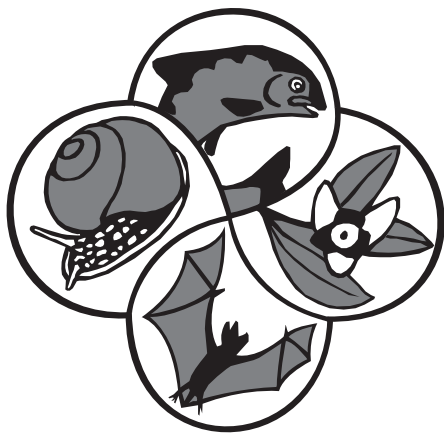


2009  
Vol. 26 No. 1 & 2  
Pages 1 - 52



# *Endangered Species*

# UPDATE

*Science, Policy & Emerging Issues*

School of Natural  
Resources and  
Environment

THE UNIVERSITY  
OF MICHIGAN



# Endangered Species UPDATE

Science, Policy & Emerging Issues

## Contents

Shark Fin Identification through DNA Barcoding.....3  
Robert D. Ward

Species at Risk: Golden Eagle Predation on Arid-Land  
Foxes .....10  
Howard O. Clark, Jr.

The Karner Blue Butterfly, Behavior, and the Role of Fire in Man-  
aging a Reintroduced Population.....15  
Bradley A. Pickens

Recent Disappearance of the Benthic Amphipod, *Diporeia* spp., in  
the Great Lakes: Spatial Patterns, Potential Causes, and Ecological  
Implications.....21  
Thomas F. Nalepa

The Kirtland's Warbler: Interagency and International Coopera-  
tion works .....31  
David N. Ewert  
Joseph M. Wunderle, Jr.  
Christopher W. Eberly  
Philip W. Huber  
Eric Carey

Fire History and its Implications for an Endemic Fire-Dependent  
Herb (*Iliamna corei* Sherff.) of the Appalachian Mountains .....39  
Charles W. Lafon  
Jennifer A. Hoss  
Henri D. Grissino-Mayer  
Serena R. Aldritch  
Georgina G. Deweese

Lion predation on the African Black Rhinoceros and Its Potential  
Effect on Management .....43  
Felix Patton

Focus on Nature: Louisian Black Bear (*Ursos americanus luteolus*) ...  
.....50  
Rochelle Mason

*Endangered Species Bulletin*, September 2007.....Insert  
Provided by the U.S. Fish and Wildlife Service

2009 Vol. 26 No. 1 & 2

Megan Banka ..... Editor  
David Faulkner..... Associate Editor  
Devan Rouse ..... Associate Editor  
Johannes Foufopoulos ..... Faculty Advisor  
Bobbi Low ..... Faculty Advisor

### Advisory Board

*Richard Block*  
Santa Barbara  
Zoological Gardens

*Susan Haig, PhD*  
Forest and Rangeland Ecosystem  
Science Center, USGS  
Oregon State University

*Patrick O'Brien, PhD*  
ChevronTexaco Energy Research Company

*Jack Ward Thomas, PhD*  
University of Montana

Subscription Information: The Endangered Species UPDATE is published four times per year by the School of Natural Resources and Environment at The University of Michigan. Annual rates are: \$78 institution, \$33 individual, \$25 student/senior, and \$20 electronic. Add \$5 for postage outside the US, and send check or money order (payable to The University of Michigan, Endangered Species UPDATE) to:

Endangered Species UPDATE  
School of Natural Resources and Environment  
The University of Michigan  
440 Church Street  
Ann Arbor, MI 48109-1041  
(734) 763-3243; fax (734) 936-2195  
E-mail: [esupdate@umich.edu](mailto:esupdate@umich.edu)  
<http://www.umich.edu/~esupdate>

Cover: A Kirtland's Warbler. Photo courtesy of XXX

The views expressed in the Endangered Species UPDATE may not necessarily reflect those of the U.S. Fish & Wildlife Service or The University of Michigan.

The Endangered Species UPDATE was made possible in part by Chevron Corporation and the U.S. Fish and Wildlife Service Division of Endangered Species.



# Shark Fin Identification through DNA Barcoding



**Robert D. Ward**

Wealth from Oceans Flagship,  
CSIRO Marine and Atmospheric Research, Hobart, Tasmania 7001, Australia

## **Abstract**

Populations of many shark species are in decline, partly due to their capture for the shark fin trade. Identification to species level of shark fins is often difficult or impossible using morphological traits, but genetic markers can facilitate unambiguous delineation. DNA barcoding – the sequencing of a 650 bp region of the cytochrome oxidase I gene of the mitochondrial DNA molecule – enables the ready separation of about 97% of all shark and ray species. Here the process of DNA barcoding is briefly outlined, and the approach applied to the assessment of a catch of illegally taken shark fins. Eighteen of these fins failed to yield a chondrichthyan sequence, but the remaining 193 fins could be identified as comprising 27 different species of sharks and rays. DNA barcoding will be increasingly recognised as an important tool for forensic identification of animals and animal products.

## **Introduction**

Escalating demand for shark fins is leading to the deaths of increasingly large numbers of sharks. Often, this finning is done on board the fishing boat, and the shark trunks discarded over-board. Trunks are discarded as they do not have the high value of the fins and occupy valuable ship space; in addition, cold-storage facilities may not be available so that the trunks might turn putrid in often warm and humid environments.

The size of the shark fin business is huge, but the illegal, unregulated or unreported (IUU) nature of much of the catch precludes precise recording of catch numbers. A recent study (Clarke et al. 2006) of the largest shark fin market, Hong Kong, gave a fishery-independent estimate of the global shark catch for fins which was substantially higher than that in the database of the Food and Agriculture Organisation (FAO). The fishery-independent estimate was some 1.7 million tonnes of fins per year (38 million individuals), compared with FAO figures in the range of 0.4 to 0.6 million tonnes per year (FAO 2005). The large disparity may be due to a combination of unrecorded landings, mis-recording of data in non-shark categories, and/or a high frequency of shark finning and carcass disposal at sea.

Populations of sharks and rays are more susceptible to fishing mortality than teleosts, as elasmobranchs generally mature late, have long life spans, and give birth to few (but well-developed) offspring. It is not surprising therefore that there have been reported declines in shark populations (Baum et al.

Freshly removed fins of thresher sharks being sorted in a processing area in Cilacap, Central Java (Indonesia). Photographer: Will White.



2003; Ward and Myers 2005), declines which are likely to be exacerbated by rapidly growing Asian economies leading to a burgeoning demand for shark fins. Initially only a few oceanic pelagic sharks were targeted, but growing demand has led to a wider range of target species. The global status of 21 oceanic pelagic shark and ray species was recently assessed (Dulvy et al. 2008), and 16 of these were classified as Threatened or Near Threatened using IUCN Red List criteria.

Species-level identification of harvested individuals is critical for the development and deployment of protocols for sustainable fisheries management. However, one of the issues faced when grappling with the extent and likely impact of shark catches is the relative paucity of good species-specific data. Only 15% of all shark catches reported to FAO were recorded by species name (Lack and Sant 2006). Morphologically similar species are often pooled in catch

data. Carcasses processed at sea can be difficult to identify at port if body parts with key identification features have been removed. Removed shark fins are especially difficult to attribute to species.

Molecular methods have long been used in identifying individual fish or parts of fish. DNA-based tools have largely replaced protein-based approaches for this, as DNA sequences can be obtained from minute tissue fragments, remain constant from tissue to tissue, and from egg to adult, permitting the identification of any life-history stage or any tissue fragment. There are several different DNA protocols that can be used, including restriction fragment length polymorphism analysis (e.g. Heist and Gold 1999), species-specific probes (e.g. Shivji et al. 2002), and DNA microarrays (e.g. Kochzius et al. 2008). Here we focus on a sequence-based approach, DNA barcoding, as a new and effective method for reliable

shark fin identification.

### **DNA barcoding**

DNA barcoding (Hebert et al. 2003) is really shorthand for species identification by DNA sequencing, the chosen barcode region for animals being a 650 bp region of the mitochondrial cytochrome oxidase I gene (COI). Where DNA barcoding differs from earlier uses of sequencing for species identification, is its advocacy of a single, uniform gene region. Previously, different studies or laboratories had used a variety of genes or gene regions, hindering the establishment of an effective sequence library for identification of a wide variety of species and taxa. This now exists for DNA barcoding as the web-based Barcode of Life Database (BOLD, [www.barcodinglife.org](http://www.barcodinglife.org), Ratnasingham and Hebert 2007). BOLD now allows some markers to be used in addition to COI, including the chloroplast markers *matK* and *rbcl* for plants (where COI performs poorly in species discrimination).

BOLD is more than just a database holding the DNA barcodes from reference material (together with associated specimen data, photographs and sequence trace files). It also includes analytic modules (such as clustering algorithms) allowing some preliminary examinations of sequence similarities, and an Identification Engine enabling an unknown sequence to be phenetically matched against the entire BOLD database.

DNA barcoding using COI is a powerful tool for fish species identification (Ward et al. 2005; Hubert et al. 2008; Steinke et al. 2009), and a global network for barcoding every fish species has been established (FISH-BOL, [www.fishbol.org](http://www.fishbol.org), Ward et al. 2009). Of the 31,000 or so fish species that have been described (Eschmeyer & Fong 2009), more than 7,300 (November 2009) have now been barcoded, from more than

50,000 specimens. It is worth pointing out that all developed and most developing countries would now have ready access to molecular laboratories that can do this work; alternatively tissue or DNA can be sent to external laboratories for sequencing.

Once the reference database has been established, DNA barcoding allows the identification of adults, juveniles, larvae and eggs, as well as tissue fragments (such as fin clips) and even some cooked product (Smith et al. 2008). DNA barcoding is primarily intended as an identification tool, but its potential uses go beyond that. Deep genetic divergences within the same nominal fish species may well signal unrecognised, cryptic species (Ward et al. 2008a; Zemlak et al. 2009), findings which will require taxonomic expertise to resolve. It is important in developing the database that wherever possible vouchers of barcoded specimens are retained in museums, as this will help resolve such issues, as well as inadvertent initial misidentifications. BOLD is a workbench comprising both public (mostly published) and private (unpublished) studies, and inevitably some of the latter projects contain a few misidentifications or mis-labelings awaiting correction.

### **DNA barcoding sharks and rays**

For DNA barcoding to function as an identification tool, each species has to have a unique barcode or a cluster of closely-related barcodes which are unambiguously distinguishable from those of all other species. The reference barcode database also has to be as complete as possible, as unknown specimens not represented in the species database will either not be able to be identified, or, conceivably, could be misidentified.

DNA barcoding is known to work well for sharks and rays. In a study of

Dried and packaged shark fins in Jakarta (Indonesia). Photographer: Simon Vieira.



210 chondrichthyan species, only two species (*Urolophus sufflavus* and *U. cruciatus*) could not be separated (Ward et al. 2008b). However, a few species were very similar to one another. For example, *Carcharhinus limbatus*, *C. tilstoni* and *C. amblyrhynchoides* can sometimes be hard to discriminate. In general, it appears that about 97% of shark and ray species can be separated by COI barcoding. At the time of writing, FISH-BOL included records from 511 elasmobranch species (46% of the 1114 then recognised shark and ray species) and from 24 holocephalans (52% of the 46 then recognised species). It is possi-

ble that some of the species that cannot be distinguished using COI barcodes might be separated by a more variable gene region, such as the control region of mitochondrial DNA: this needs further study.

### Identification of shark fins by DNA barcoding

The ability of DNA barcoding to identify species of shark fin was examined by Holmes et al. (2009), from a batch of fins confiscated from a vessel fishing illegally in Australia's northern waters. Only left pectoral fins were examined, to prevent the possibility of the same



ambiguities resulting from a single approach alone.

### Concluding Remarks

The establishment of a single DNA barcode database for animals, based on the use of a uniform gene region, will be extremely useful for the identification of whole or fragmentary parts of animals. Exactly how a sequence from an unknown specimen should be best matched against that database is still somewhat controversial, but no doubt suitable software will shortly be available. Perhaps this will be a combination of distance-based and character-based methods.

Here the focus has been on the use of DNA barcodes for identifying shark fins. But of course the approach can be, and is being, used to identify a broad range of species, including threatened species. Some recent examples include seafood market substitutions (Wong and Hanner 2008), commercially-traded bush meat (Eaton et al. 2009), turtles (Naro-Maciel et al. 2009) and birds (Kerr et al. 2007). It also is an effective species delineator in most invertebrate taxa, but is a separate topic!

### Acknowledgements

I would like to warmly thank Bronwyn Holmes (CSIRO Marine and Atmospheric Research, Hobart, Tasmania) and Dirk Steinke (Biodiversity Institute of Ontario, Guelph, Ontario) who worked with me on the shark fin identification study, and without whom that study would not have been completed.

### Literature cited

Baum, J. K., R. A. Myers, D. G. Kehler, et al. 2003. Collapse and conservation of shark populations in the northwest Atlantic. *Science* 299:389-392

Clarke, S. C., M. K. McAllister, E. J. Milner-Gulland, et al. 2006. Global estimates of shark catches using trade records from commercial markets. *Ecology Letters* 9:1115-1126.

DeSalle, R., M. G. Egan, & M. Siddall. 2005. The unholy trinity: taxonomy, species delimitation and DNA barcoding. *Philosophical Transactions of the Royal Society of London Series B-*

*Biological Sciences* 360:1905-1916

Dulvy, N. K., J. K. Baum, S. Clarke, et al. 2008. You can swim but you can't hide: the global status and conservation of oceanic pelagic sharks and rays. *Aquatic Conservation: Marine and Freshwater Ecosystems* 18:459-482.

