although data analysis has not been completed on all of these sites, mortality ranged from 0-20% with the highest level of mortality experienced in the Waterton Lakes area. Similar levels of mortality have been estimated for Canadian whitebark pine populations with levels reaching as high as 23.2% in the Selkirk Mountains (Campbell & Antos 2000, Zeglan 2002).

The high level of blister rust infection found in the southern Canadian Rockies during this study emphasizes the decline being experienced by whitebark pine. Other studies of whitebark pine in British Columbia have found similar epidemic levels of blister rust infection. In two separate studies the intensity of infection ranged from only a few infected individuals infected at sites in the central and southern Coast Mountains, to over half of the trees infected at sites in the southeastern corner of the province (Campbell & Antos 2000, Zelgan 2002). Studies in the United States have also shown areas with high levels of blister rust infection throughout western Montana, northern Idaho and the Cascade Mountains of Washingtion (Kendall & Keane 2001).

With data from this and other studies showing the threat that white pine blister rust poses to whitebark pine, a number of conservation and restoration programs have been initiated in the United States and more recently in the Canadian Rocky Mountain Parks. In 1998, a prescribed burn program was initiated in the Lake Louise, Yoho and Kootenay National Park Field Unit with the goal of maintaining healthy whitebark pine stands (Wilson *et al.* 1998). The primary objective of using prescribed fire is to increase the number of desirable areas available for nutcracker seed caching. Fire suppression may also be adding to the decline of whitebark

We plan to return and re-measure regeneration at the prescribed burn sites at least every five years to determine the changes that occur in a landscape over time. Control plots adjacent to the burns will allow us to determine how the vegetation and structure change over time and enable us to evaluate whitebark pine regeneration and ensure the species' survival in the Canadian Rockies.

Although this initiative has been focused within areas under Parks Canada management, the development of the program has drawn on input from experts throughout North America. Other agencies, such as the Forest Service in the United States have been carrying out a similar prescribed burn program for whitebark pine conservation. Global Forests, a private research organization, is helping develop a whitebark pine program in the West Kootenays in

conjunction with logging operators. Future developments will include cooperation with these and other conservation partners and the exchange of our results will further the success of our common goal.

Jon Stuart-Smith is a Park Warden, Lake Louise, Yoho and Kootenay Field Unit. Box 213 Lake Louise, AB. T0L 1E0. Tel: (403)522-1217; fax: (403)522-1223; Jon.Stuart-Smith@pc.gc.ca.

Brendan Wilson is a research consultant with Cordilleran Ecological Research, RR 1 Groupbox 15A, Comp7, Winlaw BC V0G 2J0. Tel/fax: (250) 226-7582; wilsonb@netidea.com.

Rob Walker is the Fire and Vegetation Specialist for the Lake Louise, Yoho and Kootenay Field Unit. Box 220 Radium Hot Springs, B.C. V. Tel: (250) 347-6155; fax: (250) 347-6150; Rob.Walker@pc.gc.ca.

Ellen Macdonald is a Proffessor and Associate Dean in the Department of Renewable Resources at the University of Alberta. 751 General Services Building, Edmonton, AB. T6G 2H1 Tel: (780) 492-3070; fax: (780) 492-4323; ellen.macdonald@ualberta.ca.

REFERENCES CITED

Arno, S. F. and Hoff, R. J. 1989. Silvics of Whitebark pine (Pinus albicaulis). USDA Forest Service, General Technical Report, INT-253. Intermountain Research Station, Ogden, UT 84401. 11pp.

Bock, W. J, Balda, R. P., and Vander Wall, S. B. 1973. Morphology of the sublingual pouch and tongue musculature in Clark's Nutcracker. Auk. 90:491-519.

Campbell, E. M. and Antos, J. A. 2000. Distribution and severity of white pine blister rust and mountain pine beetle on whitebark pine in British Columbia. Canadian Journal of Forest Research 30:1051-1059.

Keane, R. E. and Arno, S. F. 1993. Rapid decline of whitebark pine in western Montana: evidence from 20-year remeasurements. Western Journal of Applied Forestry 8(2):44-47.

Kendall, K. C. 1983. Use of pine nuts by grizzly and black bears in the Yellowstone area. International Conference on Bear Research and Management 5:166-173.

Kendall, K. C. and Keane, R. E. 2001. Whitebark pine decline: infection, mortality, and population trends. Pages 221-242 in: Tomback, D. F., Arno, S. F., and Keane, R. E., editors. Whitebark pine communities: ecology and restoration. Island Press, Washington, D.C. 440pp.

Lanner, R. M. 1996. Made for each other. A symbiosis of birds and pines. Oxford University Press, New York, NY. 160pp.

Mattson, D. J., Blanchard, B. M., and Knight, R. R. 1992. Yellowstone grizzly bear mortality, human habituation, and whitebark pine seeds crops. Journal of Wildlife Management 56(3):432-442.

McDonald, G. I. and Hoff, R. J. 2001. Blister Rust: An Introduced Plague. Pages 193-220 in: Tomback, D. F., Arno, S. F., and Keane, R. E., editors. Whitebark pine communities: ecology and restoration. Island Press, Washington, D.C. 440pp.

Ogilvie, R. T. 1990. Distribution and ecology of whitebark pine in Western Canada. Pages 54-60 in: Schmidt, W. C. and McDonald, K. J., Compilers. Proceedings - Symposium on whitebark pine ecosystems: ecology and management of a high mountain resource. USDA Forest Service, General Technical Report INT-270, Intermountain Research Station, Ogden, Utah. 386pp.

Stuart-Smith, G. J. 1998. Conservation of whitebark pine in the Canadian Rockies: blister rust and population genetics. M.Sc. thesis, University of Alberta, Edmonton, AB. 125pp.

Tomback, D. F. 1982. Dispersal of whitebark pine seeds by Clark's nutcracker: a mutualism hypothesis. Journal of Animal Ecology 51:451-467.

Tomback, D. F., Anderies, A. J., Carsey, K. S., Powell, M. L., and Mellmann-Brown, S. 2001. Delayed seed germination in whitebark pine and regeneration patterns following the Yellowstone fires. Ecology 82(9): 2587-2600.

Tomback, D. F. and Linhart, Y. B. 1990. The evolution of bird-dispersed pines. Evolutionary Ecology 4:185-219.

Wilson, B. C., Stuart-Smith, G. J., and Walker, R. 1998. Ecosystem restoration in the Rocky Mountain National Parks. Poster presented at Ecosystem Management of Forested Landscapes - Symposium. Nelson, B.C.

Zeglan, S. 2002. Whitebark pine and white pine blister rust in British Columbia, Canada. Canadian Journal of Forest Research 32:1265-1274.

POPULATION VIABILITY ANALYSIS APPLIED TO WOODLAND CARIBOU IN JASPER NATIONAL PARK

Kyla Flanagan and Salman Rasheed

To maintain viable wildlife populations, biologists need to estimate the size of the population that is likely to allow for the population's persistence. To understand the relationship between a population's size and the probability of persistence, biologists have established the concept of Minimum Viable Population (MVP) (Gilipin & Soule 1986). The MVP is defined as the smallest isolated population having an adequate probability of remaining extant for a defined period of time, despite the foreseeable effects of demographic, environmental, and genetic stochasticity as well as natural catastrophes (Shaffer 1981). The process of determining the MVP is referred to as Population Viability Analysis (PVA)(Soulé 1987, Shaffer 1990). PVA involves estimating population extinction probabilities and incorporating threats to the population's survival into a model, which projects the population into the future (Lacy 1993). The purpose of this paper is to briefly outline PVA methodology and use a case study to demonstrate the application of two different PVA models to a caribou population in Jasper National Park (JNP).

BACKGROUND

The concept of Population Viability Analysis arose from research showing there is a critical population size below which a population falls dramatically to extinction (MacArthur & Wilson 2001; Richer-Dyn & Goel 1972; Leigh 1975; Gilpin & Soule 1986), and that smaller populations have higher rates of extinction than larger populations (Ricklefs 1997).

Since its conceptual beginning in the early 1980's, PVA techniques have become increasingly sophisticated and the use of PVA as a conservation tool more commonplace. Numerous computer software packages have been developed and are easily accessible to resource managers (e.g., VORTEX – http://pw1.netcom.com/~rlacy/vortex.html; RAMAS - http://www.ramas.com/pva.htm). More recent application of PVA has involved a wide range of species and a variety of management questions (see Noon & Biles 1990 – Northern

Spotted Owl; Armbruster & Lande 1993 – African Elephant; Bergland 2000 – Sand Lizard; Herrero *et al.* 2000 – Grizzly Bear; Lennartsson 2000 – Pasture Plant; Taylor & Plater 2001 – Killer Whale).

POPULATION VIABILITY ANALYSIS METHODOLOGY

Population Viability Analysis (PVA) is designed to use demographic, life history, and environmental parameters to create a population and step it through annual cycles of reproduction and death until either extinction occurs or a specified time limit is reached. Typically a computer program will repeat this process for many simulated populations and combine the results to estimate the likelihood of specific outcomes. More sophisticated simulation models include genetic factors, random environmental fluctuations and catastrophic events. PVA simulation modeling also shows what kind of missing information is needed to more accurately assess a population's status and identifies those characteristics that influence the viability of the population. PVA cannot be performed by a single recipe, rather data availability, the degree to which the species ecology and life history are understood, knowledge of risk factors, and management goals should dictate the PVA methodology. PVA models range from relatively simple analytical models to more complex spatially explicit, metapopulation models (see Beissinger & Westphal (1998) for a description of the general categories of PVA models).

CASE STUDY – WOODLAND CARIBOU IN JASPER NATIONAL PARK

Studies of Woodland Caribou (*Rangifer tarandus* Linnaeus) have been going on since the early-70s (Stelfox 1974), and a monitoring program
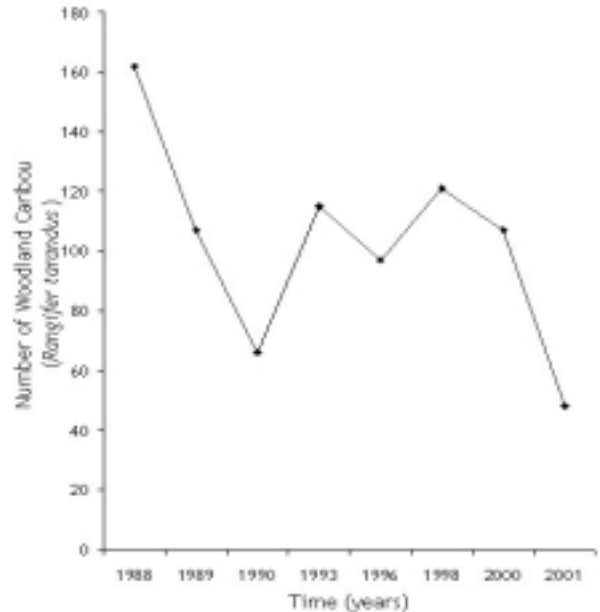

Figure1. The estimated population of Woodland Caribou *(Rangifer tarandus)* in Jasper NP from 1988-2001.

funded by Parks Canada and the World Wildlife Fund was initiated in 1988 (Brown *et al.* 1994). We used two different PVA simulation models to predict the population trajectory for Woodland caribou in Jasper National Park. To illustrate the range in complexity of PVA models, we chose a simple model incorporating count data (Dennis Method for Count-Based Extinction Analysis) and a more complex model incorporating a number of demographic parameters (VORTEX). We discuss the methodology for both analyses very simply and readers are encouraged to look for more detail in the appropriate references.

DENNIS METHOD FOR COUNT DATA

Methods

The Dennis Method of Count-Based Extinction Analysis is a relatively simple method to calculate population viability (Dennis *et al.* 1991). The Dennis Method allows managers to use routinely collected, count data to quantify the risk of extinction by incorporating previous

population trends and characteristics into viability predictions (Morris *et al.* 1999).

Morris *et al.* (1999) provides detailed instruction on calculating these parameters for the Dennis Method using count data. Count data for Woodland Caribou in JNP has been collected since 1988 (Figure 1).

A preliminary step in determining population viability using the Dennis Method involves regressing transformed population data (Figure 1) to estimate two parameters, first, μ, which governs change in the mean of the normal distribution and second, σ^2, which governs how quickly the normal distribution's variance will increase over time (Table 1). These parameters allow us to estimate continuous rate of increase (r), average finite rate of increase (l) and mean time to extinction.