Eaton, M. J., G. L. Meyers, S.-O. Kolokotronis, et al. 2009. Barcoding bushmeat: molecular identification of Central African and South American harvested vertebrates. *Conservation Genetics* (published online 01 September 2009)

W. N. Eschmeyer and J. D. Fong 2009. Species of Fishes by Family/Subfamily. On-line version dated 9 Sep 2009. <http://research.calacademy.org/research/ichthyology/catalog/SpeciesByFamily.asp>

FAO 2005 FISHSTAT Plus (v. 2.30) Capture Production Database 1970-2003. Available at: <http://www.fao.org/fi/statist/FISOF/FISH-PLUS.asp>

Hebert, P. D. N., A. Cywinska, S. L. Ball, et al. 2003. Biological identifications through DNA barcodes. *Proceedings of the Royal Society of London Series B-Biological Sciences* 270:313-321.

Heist, E. J. & J. R. Gold. 1999. Genetic identification of sharks in the U.S. Atlantic large coastal shark fishery. *Fishery Bulletin* 97:53-61.

Holmes, B. H., D. Steinke & R. D. Ward. 2009. Identification of shark and ray fins using DNA barcoding. *Fisheries Research* 95:280-288.

Hubert, N., R. Hanner, E. Holm, et al. 2008. Identifying Canadian freshwater fishes through DNA barcodes. *PLoS One* 3:e2490.

Kerr, K. C. R., M. Y. Stoeckle, C. J. Dove, et al. 2007. Comprehensive DNA barcode coverage of North American birds. *Molecular Ecology Notes* 7:535-543.

Kochzius, M., M. Nölte, H. Weber, et al. 2008. DNA microarrays for identifying fishes. *Marine Biotechnology* 10:207-217.

Lack, M. & G. Sant. 2006. *Confronting Shark Conservation* Head On. TRAFFIC International: Cambridge, UK.

Lowenstein, J. H., G. Amato, & S.-O. Kolokotronis. 2009. The real *maccoyii*: identifying tuna sushi with DNA barcodes – contrasting characteristic attributes and genetic distances. *PLoS One* 4:e7866

Naro-Maciel, E, B. Reid, N. N. Fitzsimmons, et al. 2009. DNA barcodes for globally threatened marine turtles: a registry approach to documenting biodiversity. *Molecular Ecology Resources* (published online November 2009)

Ratnasingham, S. & P. D. N. Hebert. 2007. BOLD: The Barcode of Life Data System ([www.barcodinglife.org](http://www.barcodinglife.org)). *Molecular Ecology Notes* 7:355-364.

Shivji, M. S., S. C. Clarke, M. Pank, et al. 2002. Genetic identification of pelagic shark body parts for conservation and trade monitoring.



- Conservation Biology 16:1036-1047.
- Smith, P. J., S. M. McVeagh, & D. Steinke . 2008. DNA barcoding for the identification of smoked fish products. *Journal of Fish Biology* 72:464-471.
- Steinke D., T. S. Zemlak, J. A. Boutillier, et al. 2009. DNA barcoding of Pacific Canada's fishes. *Marine Biology* 156:2641-2647.
- Ward, P. & R. A. Myers. 2005. Shifts in open-ocean fish communities coinciding with the commencement of commercial fishing. *Ecology* 86:835-847
- Ward, R. D., T. S. Zemlak, B. H. Innes, et al. 2005. DNA barcoding Australia's fish species. *Philosophical Transactions of the Royal Society of London Series B- Biological Sciences* 360:1847-1851.
- Ward, R. D., R. Hanner, & P. D. N. Hebert. 2009. The campaign to DNA barcode all fishes, FISH-BOL. *Journal of Fish Biology* 74:329-356.
- Ward, R. D., F. O. Costa, B. H. Holmes, et al. 2008a. DNA barcoding of shared fish species from the North Atlantic and Australasia: minimal divergence for most taxa but *Zeus faber* and *Lepidopus caudatus* each probably constitute two species. *Aquatic Biology* 3:71-78.
- Ward, R. D., B. H. Holmes, W. T. White, et al. 2008b. DNA barcoding Australasian chondrichthyans: results and possible uses in conservation. *Marine and Freshwater Research* 59:57-71.
- Waugh, J., L. Huynen, C. Millar, et al. 2007. DNA barcoding of animal species – response to DeSalle. *Bioessays* 30:92-93.
- Wong, E. H-K. & R. H. Hanner. 2008. DNA barcoding detects market substitution in North American seafood. *Food Research International* 41:828-837.
- Wong, E. H-K., M. S. Shivji, & R. H. Hanner. 2009. Identifying sharks with DNA barcodes: assessing the utility of a nucleotide diagnostic approach. *Molecular Ecology Resources* 9 (suppl. 1):243-256.
- Zemlak, T.S., R. D. Ward, A. D. Connell, et al. 2009. DNA barcoding reveals overlooked marine fishes. *Molecular Ecology Resources* 9 (suppl. 1):237-242.



## Species at Risk: Golden Eagle Predation on Arid-Land Foxes

### Howard O. Clark, Jr.

H.T. Harvey & Associates,  
7815 North Palm Avenue,  
Suite 310, Fresno, CA 93711-  
5511

hclark@harveyecology.com

The Golden Eagle (*Aquila chrysaetos*) in California typically inhabits grassland foothills with scattered oaks (*Quercus* sp.), sycamores (*Platanus racemosa*), or large gray pines (*Pinus sabiniana*), with sustainable California ground squirrel (*Spermophilus beecheyi*) and black-tailed jackrabbit (*Lepus californicus*) populations, either along the Coast Range or the Sierra Nevada (Grinnell and Miller 1944). The Golden Eagle is federally protected under the Bald Eagle Protection Act of 1962 (U.S. Fish and Wildlife Service 1978). Nearly 70% of direct and indirect Golden Eagle deaths are from accidental trauma, such as collisions with vehicles, power and transmission lines, and other structures, such as wind turbines (Kochert et al. 2002). Poisoning, electrocution, and illegal shooting also take their toll on Golden Eagles. Loss and destruction of habitat due to wildfires and human developments have caused loss of shrublands that support jackrabbits and have led to a decline in Golden Eagle populations (Kochert et al. 2002).

Also associated with these rolling grassland habitats are several carnivore species, including the federally endangered San Joaquin kit fox (*Vulpes macrotis mutica* – U. S. Fish and Wildlife Service 1998). In the literature, there are reports of predation of fox species by Golden Eagles, including Channel Island gray foxes (*Urocyon littoralis*) and the swift fox (*Vulpes velox*) of the Great Plains. If Golden Eagles are known to prey on these species, it is also likely that they prey upon the San Joaquin kit fox in California. Herein I review the literature regarding predation of arid-land foxes by Golden Eagles and its applicability to eagle-fox interactions within the Central Valley of California.

Golden Eagles are strong birds and have been known to kill ungulates, including mountain sheep (*Ovis Canadensis* – Bleich et al. 2004). One bird can carry up to 3.5 kg in flight (Kochert et al. 2002), and can easily take a coyote pup (Ingles 1965). Golden Eagles have also been known to take domestic calves (Phillips et al. 1996). Kit foxes, which weigh 2.3 kg on average, do not seem to be a challenge for this powerful raptor. However, the literature is sparse in documenting kit foxes and other arid-land fox species as prey items, either because it has not been a common research objective, or these interactions are infrequent, and when observed, are not being reported.

Golden Eagles are typically diurnal whereas kit foxes are nocturnal, providing an unexpected predator-prey interaction. In the northern range of the San Joaquin kit fox (Alameda, San Joaquin, and Contra Costa counties), a common prey item for the fox is the California ground squirrel which is a diurnal species (Orloff et al. 1986). The lack of well-distributed and abundant kanga-



roo rat (*Dipodomys* sp.) populations has likely led the kit fox to prey-shift to the ground squirrel, which makes the fox vulnerable to diurnal aerial predators (Clark et al. 2007).

The northern range of the kit fox also occurs in ideal Golden Eagle habitat. The rolling oak savannah habitat of the northern range has been reported to support the highest density of Golden Eagles in the world (Franklin et al. 1998, Hunt et al. 1998). The additional predatory pressure of Golden Eagles on kit foxes in the northern range may contribute to the currently declining San Joaquin kit fox populations in that portion of California (Clark et al. 2007), although this is conjectural. In the past, Grinnell et al. (1937) documented Golden Eagle predation on San Joaquin kit foxes in Fresno County. Arthur Oliver, a trapper interviewed by Grinnell, stated that trapped foxes were sometimes destroyed by Golden Eagles, which also

occasionally took adult free-roaming foxes.

In the 1980s, portions of the northern kit fox range were targeted for an extensive rodent poisoning campaign, which left large expanses of grassland devoid of the California ground squirrel for many years (Orloff et al. 1986). The loss of this prey item during this time period, as well as the elimination of most of the highest quality habitat for the kit fox through agricultural conversion and urbanization, led to a disruption in the demographics of the species, making a full recovery of the species in the northern range a challenge. Golden Eagles, too, depend on California ground squirrels for food, and a decline in the squirrel population would have led to prey-switching in Golden Eagles as well. It is unknown if kit foxes were frequently targeted as a prey item during this poisoning campaign.

The Channel Island gray fox has a

San Joaquin kit fox (*Vulpes macrotis mutica*). Photo by Brian Cypher, western Kern County.

similar weight range as the kit fox, and it too has fallen victim to Golden Eagle predation (Collins et al. 2009). Historically, Bald Eagles (*Haliaeetus leucocephalus*) were the primary raptor on the Channel Islands, but were reduced to a dozen or so breeding pairs by the 1950s due to DDT and thinning egg shells, and were eventually extirpated from the islands. The open niche led to the occupation of the Channel Islands by Golden Eagles, which initially fed on the non-native pig (*Sus scrofa*) which had been introduced to the islands in the 1850s; however, they quickly learned that island foxes, having no natural enemies, were easier to catch than the pig. Without the pigs, the Golden Eagles likely would not have stayed long enough to discover the easily caught island fox (Roemer et al. 2001, 2002; Coonan et al. 2005). This example illustrates that at least under certain circumstances, Golden Eagles can regulate and deplete a population of a small fox species.

Foxes of the Great Plains of the United States, such as the swift fox, have also been preyed upon by the Golden Eagle. Pfeifer and Hibbard (1970) report that Golden Eagles had eaten the rear portion of a male swift fox carcass in North Dakota; this indicates that Golden Eagles scavenge swift foxes. Another male swift fox in South Dakota was partially consumed by a Golden Eagle while still in a trap (van Ballenberghe 1975). At times, eagles have been suspected of taking swift foxes in Kansas (Chambers 1978), Wyoming (Madson 1987), and Canada (Herrero 1985), but direct evidence was not always available.

During other studies, direct predation of swift foxes has been confirmed by examining physical evidence left behind. In Canada, Herrero et al. (1991) lost two study animals due to Golden Eagles. Moehrenschrager et al. (2007) lost 13 Canadian study animals between 1995 and 1997. Their study also included kit foxes in Mexico, and one

eagle was observed on the ground attempting to reach a kit fox caught in a trap. Mollhagen et al. (1972) and Olen-dorff (1976) surmised that adult kit foxes were taken as prey by Golden Eagles in Texas and New Mexico because kit fox prey remains were found in eagle nests. In a Golden Eagle nest inventoried for prey remains in Mongolia, red fox (*Vulpes vulpes*) and corsac fox (*V. corsac*) remains were discovered (Ellis et al. 1999). The authors suggest that the take of 27 foxes by Golden Eagles is "not so much a preference for foxes, but rather that populations of more normal prey were depressed at this site" (Ellis et al. 1999).

Due to significant land conversion to agriculture and urban uses in the northern Central Valley of California, the remaining habitat suitable for San Joaquin kit fox occupation lies along the western edge of the Valley (U.S. Fish and Wildlife Service 1998); an ecotone between the Central Valley's agricultural matrix and upland annual grassland, which is also known to support Golden Eagles. Intraguild predation and behavioral interactions between arid-land foxes and Golden Eagles are rare in the literature, likely because observing these interactions are difficult to make when foxes occur in low densities on the landscape. Moreover, these associations were likely uncommon prior to human encroachment and development of the Central Valley for two reasons. (1) Golden Eagles tend to remain in upland areas, even where grasslands are available on flatter terrain. (2) Kit foxes, prior to human conversion and settlement of the Central Valley, occurred on the valley floor in robust and sustainable densities. Therefore, these two species likely did not interact much historically. Kit foxes are now restricted to high quality Golden Eagle habitat, especially in the northern range, and eagle-fox interactions are potentially greater than they were in the past. More research on

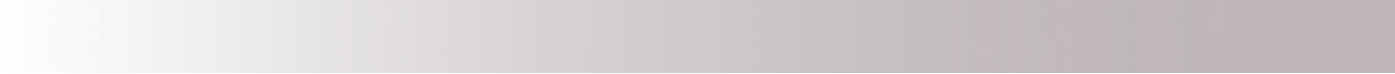
Golden Eagle predation on imperiled foxes, such as the kit fox, swift fox, and island gray fox, is needed to determine the impacts on fox population dynamics, predation avoidance behaviors, and other ecological variables, such as prey population cycles, especially when available habitat where both species currently co-occur is becoming increasingly rare in California.