Results

The Dennis Method estimated an m of -0.09357, with the population variability (σ^2) estimated at 0.1798. From these parameters, the continuous rate of increase (r) was –0.0037 (0.07, -0.08 = upper and lower 95% CI) and the average finite rate of increase (λ) was 0.996 (1.075, 0.923 = upper and lower 95% CI). The mean time to extinction was calculated at 41.4 years (135.6, 0 = upper, lower 95% CI). The number of years for which the population has less than 5% chance of surviving was estimated at 46 years (Figure 2).
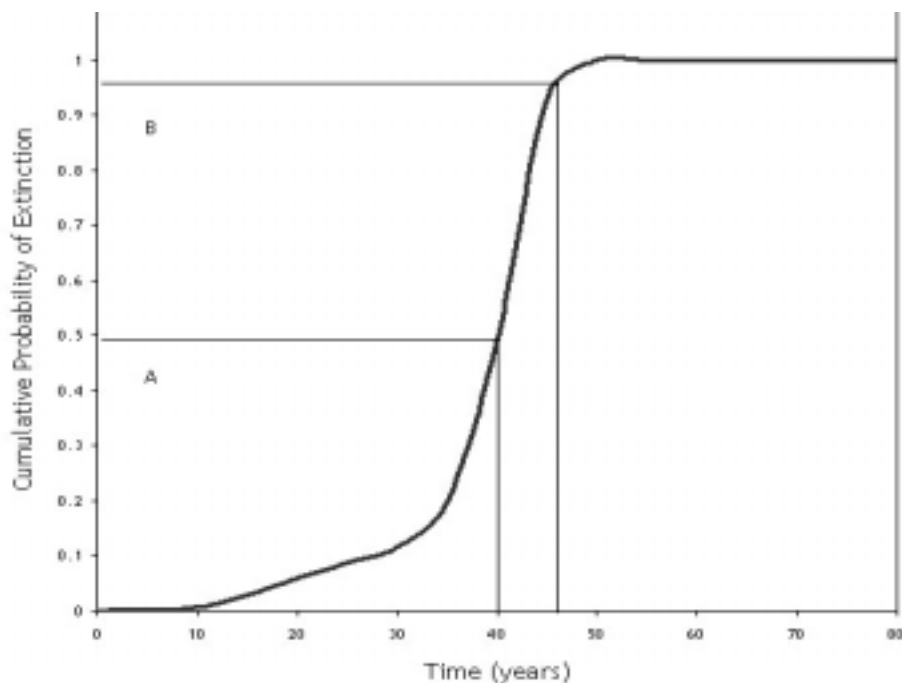
VORTEX POPULATION VIABILITY ANALYSIS MODEL

Figure 2. The cumulative distribution function of extinction time for the Jasper NP Woodland Caribou *(Rangifer tarandus)* estimated from the Dennis Method. A) indicates the median time to extinction. B) indicates the number of years for which there is a 5% chance of surviving.

VORTEX is a simulation software package that models the effect of demographic rates, demographic stochasticity, environmental variation, catastrophes, inbreeding depression, harvesting, supplementation, and metapopulation structure on a population's viability (Lacy 2000). The user can model functional relationships between demographic parameters (e.g., birth and death rates) and parameters (e.g., age, sex, population size) by including mathematical equations for these relationships. This allows the incorporation of functional relationships such as density dependence. The output of VORTEX simulations is statistics on population growth rates, probability of population extinction, extinction time and genetic variation in extant populations

(Lacy 1993). VORTEX also allows a sensitivity analysis that can:
a) examine the effects of a number of different management options,
b) determine the parameters that are most influential to the viability of the population,
c) identify parameters that must be defined more precisely to produce meaningful predictions.

Methods

The VORTEX simulation lasted 100 years and we performed 500 iterations for each simulation. We assumed a polygynous mating system, with all males in the breeding pool, equal sex ratio at birth and no harvesting or supplementation of the population. We also conducted a sensitivity analysis to assess the uncertainty associated with some of the parameters. The sensitivity analysis consisted of one simulation using minimum estimates and another using maximum estimates, with all other parameters held constant. Parameter estimates were derived from research (e.g., Brown *et al.* 1994) or personal communication (Mercer, *pers.comm.*, Smith, *pers.comm.*). See Table 2.

Table 1. Analysis of Variance Table (ANOVA) for the regression performed on the transformed Jasper NP Woodland Caribou count data. The X-Intercept coefficient estimates and the MS Residual estimates σ^2 (see text for description).

	Df	SS	MS	F	P value
Regression	1	-0.1693	-0.9683	-0.9419	> 0.05*
Residual	6	1.0787	**0.1798**		
Total	7	0.909			

	Coef.	SE	t-statistic	P value	Lower 95%	Upper 95%
X-Intercept	**-0.0936**	0.118	-0.796	0.457	-0.381	0.194

* a non-significant regression does not affect using the estimated and σ^2 because the regression is being used to find the best-fit values of given the data, rather than testing any particular hypothesis.

Caribou viability analysis applied to Woodland Caribou in Jasper National Park

Results

The VORTEX simulation using the best estimates of parameters estimated a rate of increase (r) of -0.0958 ± 0.0013 ($\pm$SE) and a mean time to extinction of 39.89 ± 0.45 years. All simulations using the estimated parameters drove the population to extinction within 100 years.

Sensitivity Analysis

Figures 3 and 4 show the sensitivity of the mean time to extinction and the mean growth rate (r) to the parameters used in the VORTEX model. Both mean time to extinction and mean growth rate are most sensitive to age at first reproduction. Mean time to extinction is also sensitive to initial population size, calf mortality, yearling mortality and female fecundity (Figure 3). Inbreeding depression, Calf mortality, yearling mortality and female fecundity influence mean growth rate (Figure 4). Mean time to extinction and mean growth rate appear to be relatively insensitive to all other model parameters.

DISCUSSION

Both analyses predicted that the JNP's caribou population will go extinct if conditions do not change. Interestingly, although using different data sources and methodologies, both PVA analyses predicted a mean time to extinction of about 40 years. The estimate for mean time to extinction for the Dennis method had large confidence limits likely a result of variation associated with the current count data (e.g., sampling variation, counts no conducted every year). In contrast the VORTEX estimate for mean time to extinction had a small measure of variation, likely a function of the more detailed model input.

More significantly, both analyses showed a declining population (negative *r*), which clearly does not bode well for the long-term persistence of the caribou.

Population viability analysis can provide useful insight into the potential outcome for a population. However, PVA depends on numerous assumptions (e.g., PVA's best applied to long-lived species with low fecundities, random mating, uniform age-specific fecundity or reproductive capacity and no time-trends/cycles in mortality or fecundity

Table 2. Estimated demographic parameters used for the JNP Woodland Caribou VORTEX simulation. The best estimate was used to detrmine the predicted growth rate and time to extinction associated with this range, and the minimum and maximum were used in the sensitivity analysis.

Model Parameter	Minimum	Best Estimate	Maximum
Metapopulation structure	No metapopulation structure was assumed and the simulation only modeled a single population		
Inbreeding depression*	1	3.14	6
Catastrophic events[a]	0	0.05	0.1
Age at first reproduction	2	2.5	3
Maximum breeding age	13	14	15
Adult female mortality rate[b]	0.313	0.3145	0.316
Adult male mortality rate[b]	0.344	0.3455	0.347
Carrying capacity	500	725	875
Initial population size[b]	48	150	200
Calf mortality (%)[c]	5.5	17.3	35
Yearling mortality (%)	10	20	30
Female fecundity (%)	70	81	94

*Number of lethal equivalents was set at 3.14 based on 40 different captive populations (Ralls *et al.* 1988), where lethal equivalents is a measure of inbreeding depression.
[a] Probability of a catastrophic event once every 20 years = 0.05
[b] Measured for female and male JNP populations.
[c] From Seip (1992)

(i.e. predator-prey cycles)), which are often violated. Violation of these assumptions causes the stochasticity of the population to be underestimated; as a result, the predicted risk to the population is also underestimated (Taylor & Plater 2001). For our case study, the empirical data is unavailable to assess assumption violation (e.g., Brown *et al.* (1994)). However, our inability to test assumptions should not affect the overall predictions of the PVA. For example, the JNP caribou went extinct in every VORTEX simulation within the variation associated with each parameter.

MANAGEMENT IMPLICATIONS

For the Woodland caribou in JNP, clearly some action is necessary to address the current decreasing population trend, especially in light of the precautionary principle. The PVA sensitivity analysis showed that age at first reproduction would most affect the population but altering age at first reproduction for large ungulates is not easily done. However, the sensitivity analysis also showed that decreasing calf and yearling mortality would increase mean time to extinction, which could be a first step in conservation. For example, a first step could be identifying direct and indirect sources of

mortality and minimize local disturbance regimes. In fact, currently, a project is being proposed in JNP to assess caribou movement patterns as a result of disturbance. In addition, several of the estimates in the VORTEX simulation require empirical validation, and this would provide, not only more information about this caribou herd, but also provide a more data-driven, robust estimate for the viability of this population.

Kyla Flanagan is a student at the University of Calgary.

Salman Rasheed is the Ecosystem Conservation Specialist at the Western Canada Service Centre, Calgary. Tel: (403) 292-4748; Sal.Rasheed@pc.gc.ca

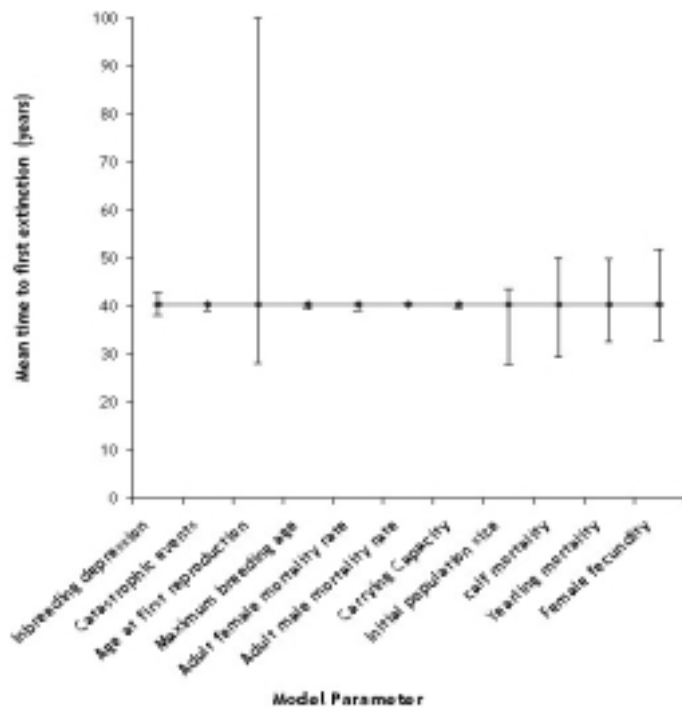# Caribou viability analysis applied to Woodland Caribou in Jasper National Park

Figure 3. Demographic sensitivity analysis for the simulated (VORTEX) JNP Woodland Caribou population. The mean time to first extinction is shown against the different model parameters used in the simulation (Table 2).
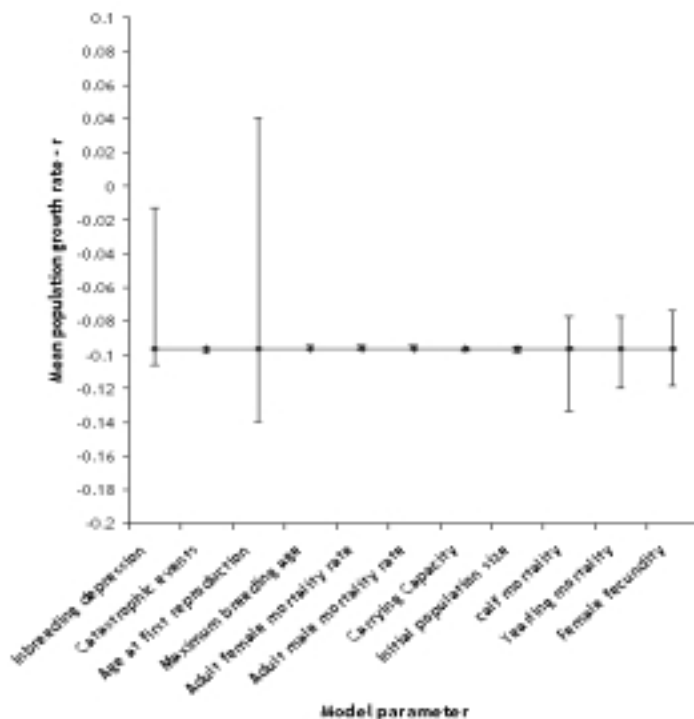

Figure 4. Demographic sensitivity analysis for the simulated (VORTEX) JNP Woodland Caribou population. The mean stochastic growth rate is shown against the different model parameters used in the simulation (Table 2).

REFERENCES CITED

Armbruster, P. & R. Lande. 1993. A population viability analysis for African Elephant (*Loxodonta africana*): How big should reserves be? Conservation Biology 3: 602-610.

Beissinger, S.R. & M.I. Westphal.1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management 62:821-841.

Berglind, S. 2000. Demography and management of relict sand lizard *Lacerta agilis* populations on the edge of extinction. Ecological Bulletins 48: 123-142.