### Acknowledgements

I thank Jeff Davis, Sue Hagen, and Dan Airola for their helpful comments.

### Literature Cited

- Bleich, V. C., E. F. Cassirer, V. L. Coggins, L. E. Oldenburg, and D. E. Hunter. 2004. Predation by a Golden Eagle, *Aquila chrysaetos*, on a juvenile mountain sheep, *Ovis Canadensis*. California Fish and Game 90:91-93.
- Clark, H.O., Jr., R.R. Duke, M.C. Orland, R.T. Gollightly, and S.I. Hagen. 2007. The San Joaquin kit fox in north-central California: a review. Transactions of the Western Section of the Wildlife Society 43:27-36.
- Chambers, G. D. 1978. Little fox on the prairie. Audubon 80:63-71.
- Collins, P. W., B. C. Latta, and G. W. Roemer. 2009. Does the order of invasive species removal matter? The case of the eagle and the pig. PLoS ONE 4:1-6.
- Coonan, T. J., C. A. Schwemm, G. W. Roemer, D. K. Garcelon, and L. Munson. 2005. Decline of an island fox subspecies to near extinction. The Southwestern Naturalist 50:32-41.
- Ellis, D. H., P. Tseneg, P. Whitlock, and M. H. Ellis. 1999. Predators as prey at a golden eagle *Aquila chrysaetos* eyrie in Mongolia. Ibis 141:139-142.
- Franklin, A. B., K. R. Wilson, and T. M. Shenk. 1998. Estimated annual rate of change in a Golden Eagle population at the Altamont Pass Wind Resource Area, California. Colorado State University, Fort Collins, Colorado.
- Grinnell, J., J. Dixon, and J. Linsdale. 1937. Kit foxes. Pages 399-420 in J. Grinnell, J. Dixon, and J. Linsdale, editors. Fur bearing animals of California. Volume 2, University of California Press, Berkeley.
- Grinnell, J., and A. H. Miller. 1944. The Distribution of the Birds of California. Pacific Coast Avifauna Number 27. Copper Ornithological Club, Berkeley, California. Reprinted by Artemisia Press, Lee Vining, California; April 1986. 617 pp.
- Herrero, S. 1985. The return of the swift fox to Canada. Pages 16-22 in: International Union of Directors of Zoological Gardens 40th Annual Conference, Calgary, Canada - Scientific Proceedings 1985. Royal Zoological Society of Scotland, Edinburgh.
- Herrero, S., C. Mamo, L. Carbyn, and M. Scott-Brown. 1991. Swift fox reintroduction into Canada. Pages 246-252 in: G. L. Holroyd, G. Burns, and H. C. Smith, editors. Proceedings of the Second Endangered Species and Prairie Conservation Workshop. Provincial Museum of Alberta, Natural History Section, Occasional Paper No. 15, Edmonton, Alberta.
- Hunt, W. G., R. E. Jackman, T. L. Hunt, D. E. Driscoll, and L. Culp. 1998. A population study of Golden Eagles in the Altamont Pass Wind Resource Area: population trend analysis 1994-1997. NREL/SR-500-26092. Predatory Bird Research Group, University of California, Santa Cruz.
- Kochert, M. N., K. Steenhof, C. L. McIntyre, and E. H. Craig. 2002. Golden Eagle (*Aquila chrysaetos*) in: A. Poole, and F. Gill, editors. The Birds of North America, No. 684. The Birds of North America, Inc., Philadelphia, Pennsylvania.
- Madson, C. 1987. Wyoming's wildlife: worth the watching. Wyoming Wildlife 51:34-35.
- Moehrenschrager, A., R. List, and D. W. Macdonald. 2007. Escaping intraguild predation: Mexican kit foxes survive while coyotes and golden eagles kill Canadian swift foxes. Journal of Mammalogy 88:1029-1039.
- Mollhagen, T. R., R. W. Wiley, and R. L. Packard. 1972. Prey remains in golden eagle nests: Texas and New Mexico. The Journal of Wildlife Management 36:784-792.
- Olendorff, R. R. 1976. The food habits of North American Golden Eagles. The American Midland Naturalist 95:231-236.
- Orloff, S., F. Hall and L. Spiegel. 1986. Distribution and habitat requirements of the San Joaquin kit fox in the northern extreme of their range. Transactions of the Western Section of the Wildlife Society 22:60-70.
- Pfeifer, W. K., and E. A. Hibbard. 1970. A recent record of the swift fox (*Vulpes velox*) in North Dakota. Journal of Mammalogy 51:835.
- Phillips, R. L., J. L. Cummings, G. Notah, C. Mullis. 1996. Golden Eagle predation on domestic calves. Wildlife Society Bulletin 24:468-470.
- Roemer, G. W., T. J. Coonan, D. K. Garcelon, J. Bascompte, and L. Laughrin. 2001. Feral pigs facilitate hyperpredation by golden eagles and indirectly cause the decline of the island fox. Animal Conservation 4:307-318.
- Roemer, G. W., C. J. Donlan, and F. Courchamp. 2002. Golden eagles, feral pigs, and insular carnivores: How exotic species turn native predators into prey. Proceedings of the National Academy of Sciences 99:791-796.

- 
- U. S. Fish and Wildlife Service. 1978. Bald Eagle Protection Act. 16 U.S.C. 668-668c. FWS/LE ENF 4. Washington, D. C.2 p.
- U. S. Fish and Wildlife Service. 1998. Recovery Plan for Upland Species of the San Joaquin Valley, California. U. S. Fish and Wildlife Service. Region 1. Portland, Oregon.
- van Ballenberghe, V. 1975. Recent records of the swift fox (*Vulpes velox*) in South Dakota. Journal of Mammalogy 56:525.

# The Karner Blue Butterfly, Behavior, and the Role of Fire in Managing a Reintroduced Population



## Bradley A. Pickens

School of Renewable Natural  
Resources, Louisiana State  
University AgCenter, Baton  
Rouge, LA 70803  
bpickens@lsu.edu

## Abstract

The Karner blue butterfly (Karner blue) [*Lycaeides melissa samuelis*] was extirpated from the oak savanna of Ohio, USA in 1989, and the butterfly was later reintroduced with little knowledge about how Karner blues respond to habitat management. While burning maintains the oak savanna habitat of the Karner blue, the consequences of burning on the butterfly remains unknown. The objective of our study was to determine the effect of prescribed burning on Karner blues. Each of the four study sites had host-plants of which 1/3 were burned, 1/3 were mowed, and 1/3 were left unmanaged each year. These treatments had been rotated within each site, to create a chronological order of succession (0-4 years since burning). We used surveys and behavioral observations to determine the effect of management on Karner blues. Our survey results showed Karner blues were not consistently selecting for management treatments. However, the oviposition rate in the unmanaged treatments was so low that only 5 of 127 eggs were oviposited in these treatments. Overall, the management strategy allowed for Karner blues to recolonize burned treatments and provided butterflies with the opportunity to make behavioral decisions that could potentially maximize their reproduction. In conclusion, we recommend burning oak savanna on a 3-4 year interval based on the Karner blue's lack of reproduction in savanna left unburned for  $\geq 4$  years.

## Introduction

The Karner blue butterfly (Karner blue) [*Lycaeides melissa samuelis*] is one of the first insects of conservation concern to be reintroduced in the United States (Figure 1). The Karner blue is an endangered species that inhabits early successional oak savanna and pine barrens ranging from New Hampshire to Wisconsin. The Karner blue is limited to an average lifespan of only 3.5 days (Knutson et al. 1999) and has a wingspan of about 2.5 cm. This small butterfly has two distinct broods per year with separate generations of adults flying in May and July. Much of the ecology of the Karner blue revolves around the fact that it is a larvae host-plant specialist that relies solely on wild blue lupine [*Lupinus perennis*], which lives in disturbed areas and nutrient poor soils associated with fire-prone ecosystems (U.S. Fish and Wildlife Service 2003).

The reasons for the butterfly's endangered status stem from massive land use changes associated with agriculture and extensive fire suppression throughout the 1900s. Fire is well known to be critical to maintain the oak savanna community (Davis et al. 2000; Peterson & Reich 2001), and many of the

Female Karner blue butterfly getting ready to oviposit on its host-plant, wild blue lupine.



rare, early successional species depend on fire to create proper habitat conditions. In particular, the Karner blue once inhabited the oak savanna of the Midwest, which has largely been converted to either agricultural lands or oak forests due to decades of fire suppression. In fact, only about 0.02% of historic oak savanna remains in the Midwest (Nuzzo 1986), and this ecosystem is home to many plants, insects, and birds of high conservation concern in the region (e.g. lark sparrow [*Chondestes grammacus*], red-headed woodpecker [*Melanerpes erythrocephalus*], frosted elfin [*Callophrys irus*], and hundreds of rare plant species).

In Ohio, the Karner blue was extirpated in 1989. The cause of extirpation remains unknown, but it was likely a result of dramatic habitat loss due to fire suppression, agriculture, and development. The resulting small popu-

lation was then subject to poor weather conditions (i.e. drought), and this probably eliminated the final few Karner blues in Ohio. After years of its absence from the region, a group of experts studied the possibility of reintroducing the species into northwest Ohio. Finally, in 1998, the Toledo Zoo, The Nature Conservancy, and the Michigan and Ohio Departments of Natural Resources decided to capture first brood Karner blues at Allegan State Game Area in Michigan (Figure 2), breed them in captivity at the Toledo Zoo, and then release the butterflies into the wild in Ohio during their second brood. The methodology included bringing butterflies into captivity, assuring proper breeding, and collecting oviposited eggs on host-plants within a greenhouse environment. One particular area was deemed most suitable for the Karner blue which was located at The Nature Conservancy's Kitty Todd Preserve in Lucas County, Ohio. At this preserve, The Nature Conservancy owns one of the largest pieces of oak savanna in Ohio. The Nature Conservancy had also instigated a prescribed burning program over the past several years prior to the butterfly's return. Additionally, the most well known factors affecting the butterfly, canopy cover and the presence of wild blue lupine, appeared most suitable at the Kitty Todd Preserve. The first reintroduction took place at one site within Kitty Todd Preserve in 1998. From 1998 to 2004, over 1500 Karner blues were released into the oak savanna.





**Figure 1:** Locations of where Karner blue butterflies were captured (Allegan SGA) and released into the wild (Kitty Todd Preserve).

Of course, prescribed burning of occupied Karner blue habitat was not the first priority after reintroducing the butterfly, since it might actually jeopardize the only Karner blue population in Ohio. There appeared to be a trade-off between the short-term negative effects of burning butterfly habitat (i.e. potentially killing eggs) and the long-term benefits of maintaining early successional oak savanna and the butterfly's host-plant. Therefore, prescribed burning was initially used sparingly. This management strategy was likely a result of the lack of available information on how Karner blue individuals would respond to prescribed burning and how a relatively small population might be affected. However, the oak savanna continued to degrade into late successional habitat without burning and the further loss of wild blue lupine was likely inevitable. In 2001, an active fire program was reinitiated in Karner blue occupied habitat (Figure 3).

For these reasons, Dr. Karen Root and I initiated a study at Bowling Green State University to investigate the con-

sequences of management on Karner blues in northwest Ohio. Our objective was to determine the effect of prescribed burning on Karner blues, and we used butterfly abundance surveys and oviposition rates as measures of habitat use. While methods such as performing surveys are difficult to interpret due to annual differences in abundance and small sample sizes, the behavior of an organism can rapidly provide insights into the perceived habitat quality of management treatments. The quantification of both butterfly abundance and reproductive rates within management treatments had the potential to determine the usefulness of these habitats.

### Methods

A thorough review of our methodology and results can be found in Pickens and Root (2009); a brief overview follows. Four distinct Kitty Todd sites had Karner blues, and each site had approximately 1/3 of the host plants burned, 1/3 mowed, and 1/3 left unmanaged each year. For the past several years, each treatment had been rotated within each site; this also represented a chronological order of succession with burned areas (0 years since burn) and mowed areas (1-2 years since burn) being early successional while the unmanaged areas had been left unburned for  $\geq 4$  years. This

A spring burn at one of our study sites at Kitty Todd Preserve, OH.



Field of wild blue lupine at the Kitty Todd Preserve, OH.



management strategy provided for four experimental sites to investigate and all sites included the three management treatments in the year of our study.

In 2005, one to three observers performed standardized butterfly survey transects within host-plant areas at Kitty Todd Preserve (Thomas 1983; Pollard & Yates 1993), and we recorded the number of Karner blues found within each management unit. Whenever we

found a female Karner blue, we observed the butterfly for a 15-minute interval to quantify foraging rates, and most importantly, where the Karner blues oviposited their eggs. Ovipositions were characterized by a Karner blue crawling down a lupine stem, flexing its abdomen, and depositing an egg. By observing these behaviors, we hoped to have a clear picture of both the habitat use and behavior of the species in rela-

tion to burning and mowing. We also quantified canopy cover and lupine area to account for these differences within management units.

## Results

Our survey results showed Karner blues were not consistently selecting for any of the treatments; essentially they were found randomly among treatments at each of the four sites. The only exception was the females in the second brood were more abundant in burned treatments compared to mowed treatments. Of course, these overall results actually meant the butterflies colonized the burned areas quite well because they were equally abundant in burned areas as in the other management treatments. The management treatments were within about 120 m of each other, therefore, the Karner blues did move substantially within each site. Incredibly, the oviposition rate in the unmanaged treatments was extremely low; only 5 of 127 eggs were oviposited in the unmanaged treatments. The pattern was statistically significant, and has important biological implications (Pickens & Root 2009). Meanwhile, there was no discernable pattern to foraging rates among management treatments.