Brown, K.W., J.L. Kansas & D.C. Thomas.1994. Greater Jasper Ecosystem Caribou Research Project Final Report. Sentar Consultants Ltd. Calgary, AB.

Dennis, B., P.L. Munholland & J.M. Scott.1991. Estimation of growth and extinction parameters for endangered species. Ecological Monographs 61:115-143.

Gilipin, M.E. & M.E. Soule. 1986. Minimum viable populations: processes of species extinction. In M.E. Soule (ed.). *Conservation Biology: The science of scarcity and diversity,* pp. 19-34. Sineauer Associates, Sunderland, MA.

Herrero, S., P.S. Miller, & U.S. Seal. 2000. Population and Habitat Viability Assessment for the Grizzly Bears of the Central Rockies Ecosystem (*Ursus arctos*). Eastern Slopes Grizzly Bear Project, University of Calgary, Calgary, Alberta, Canada and Conservation Breeding Specialist Group, Apple Valley, Minnesota, USA.

Lacy, R.C.1993. VORTEX: A computer simulation model for population viability analysis. Wildlife Research. 20:45-65.

Lacy, R.C. 2000. Structure of the VORTEX simulation model for population viability analysis. Ecological Bulletin 48: 39-51.

Leigh J.H. 1975. In Gilipin, M.E. & M.E. Soule.1986. Minimum viable populations: processes of species extinction. In M.E. Soule (ed.). *Conservation Biology: The science of scarcity and diversity,* pp.19-34. Sineauer Associates, Sunderland, MA.

Lennartsson, T. 2000. Management and population viability of the pasture plant *Gentianella campestris*: the role of interactions between habitat factors. Ecological Bulletins 48: 111-121.

MacArthur R.H & E.O Wilson. 2001 The Theory of Island Biogeography. Princeton University Press, Princeton, NJ. 224.pp.

Mercer, G. 2002. Personal Communication. Jasper National Park Wildlife Specialist.

Morris, W., D. Doak, M.Groom, P.Kareiva, J.Fieberg, L.Gerber, P. Murphy & D. Thomson. 1999. A practical handbook for population viability analysis. The Nature Conservancy.

Noon, B., and C.M. Biles. 1990. Mathematical Demography of Spotted Owls in the Pacific Northwest. Journal of Wildlife Management 54: 18-27.

Ralls, K. J.D. Ballou, & A.R. Templeton. 1988. Estimates of lethal equivalents and the cost of inbreeding in mammals. Conservation Biology 2:185-193.

Richer-Dyn, N. & N.S. Goel. 1972. In Gilipin, M.E. & M.E. Soule. 1986. Minimum viable populations: processes of species extinction. In M.E. Soule (ed.). *Conservation Biology: The science of scarcity and diversity,* pp. 19-34. Sineauer Associates, Sunderland, MA.

Ricklefs, R.E. 1997. *The economy of nature 4th ed. A textbook in basic ecology.* W.H. Freeman and Company. New York. USA.

Seip, D.R. 1992. Factors limiting woodland caribou populations and their interrelationships with wolves and moose in southeastern British Columbia. Canadian Journal of Zoology 70:1492-1503.

Shaffer, M.L. 1990. Population viability analysis. Conservation Biology 4:39-40.

Smith, K. 2002. Personal Communication. Alberta Sustainable Resource Development, Wildlife Biologist, Edson, AB.

Soulé, M.E (ed). 1987. *Viable Populations for Conservation.* Cambridge University Press, New York.

Stelfox, J.G. 1974. The abundance and distribution of Caribou and Elk in Jasper National Park, 1971-1973. Canadian Wildlife Service, Edmonton.

Taylor, M., & B. Plater. 2001. Population viability analysis for the southern resident population of Killer whale (*Orcinus orca*). The Center for Biological Diversity. Tucson, Arizona, USA. http://www.biologicaldiversity.org/swcbd/species/orca/pva.pdf

FISH, FOWL AND FOOD:
How interactions between small fish and invertebrates affect aquatic bird foraging behaviour in Alberta's aspen parkland

Caroline McParland and Cynthia Paszkowski

Aspen parkland and the aspen-dominated lower boreal forest provide breeding habitat for about 20% of all waterfowl surveyed annually in North America. The small fish commonly found in Alberta's aspen parkland and similar areas can alter aquatic invertebrate assemblage composition, and may reduce the invertebrate prey available to aquatic birds (Hanson & Riggs 1995).

Removal of small fish is associated with increased use of lakes by waterfowl in fall migration (Hanson & Butler 1994, Andersson & Nilsson 1999). Fish colonization reduces the abundance of aquatic insects and large cladocerans (Zimmer *et al.* 2001), but it is not clear whether fish change the availability of invertebrates important to breeding waterfowl.

Natural fish colonization can be artificially enhanced by land-use practices like wetland consolidation. In wetland consolidation, several small, shallow, temporary ponds are drained into one larger, deeper, permanent pond where fish survival is more likely. Consolidation may thus alter trophic relationships in aspen parkland wetlands and similar bird habitats. Protected habitats that share watersheds with surrounding developed land, where wetland consolidation has already occurred, are prone to increased fish colonization. One such example is Elk Island National Park (EINP), a remnant of aspen-dominated lower boreal forest in east-central Alberta.

This paper addresses two questions to determine whether fish presence or colonization affects aquatic invertebrates important to aquatic birds. First, is the presence of small fish associated with a reduction in invertebrates that are important to breeding birds? If so, then we hypothesize that the birds will have to invest more time foraging for those invertebrates in ponds with fish than in fishless ponds. Second, does colonization of a waterbody by small fish result in sudden reduction of important prey for aquatic birds? If so, then we hypothesize that birds will spend much more time foraging than they did before the colonization.