## Discussion

For the reintroduction of species and the management of small populations, there are likely to be trade-offs of short-term and long-term goals. Prescribed burning in an area occupied by an endangered insect is generally thought to have negative consequences, such as direct mortality (Swengel 1995), yet the consequences of burning are not fully understood. The results of our study showed that behavior can be a good indicator of management effects, since butterflies avoided ovipositing in unmanaged treatments. Reproduction primarily occurred in burned and mowed treatments, and both of these treatments

represented the most recently burned savanna.

Initially, we knew that other butterfly species were selective of host-plants due to scent (Feeny et al. 1989), host-plant nitrogen (Myers 1985; Ellis 2003; Prudic et al. 2005), size of host-plants (Ellis 2003), and the presence of ants (Pierce & Elgar 1985; Fraser et al. 2002). However, these types of studies are primarily performed as lab-based experiments, and few studies have shown host-plant selection in the field. We now have evidence that Karner blues avoid ovipositing in oak savanna that is unburned for  $\geq 4$  years, and this is likely the result of successional processes in oak savanna. Of course, the initial recolonization of recently burned savanna is necessary to ensure butterflies have a chance to oviposit there. At Kitty Todd, the burned treatments were within about 120 m of unburned treatments. This is well within the dispersal distance of Karner blues (Knutson et al. 1999), and our survey results provided evidence that butterflies were able to quickly recolonize burned treatments. The overall management strategy allowed for a source of Karner blues to recolonize and provided butterflies with the opportunity to make behavioral decisions that could potentially maximize their reproduction. Importantly, the butterfly's lack of reproduction in units left unburned for  $\geq 4$  years probably makes these areas prime for burning the following winter.

The trade-off of managing for long-term habitat or short-term populations is not unique to insects and burning. For example, the wetlands of south Florida are home to the endangered snail kite [*Rostrhamus sociabilis*], where a perceived trade-off of providing potential short-term drought refugia by impounding water was given priority over the long-term benefits of water level variability in wetlands (Kitchens et al. 2002). Of course, the long-term dynam-

ics and responses of species to management are difficult to understand, but studying behavior can provide important insights for a variety of species of concern. This is particularly true for reintroduced species because adaptive management must be conducted as soon as possible to allow small populations to persist. In conclusion, I recommend prescribed burning Karner blue habitat at approximately 3-4 year intervals based on their avoidance of reproduction in areas left unburned for longer periods.

### Acknowledgements

The Ohio Department of Natural Resources, Division of Wildlife provided funding support for this project through a State Wildlife Grant; an additional grant was provided by the Ohio Biological Survey. We are thankful for the many people who were supportive of this project: G. Haase (The Nature Conservancy), P. Tolson and C. Ellsworth (Toledo Zoo) were especially helpful. We are grateful for the data collection efforts of E. Knurek, M. Ricci, and R. Kip. J. Bouzat, C. Pickens, H. Michaels, and K. Root of Bowling Green State University (BGSU) provided valuable insights throughout this project. Endangered species permits from the U.S. Fish and Wildlife Service were obtained through the Ohio Division of Wildlife. We are appreciative of the BGSU Geology Department and the BGSU Statistical Consulting Center for providing equipment and expertise.

### Literature Cited

- Davis, M. A., D. W. Peterson, P. B. Reich, M. Crozier, T. Query, E. Mitchell, J. Huntington and P. Bazakas. 2000. Restoring savanna using fire: Impact on the breeding bird community. *Restoration Ecology*, 8:30-40.
- Ellis, S. 2003. Habitat quality and management for the northern brown argus butterfly *Artaxerxes* (Lepidoptera: Lycaenidae) in North East England. *Biological Conservation*, 113:285-294.
- Feeny, P., E. Stadler, I. Ahman and M. Carter. 1989. Effects of plant odor on oviposition by the black swallowtail butterfly, *Papilio polyxenes* (Lepidoptera: Papilionidae). *Journal of Insect Behavior*, 2:803-827.
- Fraser, A. M., T. Tregenza, N. Wedell, M. A. Elgar and N. E. Pierce. 2002. Oviposition tests of ant preference in a myrmecophilous butterfly. *Journal of Evolutionary Biology*, 15:861-870.
- Kitchens, W. M., R. E. Bennetts and D. L. DeAngelis. 2002. Linkages between the snail kite population and wetland dynamics in a highly fragmented south Florida hydroscape. p. 183-202. In J. W. Porter and K. G. Porter (eds.), *The Everglades, Florida Bay, and coral reefs of the Florida Keys: an ecosystem sourcebook*. CRC Press, Boca Raton, FL.
- Knutson, R. L., J. R. Kwilosz and R. Grundel. 1999. Movement patterns and population characteristics of the Karner blue butterfly (*Lycaeides melissa samuelis*) at Indiana Dunes National Lakeshore. *Natural Areas Journal*, 19:109-120.
- Myers, J. H. 1985. Effect of physiological condition of the host plant on the ovipositional choice of the cabbage white butterfly, *Pieris rapae*. *The Journal of Animal Ecology*, 54:193-204.
- Nuzzo, V. A. 1986. Extent and status of midwest oak savanna: presettlement and 1985. *Natural Areas Journal*, 6:6-36.
- Peterson, D. W. and P. Reich. 2001. Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. *Ecological Applications*, 11:914-927.
- Pickens, B. A. and K. V. Root. 2009. Behavior as a tool for assessing a managed landscape: a case study of the Karner blue butterfly. *Landscape Ecology*, 24:243-251.
- Pierce, N. E. and M. A. Elgar. 1985. The influence of ants on host plant selection by *Jalmenus evagoras*, a myrmecophilous lycaenid butterfly. *Behavioral Ecology and Sociobiology*, 16:209-222.
- Pollard, E. and T. J. Yates. 1993. *Monitoring Butterflies for Ecology and Conservation* 1st edition. Chapman & Hall, London.
- Prudic, K., J. Oliver, M. Bowers and O. Schmitz. 2005. Soil nutrient effects on oviposition preference, larval performance, and chemical defense of a specialist insect herbivore. *Oecologia*, 143:578-587.
- Swengel, A. B. 1995. Observations of spring larvae of *Lycaeides melissa samuelis* (Lepidoptera: Lycaenidae) in central Wisconsin. *The Great Lakes Entomologist*, 28:155-170.
- Thomas, J. A. 1983. A quick method for estimating butterfly numbers during surveys. *Biological Conservation*, 27:195-211.
- U.S. Fish and Wildlife Service. 2003. *Karner Blue Butterfly Recovery Plan (Lycaeides melissa samuelis)*. Department of the Interior, U.S. Fish and Wildlife Service, Great Lakes- Big Rivers Region (Region 3).

# Recent Disappearance of the Benthic Amphipod, *Diporeia* spp., in the Great Lakes: Spatial Patterns, Potential Causes, and Ecological Implications



**Thomas F. Nalepa**

Great Lakes Environmental  
Research Laboratory, NOAA  
4840 S. State St.  
Ann Arbor, MI 48108  
Thomas.Nalepa@noaa.gov

## **Abstract**

Over the past several decades, the once abundant benthic amphipod *Diporeia* spp. has completely disappeared from areas < 90 m in water depth in all the Great Lakes except Lake Superior. This organism may presently still be found in deeper areas (> 90 m), but abundances even in these areas are in a state of decline. Declines were first observed in the early 1990s, just a few years after two invasive mussels, *Dreissena polymorpha* (zebra mussel) and *Dreissena rostriformis bugensis* (quagga mussel) became established. While the spatial and temporal decline of *Diporeia* coincided with the expansion of these two mussel species, the exact reason for the negative response of *Diporeia* to *Dreissena* has not been clearly defined. In the Great Lakes, the rate at which the population disappeared has varied from months to many years, and has occurred in areas remote from mussel-infested regions. The loss of *Diporeia* is having a far-reaching impact on the Great Lakes food web. As *Diporeia* is a benthic detritivore that was heavily fed upon by fish, this calorie-rich organism provided an efficient pathway by which energy produced and settled from the pelagic region (primary production) was cycled to upper trophic levels. With the loss of *Diporeia* and the recent expansion of *D. r. bugensis* into both nearshore and offshore regions, the benthic community has become an energy sink rather than a pathway. As a result, the abundance, growth, and condition of fish populations are declining.

## **Introduction**

The benthic amphipod *Diporeia* spp. is part of a relic faunal group that mostly inhabit deep, cold pro-glaciated lakes, brackish estuaries, and coastal margins in the Holarctic region (Bousfield 1989). In North America, it is found in deep, boreal lakes from the lower Mackenzie River in the west to Lake Champlain in the east where near-bottom summer temperatures do not exceed 14 °C (Dadswell 1974). More specifically in the Great Lakes, *Diporeia* (actually a species complex) was the dominant benthic organism in deeper regions (> 30 m) of all the lakes, comprising over 70 % of benthic biomass in these regions (Nalepa 1989). It was present but less dominant in open, shallow regions of the main lake basins (< 30 m), and naturally absent from shallow, warm bays and basins. *Diporeia* has a life span of up to 2-3 years, and reaches a maxi-

Image of the benthic amphipod *Diporeia* spp. Mean body length in Lake Michigan is about 10 mm.



mum size of about 10 mm (Fig. 1).

As a detritivore, it burrows in the top 1-2 cm of sediments and mostly feeds on organic material that settles from overlying waters. In particular, *Diporeia* feeds heavily on diatoms that settle during the spring bloom period. Diatoms are an energy-rich phytoplankton group, and ingestion rates, energy stores (lipids), and growth rates of *Diporeia* reached a seasonal peak during or just after the spring bloom (Gardner et al. 1985, 1990, Dermott and Corning 1988). In turn, *Diporeia* was fed upon by many fish species, including the commercially important lake whitefish, and forage fish such as alewife, bloater, and sculpin that serve as prey for larger piscivores (salmon, trout) (Scott and Crossman 1973, Wells 1980). Thus, *Diporeia* played a major role in the efficient movement of energy between lower (diatoms, phytoplankton) and upper (fish) trophic levels and was considered a keystone species in the Great Lakes ecosystem.

Over the past several decades, this once widespread, abundant organism has totally disappeared from large areas in all the Great Lakes except Lake Superior, and abundances of remaining populations are in the state of decline (Nalepa et al. 2006a, 2009). In this brief paper, I provide the background for current trends, consider potential causes for the decline, and summarize realized and potential ecological consequences.

### Historical Perspective

A sufficient number of historic surveys have been conducted in the Great Lakes to conclude that the current wide-scale disappearance of *Diporeia* is unprecedented. Previous studies of temporal trends over both the long (decades) and short term (years) show that populations do fluctuate relative to environmental conditions, but the total, systematic loss of entire populations over large areas is unique. Historically, long-term trends were mostly related

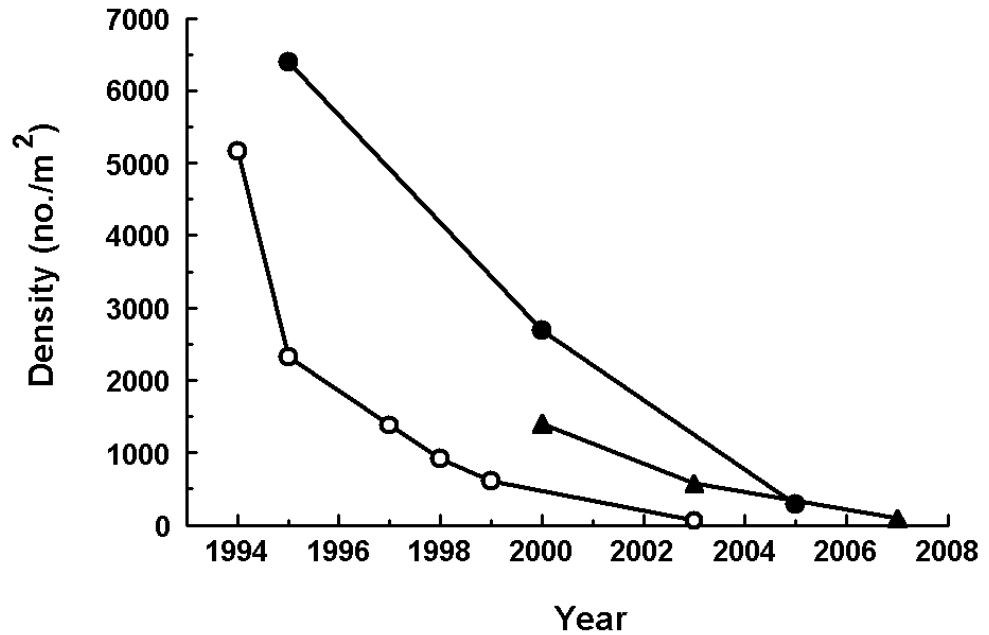
to changes in nutrient loads and pelagic productivity (Robertson and Alley 1966, Cook and Johnson 1974, Nalepa 1987). Nutrient loads (phosphorus) from both point and non-point sources increased up until the mid-1970s, leading to greater standing stocks of phytoplankton and hence greater amounts of organic material settling to the bottom. This material served as food for *Diporeia* and, as a result, *Diporeia* abundances increased. For example, in Lake Michigan at depths < 50 m, densities increased 2-fold between the 1930s and the mid-1960s (Robertson and Alley 1966), and 2 to 5-fold between the 1960s and the early 1980s (Nalepa 1987). After nutrient abatement programs in the mid-1970s, abundances declined consistent with diminished food availability (Nalepa et al. 1998). Shorter-term fluctuations have also been documented and mostly attributed to shifts in fish predators. In the Bay of Quinte, Lake Ontario, the *Diporeia* population in the late 1970s/early 1980s rapidly increased to levels more typical of those found in this portion of eastern Lake Ontario (Johnson and McNeil 1986). The population increase was related to the collapse of large populations of white perch which heavily fed on *Diporeia* and suppressed their numbers. In Lake Michigan, short-term changes in abundances were also attributed to population shifts in fish predators (McDonald et al. 1990). In each of these cases, it should be emphasized that *Diporeia* was still present, but abundances were simply lower than typically found; recovery occurred rapidly once predation pressure eased.