METHODS

We studied two aquatic bird species: the diving, piscivorous red-necked grebe (*Podiceps grisegena*) and the dabbling, non-piscivorous blue-winged teal (*Anas discors*). Study ponds were in Elk Island National Park (EINP) and the adjacent Blackfoot Provincial Recreation Area (BPRA). Both areas share a watershed with surrounding agricultural and rural residential land where wetland consolidation is likely. In 2000 and 2001, we used instantaneous time sampling to determine the mean proportion of time that
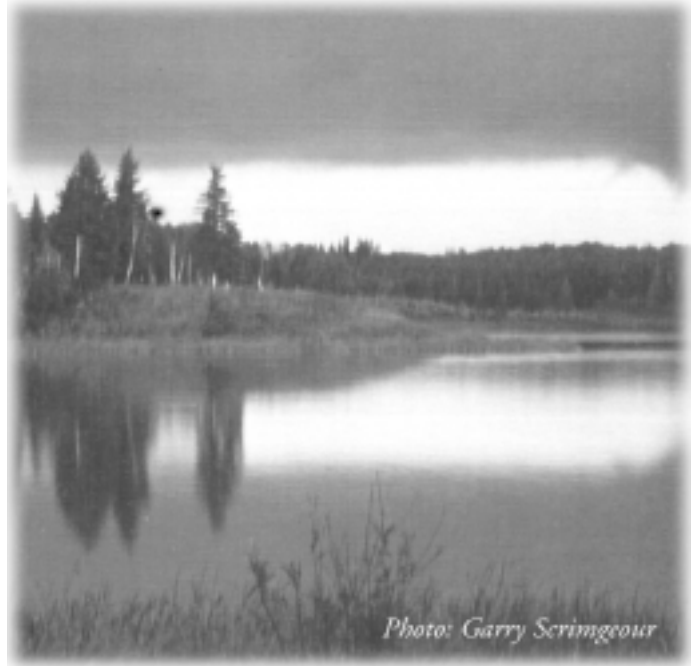

Photo: Garry Scrimgeour

Astotin Lake, EINP

breeding birds spent foraging on each of 10 fishless ponds and 8 ponds with fish. Monitoring was repeated six times during the summer; invertebrate sampling was synchronized with behaviour monitoring.

In 2001, we mimicked fish colonization by adding 4000 fathead minnows (*Pimephales promelas*) and brook sticklebacks (*Culaea inconstans*) to two ponds in BPRA that were fishless in 2000. Bird behaviour and invertebrates were monitored immediately after colonization. Follow-up sampling of these ponds is currently in progress to determine: (1) whether the introduced fish survived a long, dry winter, and, (2) whether their survival or extirpation resulted in any further changes to the invertebrates and to the birds' foraging behaviour.

PRELIMINARY RESULTS AND DISCUSSION

Invertebrate assemblages in ponds with/without fish

Of 22 invertebrate taxa examined, three were significantly less numerous in fish ponds than fishless ponds: amphipods (repeated measures ANOVAs at $\alpha = 0.10$: $F_{1,3} = 10.49$, $p = 0.048$), planorbid snails ($F_{1,3} = 7.248$, $p = 0.041$), and the leech, *Glossiphonia complanata* ($F_{1,3} = 13.545$, $p = 0.035$). Amphipods and planorbids

are important invertebrate prey of aquatic birds like blue-winged teal (Taylor 1978, Austin *et al.* 1998). Planorbid biomass was unaffected by fish presence ($F_{1,3} = 0.097$, $p = 0.776$), but amphipod biomass was significantly greater in ponds with fish present ($F_{1,3} = 8.338$, $p = 0.063$). This implies that fish presence was associated with a decrease in larger individual amphipods and planorbids. Thus, the forage base would contain fewer, larger individuals when small fish are present than when they are absent.

The effects of fish presence on the entire invertebrate assemblage in each sampling period were analyzed using Correspondence Analysis (CA). There was no overall difference in the taxonomic composition of invertebrate assemblages on ponds with and without fish in any sampling period (e.g., Figure 1). The small-bodied fish in this study cannot consume all invertebrate taxa, since some invertebrates are too large. The fish may compete with some of these larger macroinvertebrates, but they may only reduce the numbers and biomass of these invertebrates if they are at very high density (Hanson & Riggs 1995).

Bird behaviour in ponds with and without fish

Neither blue-winged teals nor red-necked grebes spent significantly greater time foraging on ponds with fish than on ponds without fish (Friedman tests: for teals $c^2 = 8$, $p = 0.156$; for grebes, $c^2 = 6$, $p = 0.306$). For red-necked grebes, which use only one pond throughout summer, these results suggest that fish and invertebrates were equivalent prey. For blue-winged teals, the lack of statistical significance was potentially due to low sample size (n = 8). Teals tended to spend more time foraging on ponds with fish than on ponds without fish (e.g. for time period 6, in mid-August, teals spent a mean of 28.6% of observed time foraging on fishless ponds, and 44% on fish ponds).

Birds' response to fish "colonization"

The trend towards increased foraging time by teals on ponds with fish was supported by the results of the fish colonization experiment. Blue-winged teals spent significantly more time foraging on the two experimental ponds after the fish addition than they had in the pre-treatment, fishless state. Typically, teals spent 0-16% of observed time foraging before colonization, and 25-48% afterwards. Red-necked grebes also spent more time foraging after the fish colonization than before the colonization, usually 10% before colonization and 30% after (Wilcoxon tests: for teals, $Z = 2.201$, $p = 0.028$; for grebes, $Z = 1.992$, $p = 0.046$).

The abundance of some important aquatic bird prey was reduced in the presence of fish; thus, it appears that fish colonization or fish presence can increase the quantity of time invested by blue-winged teals in foraging behaviour. The drought that began in 2000 may be another factor in the teals' behaviour, since the drying of ponds may affect invertebrate availability.
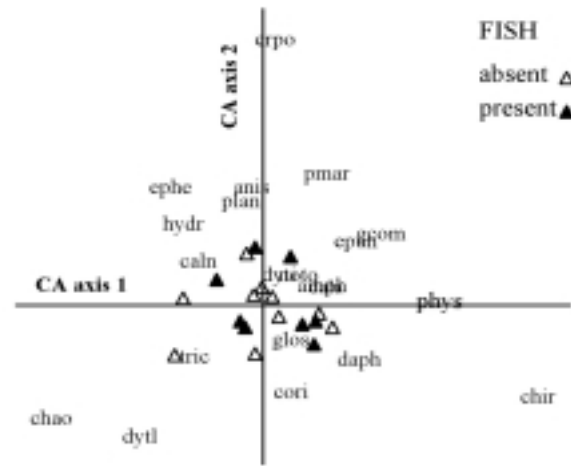

Figure 1. Results of Correspondence Analysis on the catch per unit effort data collected in time period 1 (early June) for invertebrate taxa on 10 fishless ponds and 8 fish ponds in 2000 and 2001. Hollow triangles indicate fishless ponds; black triangles indicate fish ponds. Abbreviations for the invertebrate taxa are given in Table 1 (on page 22). The clustering of fish and fishless ponds indicates that these sites have similar overall invertebrate taxonomic compositions, although individual taxa may be more strongly associated with particular sites. Invertebrate taxa that are close to a given site are more abundant at that site. Taxa far away from a given site are rare or absent from that site.