### Recent Declines

Beginning in the early 1990s, *Diporeia* populations began to decline and eventually disappear from large areas, including southeastern Lake Michigan, outer Saginaw Bay, eastern Lake Erie, and eastern Lake Ontario (Dermott and Kerec 1997, Dermott 2001, Lozano et al.

2001, Nalepa et al. 2003, 2006b). Initially, the decline was focused in near-shore areas (< 50 m), but over time progressed to deeper areas as well. In each lake area, the decline coincided with the introduction and spread of two invading bivalve mollusks, *Dreissena polymorpha* (zebra mussel) and *Dreissena rostriformis bugensis* (quagga mussel). The chronology of the *Diporeia* decline relative to the expansion of *Dreissena* was well-documented in Lake Michigan. Declines in the *Diporeia* population were first observed in the southern portion of the lake in the early 1990s, a few years after *D. polymorpha* colonized that region in 1989 (Nalepa et al. 1998). Abundances of *Diporeia* continued to decrease in the 1990s as *D. polymorpha* spread throughout the lake at depths < 50 m. By 2000, *Diporeia* was rare or completely extirpated from these shallower depth regions in the far southern and northern portions of the lake, and along the eastern shoreline (Nalepa et al. 2006b). The other dreissenid species, *D. r. bugensis*, became established in the lake in 1997, and proceeded to colonize deeper regions of the lake where *D. polymorpha* was never found (> 50 m), and to attain higher densities than *D. polymorpha* in shallow regions (< 50 m) (Nalepa et al. 2001, 2009). Consequently, the *Diporeia* population continued to decline throughout the 2000s, with declines most evident at depths > 50 m. Similar patterns of decline relative to *Dreissena* expansion were observed in Lakes Ontario and Huron. Based on the most recent lakewide surveys, the *Diporeia* population at 30-90 m declined 96 % between 1995 and 2005 in Lake Michigan, declined 99 % between 1994 and 2003 in Lake Ontario, and declined 93 % between 2000 and 2007 in Lake Huron (Fig. 2). In Lake Erie, the population began to decrease in deeper regions (> 20 m) of the central and eastern basins in 1992, and was completely gone from the lake by 1998 (Dermott

**Figure 1:** Temporal trends in density (no./ m<sup>2</sup>) of the amphipod *Diporeia* based on latest surveys in Lake Michigan (Nalepa et al. 2009) , Lake Ontario (Watkins et al. 2007), and Lake Huron (Nalepa et al. 2003, Nalepa unpublished data). Densities given as the mean value at a depth of 30-90 m in each lake. Lake Michigan = solid circle, Lake Ontario = open circle, Lake Huron = solid triangle (from Nalepa et al. 2009).



and Kerec 1997, MacDougall et al. 2001). Presently, *Diporeia* abundances in Lake Superior are stable, likely because *Dreissena* populations are relatively low and mainly confined to the far western portion of the lake (Scharold et al. 2004).

While the decline of *Diporeia* clearly coincided with the expansion of *Dreissena* in the Great Lakes, some aspects of its disappearance relative to *Dreissena* are inconsistent and not easily explained. For instance, at a 45-m site in southeastern Lake Michigan, *Diporeia* densities declined from 10,000/m<sup>2</sup> to near 0/m<sup>2</sup> in just 6 months in 1992 (Nalepa et al. 1998). This rapid rate of decline occurred despite the fact that *Dreissena*, although present at shallower depths, was not present at the site itself. In contrast, it took 16 years for *Diporeia* to disappear at a 20-m site in southwestern Lake Michigan, even though *Dreissena* was present at the site over the entire 16-year period (1992-2008).

*Diporeia* typically declines not only in areas where dreissenids are present, but also in areas that are far-removed from dreissenid populations (Dermott 2001, Nalepa et al. 2003, 2006b, Watkins et al. 2007).

#### Potential Causes for the Decline

A common hypothesis for the loss of *Diporeia* is that food availability has decreased because of the filtering activities of *Dreissena* (termed “food limitation” hypothesis). *Diporeia* feeds in the upper sediment layers, whereas *Dreissena* filter-feeds at the sediment surface. Thus, phytoplankton that settles to the bottom is likely intercepted and utilized by *Dreissena* before it actually reaches the upper sediments and becomes available to *Diporeia*. While some amphipod species that inhabit shallow bays and basins have increased in abundance since *Dreissena* became established, partly because they can feed on mussel biodeposits, in theory, *Diporeia*



does not feed on this material but is more dependent on freshly-settled phytoplankton (mostly diatoms). Because of water currents and other physical forces, the filtering impacts of *Dreissena* on phytoplankton extend beyond areas where populations are present, which would lead to subsequent declines in *Diporeia* over wide areas as observed. While plausible, there are several inconsistencies with the food-limitation hypothesis. As noted, past abundances were directly linked to pelagic productivity and amounts of food settling to the bottom. Under this scenario, present abundances should decline in direct proportion to declines in phytoplankton in areas where there are no dreissenids. This has not been the case. In eastern Lake Ontario, diatom biomass declined by 80% after *Dreissena* became established (Dermott 2001), but the *Diporeia* population, instead of declining by 80%, completely disappeared. Further, at the previously-mentioned site in southern Lake Michigan where *Diporeia* disappeared in 6 months, at least some potential food (diatoms) was still settling to the bottom as indicated by near-bottom sedimentation traps (Nalepa et al. 2006b).

If food limitation is the major cause of the decline in *Diporeia*, individuals should logically exhibit some physiological signs of starvation during the period of population loss. In *Diporeia*, lipid concentrations provide a good indicator of food availability. Levels increase after the spring diatom bloom when food inputs to the bottom are at a seasonal peak, and decline as food deprivation occurs during the summer stratification period when little food settles to the bottom (Gardner et al. 1985, 1990). Further, during food-deprivation experiments, lipid levels in *Diporeia* gradually declined over a period of several months as the animals utilized lipid stores to meet metabolic needs (Gauvin et al. 1989). Individual

lipid levels and weights were documented as the population declined in southeastern Lake Michigan in the late 1990s (Nalepa et al. 2006b). Levels initially declined, but then increased such that when the population eventually disappeared, levels were as high as, or higher, than levels found in the 1980s prior to dreissenid colonization. Moreover, individuals did not lose weight as typically found when amphipods are food deprived. If the food-limitation hypothesis is correct, then why lipids and weight did not decline as the *Diporeia* population disappeared remains unresolved. Efforts to link the decline to diseases, pathogens, and parasites have shown similar inconsistencies (Messick et al. 2004, Foley et al. 2006). Commonly, *Diporeia* disappears in areas far-removed from *Dreissena* colonies, particularly in areas with high rates of sedimentation (Nalepa et al. 2006b, Watkins et al. 2007). This suggests perhaps that a toxic substance directly or indirectly associated with *Dreissena* is being transported via currents and then deposited. In laboratory experiments, 100 % mortality was observed in *Diporeia* exposed to the cyanobacterium *Microcystis aeruginosa* (Kainz et al. 2010). This phytoplankton species produces the toxin microcystin, and blooms of this species and toxin concentrations have increased since the establishment of *Dreissena* (Vanderploeg et al. 2001, Dyble et al. 2008). Further, the incidence of type E botulism has increased in the Great Lakes since *Dreissena* became established (Perez-Fuentetaja et al. 2006). The bacterium that produces the toxin, *Clostridium botulinum* type E, thrives under anoxic conditions, such as those associated with decomposing mussel tissue or mussel biodeposits. Certainly, the susceptibility of *Diporeia* to these and other potentially toxic substances, diseases, and pathogens are in need of further study. Considering all the evidence, there may not be one fac-

tor causing the decline of *Diporeia*, but perhaps a multitude of factors, with a decline in food making organisms more susceptible to other environmental stresses (Nalepa 2006a, 2006b, Watkins et al. 2007).

Another enigma of the *Diporeia* decline relative to *Dreissena* is the coexistence of these two organisms in the Finger Lakes, New York. While *Dreissena* colonized these deep, summer-cold lakes in the mid -1990s and have attained densities comparable to those in the Great Lakes, *Diporeia* remains abundant, and has actually increased in Lakes Cayuga and Seneca (Dermott et al. 2006). A theory for their coexistence is that the two organisms have different food sources in the Finger Lakes (Dermott et al. 2006). Sediments of the Finger Lakes have an abundance of leaf material that is too coarse to be filtered by *Dreissena*. This material and associated bacteria could be available as food items for *Diporeia* since, depending upon circumstances, both detritus/bacteria and epilithic algae have been utilized as a nutritional resource (Guiguer and Barton 2002, Sierszen et al. 2006). Given the steep slopes of these lakes, the detrital material would be rapidly transported from shallow to deep, colder regions inhabited by *Diporeia*. Conceivably, the presence of at least some coarse, organic material that is nonfilterable by dreissenids may explain why *Diporeia* only gradually declined in some areas of the Great Lakes (i.e., the 20-m site in southwestern Lake Michigan).

### **Benthic Transformation of Lake Michigan: From *Diporeia* to *D. r. bugensis***

Regardless of the exact cause for the decline, recent studies have detailed how nutrient/energy flow has been disrupted in Lake Michigan as *Diporeia* decreased and *D. r. bugensis* increased (Nalepa et al. 2009, Fahnenstiel et al.

2010). Energy once efficiently cycled through *Diporeia* within the food web now resides in dreissenid mass (tissue and shell), and the benthic community now serves as an energy sink rather than a trophic pathway. In the 1980s when *Diporeia* was abundant and *Dreissena* was not yet present, mean densities of *Diporeia* at depth intervals of 16-30 m, 31-50 m, 51-90 m, and > 90 in the southern basin of Lake Michigan were 7, 171/m<sup>2</sup>, 10,677/m<sup>2</sup>, 6,459/m<sup>2</sup>, and 4,014/m<sup>2</sup> (Nalepa et al. 1998). In 2008, mean densities at the same depth intervals were 3/m<sup>2</sup>, 0/m<sup>2</sup>, 262/m<sup>2</sup>, and 796/m<sup>2</sup>, respectively (Nalepa et al. 2009).

Despite these declines, total mass of the benthic community actually increased because of the recent expansion of *D. r. bugensis*. The depth-weighted, mean biomass of *Diporeia* in the southern basin in the 1980s was 2.5 g/m<sup>2</sup>, whereas the mean depth-weighted biomass of *D. r. bugensis* in 2008 was 10.9 g/m<sup>2</sup>. Since nutrient loads and thus primary production within the lake have declined over the same period (Mida et al. 2010), how can *D. r. bugensis* achieve such high standing stocks compared to past populations of *Diporeia*? The evidence suggests that *D. r. bugensis* is physiologically more efficient than *Diporeia*, and thus able to sustain higher standing stocks at a given food level. *D. r. bugensis* has an assimilation efficiency (food assimilated relative to food ingested) that is 2 times greater than *Diporeia*, and a respiration rate that is 1/3 lower (Nalepa et al. 2009). Both of these attributes allow *D. r. bugensis* to allocate more energy to growth and reproduction, and less to metabolic maintenance. Also, *D. r. bugensis* can achieve higher standing stocks because of its feeding mode. Being an active filter feeder, *D. r. bugensis* has access to food resources within the entire water column during the unstratified period (October to May). The water column is well mixed at this time, and phytoplankton pro-

duced in upper, lighted waters is circulated to near bottom waters. On the other hand, *Diporeia* is a passive detritivore and has access to phytoplankton mostly in the spring when conditions favor material settling to the bottom.

The supposition that the benthic community has now become an energy sink is a consequence of not only the 4-fold increase in benthic standing stocks as *Diporeia* decreased and *D. r. bugensis* increased, but also a consequence of the great difference in nutritional value between the two organisms. *Diporeia* has a lipid content up to 30-40 % of its dry weight, making it rich in calories and a valued energy source. In contrast, *D. r. bugensis* has a lipid level consistently below 20 % and a relatively low energy content (Nalepa et al. 2010, McNickle et al. 2006). When feeding on a calorie-rich, readily available food item like *Diporeia*, fish maximize energetic benefits relative to expenditures. With the loss of *Diporeia*, benthic-feeding fish are switching to other food items, including *D. r. bugensis*, which do not offer the same energetic benefits. When a fish feeds on *D. r. bugensis* instead of *Diporeia*, it is acquiring 14 times less energy per wet weight mass (318 joules/g compared to 4,429 joules/g; given in McNickle et al. 2006). The relatively low energy density of *D. r. bugensis* is a result of its shell, which has no energy content but comprises up to 91 % of total mass. When ingested, the shell imposes energetic costs to the fish in terms of handling and eventual egestion, decreasing net feeding efficiency. Moreover, the shell imposes other ecological costs. The amount of energy used by *D. r. bugensis* to create the shell is unknown, but shell production comprised 37 % of total production (shell growth, tissue growth, reproduction) in *D. polymorpha* (Chase and Bailey 1999). Thus, energy is lost to the food web when the shell is produced by the mussel, and also lost when it is handled

and egested by fish.