Grebes may have spent more time foraging for fish after colonization because the fish were added to the experimental ponds at lower densities than normally found in the area. Contrary to our earlier suggestion that fish and invertebrates were equivalent grebe prey, the fish may have been superior prey to the invertebrates that were the only prey available for breeding grebes before colonization. However, grebes breed successfully on fishless ponds in aspen parkland and elsewhere (C. McParland, *pers. obs.*, Stout & Nuechterlein 1999). There is currently no information to indicate that red-necked grebes differ in reproductive output on ponds with and without fish.

In conclusion, fish presence was associated with reductions in the abundance of some invertebrate taxa that are important prey of breeding aquatic birds. Non-piscivorous, blue-winged teals showed a trend toward spending more time foraging in ponds with fish than in fishless ponds, and spent markedly greater time foraging on ponds that fish artificially "colonized." Grebes showed no difference in foraging investment between ponds with and without fish, but increased their foraging investment when a low density of fish was added in the colonization experiment.

MANAGEMENT IMPLICATIONS

The presence of small fishes is associated with reductions in the abundance of important aquatic bird foods. Artificially enhanced fish colonization can require blue-winged teals and birds with similar requirements to spend more time foraging for invertebrate prey.

Fish occur naturally in the aspen-dominated lower boreal forest of EINP and BPRA. However, fish can colonize and survive in ponds that they would not normally reach through land-use practices like wetland consolidation; this colonization can potentially increase the proportion of ponds with poorer prey bases for aquatic birds. Currently, there is little or no information on the extent of wetland consolidation in the developed areas that share a watershed with the two parks. With recent declines of some of North America's waterfowl populations, managers seeking to maintain the ecological integrity of Elk Island and similar habitats must seriously consider: (1) documenting the precise extent of wetland consolidation around these important aquatic bird habitats, and, (2) discouraging the extensive practice of wetland consolidation in the surrounding developed areas.

ACKNOWLEDGEMENTS

Caroline E. McParland is currently a Ph.D. candidate at the University of Alberta. Tel: (780) 492-5751;
cem3@ualberta.ca.

Cynthia A. Paszkowski is Associate Professor/Research Chair of Biological Sciences at the University of Alberta.
Tel: (780) 492-1286; cindy.paszkowski@ualberta.ca.

Table 1. Invertebrate taxa examined and abbreviated in Figure 1. Taxa marked with * are important foods of Blue-winged teals (Taylor 1978), Taxa marked with ** are important prey of Red-necked grebes (Stout & Nuechterlein 1999)

| Invertebrate Taxa	Abbreviation on CA Graph
Amphipoda	amph*
Anisoptera	anis**
Calanoid copepods	caln
Chaoboridae	chao
Chironimidae	chir*
Corixidae	cori
Dytiscid larvae	dytl**
Dytiscidae adults (excl. *D. alaskanus*)	dyti
Dysticus alaskanus	dala**
Ephemeroptera	ephe
Erpobdella punctata	epun**
Erpobdellidae	erpo*
Glossiphonia complonata	gcom
Hydracnidia	hydr
Lymnacidae	lymn*
Notonecta sp.	noto
Percymoorensis marmorata	pmar**
Physidae	phys*
Planorbidae	plan*
Tricoptera	tric*
Zygoptera	zygo**

REFERENCES CITED

Austin, J.E., C.M. Custer, and A.D. Afton. 1998. Lesser Scaup. *The Birds of North America* **338**: 1-31.

Andersson, G., and L. Nilsson. 1999. Autumn waterfowl abundance in Lake Ringsjon, 1968-1996. *Hydrobiologia* **404**: 41-51.

Hanson, M.A., and M.G. Butler. 1994. Responses to food web manipulation in a shallow waterfowl lake. *Hydrobiologia* **279-280**: 457-466.

Hanson, M.A., and M.R. Riggs. 1995. Potential effects of fish predation on wetland invertebrates: a comparison of wetlands with and without fathead minnows. *Wetlands* **15**: 167-175.

Stout, B.E., and G.L. Neuchterlein. 1999. Red-necked Grebe. *The Birds of North America* **465**: 1-21.

Taylor, T.S. 1978. Spring foods of migrating blue-winged teals on seasonally flooded impoundments. *Journal of Wildlife Management* **42**: 900-903.

Zimmer, K.D., M.A. Hanson, and M.G. Butler. 2001. Effects of fathead minnow colonization and removal on a prairie wetland ecosystem. *Ecosystems* **4**: 346-357.

Movement of wolves in response to human development

Mace, R.D., J.S. Waller, T.L. Manley, L.J. Lyon, and H. Zuuring. 1996. Relationships among grizzly bears, roads and habitat in the Swan Mountains, Montana. *Journal of Applied Ecology* **33**: 1395-1404.

Miller, J.R., and N.T. Hobbs. 2000. Recreational trails, human activity, and nest predation in lowland riparian areas. *Landscape and Urban Planning* **50**: 227-236.

Miller, S.G., R.L. Knight, and C.K. Miller. 2001. Wildlife responses to pedestrians and dogs. *Wildlife Society Bulletin* **29**: 124-132.

Mladenoff, D.J., T.A. Sickley, R.G. Haight, and A.P. Wydeven. 1995. A regional landscape analysis and prediction of favorable gray wolf habitat in the Northern Great Lakes Region. *Conservation Biology* **9**: 279-294.

Mladenoff, D.J., T.A. Sickley, and A.P. Wydeven. 1999. Predicting gray wolf landscape recolonization: logistic regression models vs. new field data. *Ecological Applications* **9**: 37-44.

Noss, R.F., H.B. Quigley, M.G. Hornocker, T. Merrill, and P.C. Paquet. 1996. Conservation biology and carnivore conservation in the Rocky Mountains. *Conservation Biology* **10**: 949-963.

Papouchis, C.M., F.J. Singer, and W.B. Sloan. 2001. Responses of desert bighorn sheep to increased human recreation. *Journal of Wildlife Management* **65**: 573-582.

Rowland, M.M., M.J. Wisdom, B.K. Johnson, and J.G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management* **64**: 672-684.

Schultz, C.B., and E.E. Crone. 2001. Edge-mediated dispersal behavior in a prairie butterfly. *Ecology* **82**: 1879-1892.

Telfer, E.S., and J.P. Kelsall. 1984. Adaptation of some large North American mammals for survival in snow. *Ecology* **65**: 1828-1834.

Thurber, J.M., R.O. Peterson, T.R. Drummer, and S.A. Thomasma. 1994. Gray wolf response to refuge boundaries and roads in Alaska. *Wildlife Society Bulletin* **22**: 61-68.