### Impact on Fish

Recent studies in Lake Michigan and the other Great Lakes suggest that the disappearance of *Diporeia* is having adverse impacts on the fish community. For example, coincident with the loss of *Diporeia*, the condition, energy density, and abundance of lake whitefish has declined in Lakes Michigan, Huron, and Ontario (Mohr and Nalepa 2005). Lake whitefish is an important commercial species that historically fed heavily on *Diporeia*. Specifically in southern Lake Michigan, condition (i.e., weight per length) of lake whitefish declined 27 % as fish were forced to feed on alternate prey items including *Dreissena* (Pothoven et al. 2001). Populations have been further affected as fish now occupy habitats outside their preferred temperature range as they search for alternate food items (Hoyle 2005, Rennie et al. 2009). Besides lake whitefish, preyfish species such as alewife, sculpin, and bloater have also been impacted. In a study in southeastern Lake Michigan, the diet, abundance, and energy density of these fish species were examined in an area where *Diporeia* had disappeared, and in an area where, at the time, it was still present (Hondorp et al. 2005). Fish in the former area had altered feeding habits, a lower energy density, and diminished abundance compared to fish from the latter area. Consistent with the concept that *D. r. bugensis* is sequestering energy that is inefficiently transferred to fish, standing stock biomass of preyfish has declined dramatically in Lake Michigan. Total prey fish mass (wet weight) declined from 91 kilotonnes in 2005 to 31 kilotonnes in 2007, which is down from 450 kilotonnes in 1989 (Bunnell et al. 2009). Total mass decreased further to 24 kilotonnes in 2008 (C. Madenjian, USGS, personal communication). Similar declines in preyfish biomass have

been documented in Lake Huron; lake-wide biomass of deepwater demersal preyfish declined 87 % between 1994 and 2007 (Riley et al. 2008). While some argue that the decline in preyfish relative to the expansion of *D. r. bugensis* may be more coincidental than mechanistic (Bunnell et al. 2009), continued monitoring of both populations will further define how closely these trends are linked.

### Final Considerations

The decline of *Diporeia* in all the Great Lakes except Lake Superior has reached a point where this native, once-abundant organism no longer plays a significant role in the food web of these lakes. Besides having a direct impact on fish, the loss of *Diporeia* will likely have a cascading, indirect effect on other food web components as fish seek alternate food sources. Indeed, recent studies have shown that pelagic invertebrates such as zooplankton and the opossum shrimp *Mysis* are also declining (Barbiero et al. 2009, Pothoven et al. 2010). Such indirect impacts on other food web components are consistent with early simulation models predicting the consequences of a decline in *Diporeia* (Kitchell et al. 2000). While these declines may be attributed to intensive predation pressure from fish because *Diporeia* is no longer available as a food source, declines can also be attributed to lowered food availability (phytoplankton) resulting from extensive dreissenid filtering activities. As such, the ecological consequences of the *Diporeia* decline may never be fully defined.

Currently, *Diporeia* is still present at depths > 90 m in Lakes Michigan, Huron, and Ontario, but numbers are declining coincident with the continued offshore expansion of *D. r. bugensis* (Watkins et al. 2007, Nalepa et al. 2009). Typically, *Dreissena* populations increase at a rapid rate during the initial expansion phase, but then decline to levels more sustain-

able by the surrounding environment (Strayer and Malcom 2006). This has already happened in some shallow bays and basins of the Great Lakes for *D. polymorpha* (Nalepa et al. 2003, Hunter et al. 2004), and will eventually also happen in deep, offshore regions for *D. r. bugensis*. Yet even if populations of *D. r. bugensis* decline in the future, this species will still be present at some level, making it very unlikely that *Diporeia* will recover. *Diporeia* populations decline and eventually disappear even when *Dreissena* abundances are low, and even in areas far-removed from dreissenid colonies. Without *Diporeia*, future food web models and energy-flow paradigms will need to account for a benthic community structure that no longer efficiently transfers energy, and subsequently is no longer able to support the level of fish resources found in the past.

### Acknowledgements

This is GLERL Contribution No. 1554.

### Literature Cited

- Barbiero, R. P., M. Balcer, D.C. Rockwell, and T. L. Tuchman. 2009. Recent shifts in the crustacean zooplankton community of Lake Huron. *Can. J. Fish. Aquat. Sci.* 66: 816-828.
- Bousfield, E.L. 1989. Revised morphological relationships within the amphipod genera *Pontoporeia* and *Gammaracanthus* and the "glacial relict" significance of their postglacial distributions. *Can. J. Fish. Aquat. Sci.* 46: 1714-1725.
- Bunnell, D. B., C. P. Madenjian, J. D. Holuszko, J. V. Adams, and J. R. P. French, III. 2009. Expansion of *Dreissena* into offshore waters of Lake Michigan and potential impacts on fish populations. *J. Great Lakes Res.* 35, 74-80
- Chase, M. E. and R. C. Bailey. 1999. The ecology of the zebra mussel (*Dreissena polymorpha*) in the lower Great Lakes of North America: II. Total production, energy allocation, and reproductive effort. *J. Great Lakes Res.* 25, 1122-134.
- Cook, D. G. and M. G. Johnson. 1974. Benthic macroinvertebrates of the St. Lawrence Great Lakes. *J. Fish. Res. Bd. Can.* 31: 763-782.
- Dadswell, M. J. 1974. Distribution, ecology, and postglacial dispersal of certain crustaceans

- and fishes in eastern North America. Natl. Mus. Nat. Sci. Can. Publ. Zool. Vol.11, 110 p.
- Dermott, R. 2001. Sudden disappearance of the amphipod *Diporeia* from eastern Lake Ontario, 1993-1995. *J. Great Lakes Res.* 27: 423-433.
- Dermott, R. and K. Corning. 1988. Seasonal ingestion rates of *Pontoporeia hoyi* (Amphipoda) in Lake Ontario. *Can. J. Fish. Aquat. Sci.* 45: 1886-1895.
- Dermott, R. and D. Kerec. 1997. Changes in the deepwater benthos of eastern Lake Erie since the invasion of *Dreissena*: 1979-1993. *Can. J. Fish. Aquat. Sci.* 54: 922-930.
- Dermott, R., R. Bonnell, and P. Jarvis. 2006. Changes in abundance of deep water amphipod *Diporeia* (Pontoporeiidae) in eastern North American lakes with or without *Dreissena* mussels. *Can. Tech. Rep. Fish. Aquat. Sci.*, 88 p.
- Dyble, J., G. L. Fahnenstiel, R. W. Litaker, D. F. Millie, and P. A. Tester. 2008. Microcystin concentrations and genetic diversity of *Microcystis* in the lower Great Lakes. *Environ. Toxicol.* 23:507-516.
- Fahnenstiel, G. L., S. A. Pothoven, T. F. Nalepa, H. A. Vanderploeg, D. Klarer, and D. Scavia. 2010. Recent changes in primary production and phytoplankton in the offshore region of southeastern Lake Michigan. *J. Great Lakes Res.* 36 (Suppl. 3):20-29.
- Foley, A. J. III, T. F. Nalepa, G. K. Walker, and M. J. McCormick. 2006. Epibiont populations associated with *Diporeia* spp. (Amphipoda) from Lake Michigan. *Verh. Internat. Verein. Limnol.* 29:1205-1211.
- Gardner, W.S., T. F. Nalepa, W. A. Frez, W.A., E. A. Cichocki, and P. F. Landrum. 1985. Seasonal patterns in lipid content of Lake Michigan macroinvertebrates. *Can. J. Fish. Aquat. Sci.* 42: 1827-1832.
- Gardner, W.S., M. A. Quigley, G. L. Fahnenstiel, D. Scavia, and W. A. Frez. 1990. *Pontoporeia hoyi* - a direct trophic link between spring diatoms and fish in Lake Michigan. In *Large Lakes: Structure and Functional Properties*. Edited by M.M. Tilzer and C. Serruya. Springer, pp. 632-644.
- Gauvin, J. M., W. S. Gardner, and M. A. Quigley. 1989. Effects of food removal on nutrient release rates and lipid content of Lake Michigan *Pontoporeia hoyi*. *Can. J. Fish. Aquat. Sci.* 46: 1125-1130.
- Guiguer, K. R. A. and D. R. Barton. 2002. The trophic role of *Diporeia* (Amphipoda) in Colpoys Bay (Georgian Bay) benthic food web: a stable isotope approach. *J. Great Lakes Res.* 28: 228-239.
- Hondorp, D. W., S. A. Pothoven, and S. B. Brandt. 2005. Influence of *Diporeia* density on the diet composition, relative abundance, and energy density of planktivorous fishes in southeast Lake Michigan. *Trans. Amer. Fish. Soc.* 134: 588-601.
- Hoyle, J. A. 2005. Status of lake whitefish (*Coregonus clupeaformis*) in Lake Ontario and the response to the disappearance of *Diporeia* spp. In *Proceedings of a workshop on the dynamics of lake whitefish (*Coregonus clupeaformis*) and the amphipod *Diporeia* spp. in the Great Lakes*. Edited by L. C. Mohr and T. F. Nalepa. *Great Lakes Fish. Comm. Tech. Rep.* 66, pp. 47-66.
- Hunter, R. D. and K. A. Simons. 2004. Dreissenids in Lake St. Clair in 2001: evidence for population regulation. *J. Great Lakes Res.* 30: 528-537.
- Johnson, M.G. and O. C. McNeil. 1986. Changes in abundance and species composition in benthic macroinvertebrate communities of the Bay of Quinte, 1966-84. *Can. Spec. Publ. Fish. Aquat. Sci.* 86: 177-189.
- Kaiz, M.J., O. E. Johannsson, and M. T. Arts. 2010. Diet effects on lipid composition, somatic growth potential, and survival of the benthic amphipod *Diporeia* spp. *J. Great Lakes Res.* 36: 351-356.
- Kitchell, J.F., S. P. Cox, C. J. Harvey, T. B. Johnson, D. M. Mason, K. K. Schoen, K. Aydin, C. Bronte, M. Ebener, M. Hansen, M. Hoff, S. Schram, D. Schreiner, and C. J. Walters. 2000. Sustainability of the Lake Superior fish community: interactions in a food web context. *Ecosystems* 3: 545-560.
- Lozano, S.J., J. V. Scharold, and T. F. Nalepa. 2001. Recent declines in benthic macroinvertebrate densities in Lake Ontario. *Can. J. Fish. Aquat. Sci.* 58: 518-529.
- MacDougall, T. M., H. P. Benoit, R. Dermott, O. E. Johannsson, T. B. Johnson, E. S. Millard, and M. Munawar. 2001. Lake Erie 1998: Assessment of abundance, biomass, and production of the lower trophic levels, diets of juvenile yellow perch and trends in the fishery. *Can. Tech. Rep. Fish. Aquat. Sci.* No. 2376, 190 p.
- McDonald, M.E., L. B. Crowder, and S. B. Brandt. 1990. Changes in *Mysis* and *Pontoporeia* populations in southeastern Lake Michigan: A response to shifts in the fish community. *Limnol. Oceanogr.* 35: 220-227.
- McNickle, G. G., M. D. Rennie, and W. G. Sprules. 2006. Changes in benthic communities of South Bay, Lake Huron following invasion by zebra mussels (*Dreissena polymorpha*), and potential effects on Lake Whitefish (*Coregonus clupeaformis*) diet and growth. *J. Great Lakes Res.* 32:180-913.
- Messick, G. A., R. M. Overstreet, T. F. Nalepa, and S. Tyler. 2004. Prevalence of parasites in amphipods *Diporeia* spp. from Lakes Michigan and Huron, USA. *Diseases of Aquatic Organisms* 59:159-170.
- Mida, J. L., D. Scavia, G. L. Fahnenstiel, S.A.

- Pothoven, H.A. Vanderploeg, and D.M. Dolan. 2010. Long-term and recent changes in southern Lake Michigan water quality with implications for primary production. *J. Great Lakes Res.* 36 (Suppl. 3): 42-49.
- Mohr, L. C. and T. F. Nalepa. 2005. Proceedings of a workshop on the dynamics of lake whitefish (*Coregonus clupeaformis*) and the amphipod *Diporeia* spp. in the Great Lakes. Great Lakes Fish. Comm. Tech. Rep. 66, Ann Arbor, MI.
- Nalepa, T. F. 1987. Long term changes in the macrobenthos of southern Lake Michigan. *Can. J. Fish. Aquat. Sci.* 44: 515-524.
- Nalepa, T. F. 1989. Estimates of macroinvertebrate biomass in Lake Michigan. *J. Great Lakes Res.* 15:437-443.
- Nalepa, T. F., D. J. Hartson, G. L. Fanslow, G. A. Lang, and S. J. Lozano. 1998. Declines in benthic macroinvertebrate populations in southern Lake Michigan, 1980-1993. *Can. J. Fish. Aquat. Sci.* 55: 2402-2413.
- Nalepa, T. F., D. W. Schloesser, D. W., S. A. Pothoven, D. W. Hondorp, D. L. Fanslow, M. L. Tuchman, and G. W. Fleischer. 2001. First finding of the amphipod *Echinogammarus ischnus* and the mussel *Dreissena bugensis* in Lake Michigan. *J. Great Lakes Res.* 27: 384-391.
- Nalepa, T. F., D. L. Fanslow, M. B. Lansing, and G. A. Lang. 2003. Trends in the benthic macroinvertebrate community of Saginaw Bay, Lake Huron, 1987 to 1996: responses to phosphorus abatement and the zebra mussel, *Dreissena polymorpha*. *J. Great Lakes Res.* 29: 14-33.
- Nalepa, T. F., D. C. Rockwell, and D. W. Schloesser. 2006a. Disappearance of the amphipod *Diporeia* spp. in the Great Lakes: Workshop summary, discussion, and recommendations. NOAA Technical Memorandum GLERL-136. NOAA, Great Lakes Environmental Research Laboratory, Ann Arbor, MI, 20 pp.
- Nalepa, T. F., D. L. Fanslow, A. F. Foley III, G. A. Lang, B. J. Eadie, and M. A. Quigley. 2006b. Continued disappearance of the benthic amphipod *Diporeia* spp. in Lake Michigan: is there evidence for food limitation? *Can. J. Fish. Aquat. Sci.* 63: 872-890.
- Nalepa, T. F., D. L. Fanslow, and G. A. Lang. 2009. Transformation of the offshore benthic community in Lake Michigan: recent shift from the native amphipod *Diporeia* spp. to the invasive mussel *Dreissena rostriformis bugensis*. *Freshwater Biology* 54, 466-479.
- Nalepa, T. F., D. L. Fanslow, and S. A. Pothoven. 2010. Recent changes in density, biomass, recruitment, size structure, and nutritional state of *Dreissena* populations in southern Lake Michigan. *J. Great Lakes Res.* 36 (Suppl. 3): 5-19.
- Perez-Fuentetaja, M. D. Clapsadl, D. Einhouse, P. R. Bowser, R. G. Getchell, and W. T. Lee. 2006. Influence of limnological conditions on *Clostridium botulinum* type E presence in eastern Lake Erie (Great Lakes, USA) sediments. *Hydrobiologia* 563: 189-200.
- Pothoven, S.A., T. F. Nalepa, P. J. Schneeberger, and S. B. Brandt. 2001. Changes in diet and body condition of lake whitefish in southern Lake Michigan associated with changes in benthos. *N. Amer. J. Fish. Managmnt.* 21: 876-883.
- Pothoven, S. A., G. L. Fahnenstiel, and H. A. Vanderploeg. 2010. Temporal trends in *Mysis relicta* abundance, production, and life history characteristics in southeastern Lake Michigan. *J. Great Lakes Res.* 36 (Suppl. 3): 60-64.
- Rennie, M. D., W. G. Sprules, and T. B. Johnson. 2009. Resource switching in fish following a major food web disruption. *Oecologia* 159: 789-802.
- Riley, S. C., E. F. Roseman, S. J. Nichols, J. E. O'Brien, and J. S. Schaeffer. 2008. Deepwater demersal fish community collapse in Lake Huron. *Trans. Amer. Fish. Soc.* 137: 1879-1890.
- Robertson, A. and W. P. Alley. 1966. A comparative study of Lake Michigan macrobenthos. *Limnol. Oceanogr.* 11:576-583.
- Scharold, J. V., S. J. Lozano, and T. D. Corry. 2004. Status of the amphipod *Diporeia* spp. in Lake Superior, 1994-2000. *J. Great Lakes Res.* 30 (Suppl.1): 360-368.
- Scott, W.B. and E. J. Crossman. 1973. *Freshwater Fishes of Canada*. Bull. Fish. Res. Board Can. 184.
- Sierszen, M. E., G. S. Peterson, and J. V. Scharold. 2006. Depth-specific patterns in benthic-pelagic food web relationships in Lake Superior. *Can. J. Fish. Aquat. Sci.* 63: 1496-1503.
- Strayer, D. L. and H. M. Malcom. 2006. Long-term demography of a zebra (*Dreissena polymorpha*) population. *Freshwater Biology* 51: 117-130.
- Vanderploeg, H. A., J. R. Liebig, W. W. Carmichael, M. A. Agy, T. H. Johengen, G. L. Fahnenstiel, and T. F. Nalepa. 2001. Zebra mussel (*Dreissena polymorpha*) selective filtration promoted toxic *Microcystis* bloom in Saginaw Bay (Lake Huron) and Lake Erie. *Can. J. Fish. Aquat. Sci.* 58:1208-1221.
- Watkins, J. M., R. Dermott, S. J. Lozano, E. L. Mills, L. G. Rudstram, J. V. Scharold. 2007. Evidence for remote effects of dreissenid mussels on the amphipod *Diporeia*: Analysis of the Lake Ontario benthic surveys, 1972-2003. *J. Great Lakes Res.* 33, 642-657.
- Wells, L. 1980. Food of alewives, yellow perch, spottail shiners, trout-perch, and slimy and fourhorn sculpins in southeastern Lake Michigan. *U. S. Fish Wildl. Serv. Tech. Rep.* 98.



# The Kirtland's Warbler: Interagency and International Cooperation works

David N. Ewert<sup>1</sup>

Joseph M. Wunderle, Jr.<sup>2</sup>

Christopher W. Eberly<sup>3</sup>

Philip W. Huber<sup>4</sup>

Eric Carey<sup>5</sup>

<sup>1</sup>The Nature Conservancy  
Lansing, Michigan

<sup>2</sup>U.S. Forest Service  
Luquillo, Puerto Rico

<sup>3</sup>U.S. Department of Defense,  
Partners in Flight  
Warrenton, Virginia

<sup>4</sup>U.S. Forest Service  
Mio, Michigan

<sup>5</sup>Bahamas National Trust  
Nassau, The Bahamas

## Introduction

Protection of migratory species requires understanding the ecology and demography of a species throughout its life cycle. This requires evaluating ecological requirements and survivorship during the breeding and wintering seasons as well as the migratory period. Obtaining these data is exceedingly difficult due to methodological limitations associated with (1) tracking individual birds throughout their life cycle and (2) being able to assess population changes during each part of the life cycle (Sillert and Holmes 2002, Webster and Marra 2005). Equally challenging is the ability to coordinate conservation work across jurisdictional boundaries, especially international boundaries where different legal, cultural and socioeconomic factors must be integrated to ensure that effective conservation programs can be sustained. This paper focuses on recent efforts in the Bahamian wintering grounds of the Kirtland's Warbler (*Dendroica kirtlandii*), a federally listed species in the United States and Canada, to fill gaps needed to develop a comprehensive range wide conservation program for this species.

The Kirtland's Warbler, North America's rarest songbird, has one of the most limited ranges of any landbird on the continent and breeds almost exclusively in Michigan, though some breeding occurs in Wisconsin and Ontario, in early succession jack pine (*Pinus banksiana*) stands. It winters in short, dense scrub in The Bahamas and Turks and Caicos Islands. The rarity of the Kirtland's Warblers coupled with its highly specific habitat requirements has required intensive research to identify appropriate management strategies to ensure survival of this conservation-reliant species. Research on the breeding grounds has guided development of two principal management activities which have successfully improved habitat and habitat conditions: (1) management of jack pine forests to ensure there are relatively large patches of early successional jack pine forest and (2) population control of a nest parasite, the Brown-headed Cowbird (*Molothrus ater*). Approximately 87,200 ha are dedicated to Kirtland's Warbler management in Michigan. About 1,400 hectares are planted with jack pine annually to ensure there is sufficient habitat for the species to maintain the recovery goal of a minimum of 1,000 pairs. Response of Kirtland's Warblers to these treatments has been evaluated through demographic studies (Bocetti 1994; Sarah Rockwell, personal communication, 2009), annual censuses conducted since 1971 (Mayfield 1992), monitoring productivity of parasitized versus nonparasitized nests (Bocetti 1994), and many other studies. This work has been coordinated through

This photo is of the 2009 Kirtland's Warbler Research and Training Project taken at Tarpum Bay, Eleuthera, Bahamas. From left to right (back row): Dave Ewert, Joe Wunderle. From left to right (front row): Elise Corliss, Jennifer Thieme, Scott Johnson, Jennifer White.



the Kirtland's Warbler Recovery Team (KWRT), a consortium of agency biologists and managers responsible for management of the species, including a representative from the Bahamas National Trust, which has overseen the recovery of the species from an estimated 167 singing males in Michigan's lower peninsula in 1987 to approximately 1,813 singing males in both the lower and upper peninsulas of Michigan, Wisconsin and Ontario in 2009.

Recently there has been increased emphasis on the study and conservation of the Kirtland's Warbler during migration and on the wintering grounds. All known locations of Kirtland's Warblers found during migration have been compiled by Petrucha and Huber (unpublished data) and priority areas for protection of landbird stopover sites have been compiled in the western Lake Erie basin of Ohio and Michigan,

where Kirtland's Warblers are reported relatively frequently during migration (Ewert et al. 2006). This paper focuses primarily on ongoing studies of wintering Kirtland's Warblers in The Bahamas, primarily through the Kirtland's Warbler Research and Training Project and related work, including locating Bahamas-banded birds on the Michigan breeding grounds. We briefly describe results from the work to date and emphasize how collaboration has furthered our goal of contributing to a comprehensive, range-wide approach to conservation of this highly endangered species.

### **The Kirtland's Warbler Research and Training Project: a focus on the wintering grounds**

The Kirtland's Warbler Recovery Plan lists activities to ensure survival of the Kirtland's Warbler on the wintering



grounds – identification of habitat used by Kirtland’s Warblers, management activities needed to create this habitat, and increasing conservation capacity to ensure habitat availability on The Bahamas (Kirtland’s Warbler Recovery Team 1985). In response to these needs, we initiated the Kirtland’s Warbler Research and Training Project in 2002, with four principal objectives: (1) to provide a description of the ecology of wintering Kirtland’s Warblers and associated resident and migratory species on the Bahamian wintering grounds, (2) to examine connectivity between the wintering and breeding grounds, (3) to convert research findings to management guidelines and other outreach activities for the Kirtland’s Warblers on the wintering grounds, and (4) to increase conservation capacity in The Bahamas. These objectives were formulated based on needs articulated by the Kirtland’s Warbler Recovery Team and subsequent discussions between Bahamian and United States conservationists, biologists, and others responsible for protection of the Kirtland’s Warbler. Objectives for the wintering grounds coincided with Bahamian conservation needs as expressed by a past president of the Bahamas National Trust who stated “...Our [Bahamas] greatest needs are massive public and decision-maker education and intensified scientific research.” (in Raffaele et al. 1998). These conservation approaches are needed to implement habitat restoration and increase quality of human life and thus compensate for past land-use practices that

have degraded the biodiversity of The Bahamas

The Kirtland’s Warbler Research and Training Project requires integration of research, conservation planning, fund raising, education, communication, knowledge of regulations in both The Bahamas and the United States and understanding of Bahamian and upper Midwest cultural and economic issues, all of which require extensive collaboration to achieve our ultimate goal of defining management practices needed to sustain the species in the Bahamas archipelago.

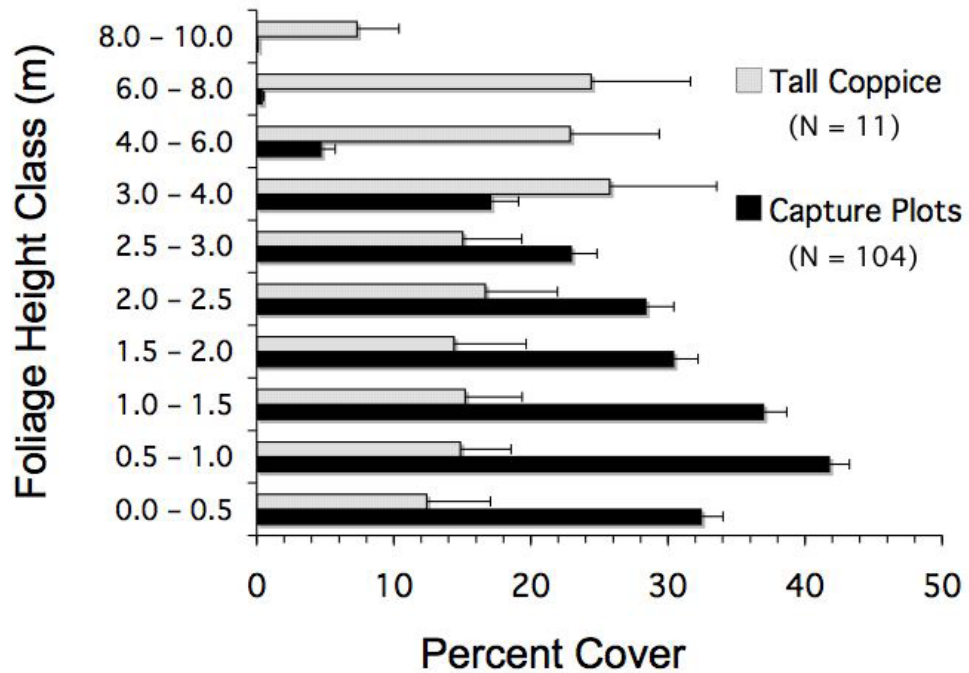
### *Research*

Wintering grounds results. Kirtland’s Warblers are difficult to locate on the wintering grounds where they are unevenly distributed across the landscape (Mayfield 1992, Currie et al. 2005). In spite of this challenge, the wintering