Trombulak, S.C., and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* **14**: 18-30.

Turchin, P. 1998. *Quantitative analysis of movement: measuring and modeling population redistribution of animals and plants*. Sinauer Associates, Inc. Sunderland, Massachusetts, USA. 396pp.

Recently In Print
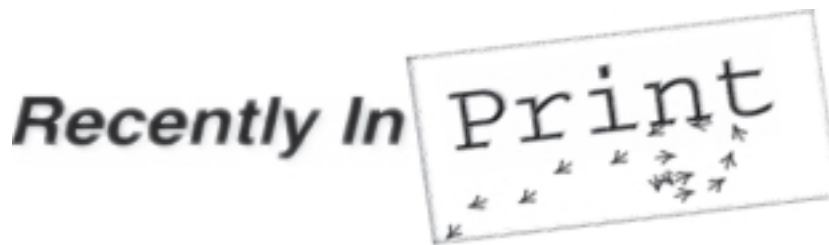
Sloan, N.A. 2002. History and Application of the Wilderness Concept in Marine Conservation. Conservation Biology 16(2): 294-305

Hansen, M.J., S.E. Franklin, C.G. Woudsma and M. Peterson. 2001. Caribou habitat mapping and fragmentation analysis using Landsat MSS, TM, and GIS data in the North Columbia Mountains, British Columbia, Canada. Remote Sensing of Environment 77: 50-65

Hansen, M.J., S.E. Franklin, C. Woudsma and M. Peterson. 2001. Forest Structure Classification in the North Columbia Mountains Using the Landsat TM Tasseled Cap Wetness Component. Canadian Journal of Remote Sensing 27(1): 20-32

Whittington, J. 2002. Movement of wolves (Canis lupus) in response to human development in Jasper National Park, Alberta. Msc Thesis. University of Alberta, Edmonton, Alberta.

Little, S.J., R.G. Harcourt and A.P. Clevenger. 2002. Do wildlife passages act as prey-traps? Biological Conservation 107: 135-145

Clevenger, A.P., J. Wierzchowski, B. Chruszcz and K. Gunson. 2002. GIS-Generated, Expert-Based Models for Identifying Wildlife Habitat Linkages and Planning Mitigation Passages. Conservation Biology 16(2): 503-514

Fitzsimmons, M. 2002. Estimates rates of deforestation in two boreal landscapes in central Saskatchewan, Canada. Canadian Journal of Forestry Research 32: 843-851

Rare Vascular Plants of Alberta

Kershaw, L., J. Gould, D. Johnson and J. Lancaster (eds.) 2001. Rare Vascular Plants of Alberta. University of Alberta Press and the Canadian Forest Service, Edmonton, Alberta 484 p.

ISBN 0-88864-319-5 (softcover)

"This publication describes approximately 485 vascular plants that are considered rare in Alberta. In a one-species-per-page format, this manual gives the name of the species, most common synonyms, description, habitat and notes on similar species. Species are illustrated with colour photographs and/or line drawings. For most species, maps of their distribution in North America and in Alberta are provided. The work is a result of intensive floristic surveys that included 30 direct contributors and about 100 amateur botanists.

Besides the species treatments, the book gives a good introduction to problems of rare plant studies, and a good description of the phytogeography of Alberta. Several appendicies provide interesting reading and an illustrated glossary. Close to 300 literature citations conclude this handy publication."

Meetings of Interest

January 17-18, 2003   Climate Change in the Columbia Basin. Prestige Inn, Cranbrook, BC. The workshop will address what climate change might mean for southeastern BC, and encourage an exchange of ideas among regional stakeholders in regard to adaptation to a changing climate. Major themes include: our changing climate from provincial and regional perspectives, impacts of climate change in the Columbia Basin, and "What is climate change adaptation planning?" For information or to register, contact: Columbia Mountains Institute of Applied Ecology, Tel: (250) 837-9311; fax: (250) 837-4223; cmi@revelstoke.net; www.cmiae.org/conferences.htm

February 5-6, 2003   "Helping Put Natural resource Knowledge to Work." Natural Resources Information Management Forum. Vancouver, BC. Forest Research Extension Partnership (FORREX) in collaboration with a variety of organizations is facilitating this 2-day forum that will address strategies, tools an technologies for collecting, organizing, sharing and applying social, economic and environmental information. This event is targeted at natural resource agencies and individuals that have an interest in management of scientific, biological or geospatial information. For more information in this forum contact Trina Innes: Tel: (250) 371-3955; fax: (250) 371-3997, or check the website: www.forrex.org/imforum

March 18-19, 2003   Access Management: policy to practice. Presented by the Alberta Society of Professional Biologists (ASPB). Telus Convention Centre, Calgary, AB. This conference will explore the ongoing debate surrounding access management and the roles of government, industry, agriculture, aboriginal communities, conservation groups and the general public as cooperating and competing forces to advance approaches to ecological and natural resource sustainability. Proposed sessions will address issues including: . Regulatory Framework – From policy development at the federal level to regional planning boards; Legal issues; The role of Aboriginal communities in decision-making; and Science and the public in managing resources. For information contact: ASPB Conference 2003 (Edmonton) Tel: 1-800-711-5765 or (780) 434-5765; fax: (780) 413-0076; pbiol@aspb.ab.ca; www.aspb.ab.ca

March 28-30, 2003   Ecosystem at Risk: Antelope Brush Restoration. Sponsored by the BC Chapter of the Society for Ecological Restoration, Osoyoos Desert Society, Canadian Wildlife Service and The Nature Trust. Osoyoos, BC. The conference will focus on the Antelope Brush ecosystem and related ecosystems restoration work and research. The deadline for abstract submissions is December 31, 2002. Completed papers must be received prior to February 15, 2003. Contact: Robert Seaton, Papers Chair, Brinkman and Associates Reforestation, 521 Sharpe Street, New Westminster, BC. robert.seaton@attcanada.net for abstract guidelines.

April 14-18, 2003   Protecting Our Diverse Heritage,: The Role of Parks, Protected Areas and Cultural Sites. San Diego, CA. This is a joint conference of the George Wright Society (GWS) and the National Park Service (NPS). It is the second in a series of nationwide forums, convened by the National Park Service, for discussing how to increase awareness of the value of cultural resources, for strengthening communications among cultural resources staff and NPS partners, and for discussing best practices and recent developments in cultural resources management. For more information check the GWS website: www.georgewright.org/2003overview.html

Parks Canada   Parcs Canada

Canada