**Figure 1:**

Foliage height profiles showing percentage cover at various height categories in Kirtland's Warbler capture plots and randomly selected tall coppice plots from Eleuthera, The Bahamas. N indicates the number of plots sampled in capture and tall coppice plots. Foliage profiles indicate Kirtland's Warblers occupy low stature brushy or scrubby sites rather than tall, mature coppice (forest) sites. Figure modified from Wunderle et al. (2010) where methods are described.



grounds research on the island of Eleuthera has been successful in characterizing the winter habitat (Wunderle et al., 2010) and has demonstrated that Kirtland's Warblers use anthropogenically disturbed early successional habitats that range in age from 3 to 28 years following disturbance. Warblers occurred in sites that had been disturbed by clearing for construction, agriculture, goat farming and fire. Sites where the warblers were captured had canopies of lower stature (mean 1.8 m) and had more foliage near the ground (Figure 1) than late successional tall coppice sites (mean 6.3 m). These sites typically had lower canopies (mean 1.8 m) than randomly selected sites (mean 2.7 m). Although the seven study sites were heterogeneous in terms of time since last disturbance, most foliage height categories, canopy height, stem density and five ground cover traits (e.g., leaf litter, grass, wood, rock, bare ground), the density of vegetation at 0.5-1 m was

consistent among study sites sampled. Kirtland's Warbler frequently foraged on small fruits on shrubs, especially those on wild sage (*Lantana involucrata*), black torch (*Erithalis fruticosa*), and snowberry (*Chiococca alba*); these species occurred at higher frequency on plots where Kirtland's Warblers were found compared to random plots (Wunderle et al. 2010). At both site and landscape levels, fruiting of these three shrubs can vary considerably between the times Kirtland's Warblers arrive in The Bahamas in the fall until they depart in April. Kirtland's Warblers appear to track fruit abundance as some individuals moved up to several kilometers from fruit-poor to fruit-rich sites during the winter. These results suggest that a landscape approach will be essential to ensure availability of high quality habitat across time and space for this species. Research collaborators include Dr. Peter Marra (Smithsonian Institution) and Ph.D. student Sarah Rockwell, Dr.

Eileen Helmer (International Institute of Tropical Forestry, U.S. Forest Service, Rio Piedras, Puerto Rico), Dr. Charles Kwit (University of Tennessee), Dr. Bernard Parresol (Southern Research Station, U.S. Forest Service, Asheville, North Carolina), Dr. Joseph O'Brien (Southern Research Station, U.S. Forest Service, Athens, Georgia), and Dr. Kimberly Hall (The Nature Conservancy).

Connectivity results. Two approaches are being applied to evaluate connectivity between wintering sites on Eleuthera, The Bahamas, and the breeding grounds: (1) observations of color-banded birds seen in both The Bahamas and the breeding grounds and (2) isotope signatures. Of the approximately 50 color-banded individuals found on the breeding grounds that were banded in The Bahamas, most were found widely scattered in Michigan's Lower Peninsula but some were also found in Wisconsin, Michigan's Upper Peninsula, and Ontario (Ewert et al., unpublished data). These observations suggest that Kirtland's Warblers wintering at one site widely disperse across the breeding range and further emphasizes the need for a landscape approach to habitat protection in The Bahamas. Collection and analysis of isotope data, based on collections of feathers and other tissues in both The Bahamas and Michigan are underway. These data will be used to better delineate connectivity between breeding and wintering areas, establish relationships between arrival times on the breeding grounds and environmental conditions on The Bahamas, and evaluate diet of overwintering Kirtland's Warblers.

### ***Outreach and management***

Outreach results and projected activities. Outreach has been achieved through extensive collaboration with local schools in southern Eleuthera, the Bahamas National Trust, Kirtland Com-

munity College, The Nature Conservancy, and the U.S. Forest Service in Michigan. In The Bahamas, student project participants give presentations on the Kirtland's Warbler and associated flora and fauna to primary and secondary students. In addition, the Bahamian student project participants coordinate local elementary school participation in Kirtland's Warbler artwork contributed to the annual Kirtland's Warbler calendar contest sponsored by Kirtland Community College. These activities generate local interest for the project, facilitate access to potential study sites and set the stage for potential management work with local landowners. In addition, presentations on the project have been made to the public, at science conferences, and to government officials in the Bahamas. Project brochures and posters have been produced and distributed throughout much of the Bahamas. Increased outreach activities are planned including distribution of posters on the project at public centers around the country. Other presentations have been made on the project at meetings and conferences throughout the United States, the Caribbean, Mexico, and Germany to increase support for conservation of the Kirtland's Warbler. Presentations are given annually at the Kirtland's Warbler Festival sponsored by Kirtland Community College, Roscommon, Michigan.

Proposed management results activities. Experimental management activities are now being initiated and require testing before encouraging implementation of these techniques. Our research indicates that re-occurring habitat disturbance will be required to produce early successional habitats with key fruiting species needed by wintering Kirtland's Warblers. We are focusing potential management on activities that produce, enhance, or maintain wintering Kirtland's Warbler habitat that will

be cost-effective, practical and sustainable over the long-term; it is unlikely there will be extensive management solely for the Kirtland's Warbler at a large scale in the Bahamas archipelago. Thus a key to success will be identifying those re-occurring natural and anthropogenic disturbance types which can be harnessed in a cost effective manner to facilitate production and maintenance of the warbler's winter habitat. To increase the likelihood of success we are initiating trial conservation programs in The Bahamas with diverse groups from the public and private sectors including: (1) working with resorts and botanical gardens where the presence of Kirtland's Warblers could enhance ecotourism, (2) working with utility companies to manage rights-of-ways that will create or maintain Kirtland's Warbler habitat by retarding vegetative succession, (3) working with goat farmers to encourage goat management practices (goats avoid foraging on two shrub species, wild sage and black torch, favored by Kirtland's Warblers) that retain the fruiting shrubs used extensively by Kirtland's Warblers, (4) evaluating the planting of firebreaks with shrubs used by Kirtland's Warblers and (5) working with local schools to encourage the planting of wild sage, black torch and snowberry. On the northern pine islands of The Bahamas (especially Andros, Grand Bahamas and Abaco) it may be possible to develop fire management approaches to create early successional habitat which benefit Kirtland's Warblers if Bahamian legislation permitting prescribed fires is adopted.

To implement these conservation approaches and activities the Kirtland's Warbler Research and Training Project will continue to build on our established working relationships with owners or managers of private or community lands and government officials. Thus far, many collaborators, such as the fol-

lowing, have actively participated with this aspect of the work or expressed interest in considering potential management activities: Bahamas National Trust, Bahamian government, Friends of the Environment (Abaco), Bahamas Electricity Corporation, The Nature Conservancy, the U.S. Forest Service, Leon Levy Native Plant Preserve, Cape Eleuthera Development Corporation, goat farmers on Eleuthera, community organizations (commonage and generational land committees) on Eleuthera and others.

### *Increasing conservation capacity*

Results. The project has successfully provided rigorous field training for nine Bahamian students. The project also provided each student an opportunity to spend one summer in Michigan to become acquainted with the Kirtland's Warbler on the breeding grounds, learn complementary skills to those acquired during the winter project and to establish working relationships with United States conservationists. The Huron-Manistee National Forests and Kirtland Community College were instrumental among the collaborators who facilitated this aspect of the training program.

Seven of the students have gone on to attend universities in the United States (Cornell University, University of Minnesota, Northern Michigan University, University of Maryland-Eastern Shore, State University of New York at Buffalo). Each of these universities provided funding to help support education of these students in addition to funding provided by private donors and foundations through The Nature Conservancy. So far, four of these students have graduated with a B.S. degree in Biology or natural resource management from United States universities and one graduated from the College of the Bahamas. Two of these Bahamian students continued their studies in the United States and have obtained M.S.

degrees in ecology. Thus far, two students have returned to The Bahamas and taken conservation positions, one with The Nature Conservancy and one with the Bahamas Environment, Science and Technology Commission. Their work in The Bahamas includes analysis of environmental impact statements, conservation planning for marine and terrestrial parks and outreach activities.

Projected activities. We will continue to work with the College of the Bahamas, the Bahamas National Trust and the Northern Caribbean program of The Nature Conservancy to build educational and conservation capacity by encouraging student participation in the project and encouraging students to consider research and conservation internships with these organizations. The ultimate goal of the project is to have Bahamians assume management of the project and become conservation custodians for the Kirtland's Warblers throughout the country.

### Summary

As habitat continues to be lost to development in The Bahamas, the two main challenges facing implementation of conservation programs for the Kirtland's Warbler are: (1) the lack of protected areas, which means that much Kirtland's Warbler habitat conservation must occur on private or community lands and (2) the need for re-occurring habitat disturbance to produce the ear-



ly successional habitats with key fruiting plant species required by the warbler. These two challenges could prove prohibitively expensive for Kirtland's Warbler conservation on the wintering grounds and therefore it is essential to identify the most cost effective and sustainable means for habitat conservation. Ideally, we wish to identify management strategies that pay their way to ensure sustainability, although we recognize that the economic viability of any management strategy will vary over time. Consequently, we propose several conservation approaches, which will allow an assessment of the most effective and sustainable model(s) for Kirtland's Warbler habitat conservation. Conditions require that we must be opportunistic in our selection of conservation approaches and sites and work with a diversity of coopera-

tors on an array of Kirtland's Warbler management sites scattered across the Eleuthera landscape and elsewhere in The Bahamas.

The Kirtland's Warbler is particularly amenable to range wide conservation approaches due to its limited range and the highly effective, multinational KWRT which promotes collaborative work in the four countries inhabited by this species, the United States, Canada, The Bahamas and Turks and Caicos Islands. These countries are strongly linked economically and share many cultural and legal frameworks due to their history and proximity. These linkages facilitate establishing collaborative programs, such as the Kirtland's Warbler Research and Training Project, which is focused on the ecology and management of this species on the Bahamian archipelago wintering grounds. The collective work of many institutions and individuals, including the Kirtland's Warbler Research and Training Project, is furthering range-wide conservation of the Kirtland's Warblers on the breeding and wintering grounds and along the migration route.

### *Acknowledgements*

We thank Paul Dean and others from the Ornithology Group of the Bahamas National Trust for alerting us to the presence of a concentration of Kirtland's Warblers on Eleuthera. We thank project field directors, including Dave Currie, Jennifer White, and Genie Fleming and a diverse group of North American field assistants for their dedicated efforts. The project has benefited from the participation of Bahamian students, including Ancilleno Davis, Scott Johnson, Everton Joseph, Samara Lawrentz, Zeko McKenzie, Ingeria Miller, Keith Philippe, Montara Roberts, and Jasmin Turner. Funding has been primarily provided by International Programs of the U.S. Department of Agriculture Forest Service, the Institute of Tropical Forestry of the U.S. Department of Agriculture Forest Service, U.S. Department of Defense, The Nature Conservancy, Huron-Manistee National Forests, and the U.S. Fish and Wildlife Service. Field work on the breeding grounds has been facilitated by the Kirtland's Warbler Recovery Team, Huron-Manistee National Forest, U.S. Fish and Wildlife Ser-

vice, Department of National Defence (Canada), Environment Canada, Michigan Department of Natural Resources and Environment, Kirtland Community College, Smithsonian Institution, and others. We also thank the many collaborators who have worked with us in The Bahamas, including the Bahamas National Trust, Cape Eleuthera Development Corporation, the Island School, commonage committees on south Eleuthera, and various landowners who have provided us access to their land for our work.

### **Literature Cited**

- Bocetti, C.I. 1994. Density, demography, and mating success of Kirtland's Warblers on managed and natural habitat. Dissertation, The Ohio State University, Columbus.
- Currie, D., J.M. Wunderle, Jr., D.N. Ewert, M. Anderson, A. Davis, and Z. McKenzie. 2005. Winter avian distribution in six terrestrial habitats on southern Eleuthera, The Bahamas. *Caribbean Journal of Science* 41:88-100.
- Ewert, D.N., G.J. Soulliere, R.D. Macleod, M.C. Shieldcastle, P.G. Rodewald, E. Fujimura, J. Shieldcastle, and R.J. Gates. 2006. Migratory bird stopover site attributes in the western Lake Erie basin. Final report to The George Gund Foundation.
- Kirtland's Warbler Recovery Team. 1985. Kirtland's Warbler Recovery Team 1976, Updated 1985.
- Mayfield, H.F. 1992. Kirtland's Warbler (*Dendroica kirtlandii*), no. 19. The birds of North America, ed. A. Poole. Philadelphia: Academy of Natural Sciences.
- Raffaele, H., J. Wiley, O. Garrido, A. Keith, and J. Raffaele. 1998. A guide to the birds of the West Indies. Princeton University Press, Princeton, New Jersey.
- Sillett, T.S. and R. T. Holmes. 2002. Variation in survivorship of a migratory songbird throughout its annual cycle. *Journal of Animal Ecology* 71:296-308.
- Webster, M.S. and P.P. Marra. 2005. The importance of understanding migratory connectivity and seasonal interactions. Pp 199-209. In *Birds of Two Worlds: the ecology and evolution of migration* (R. Greenberg and P.P. Marra, editors). John Hopkins University Press, Baltimore, Maryland.
- Wunderle, J.M., Jr., D. Currie, E.H. Helmer, D.N. Ewert, J.D. White, T.S. Ruzycski, B. Parresol, and C. Kwit. 2010. Kirtland's Warblers in anthropogenically disturbed early-successional habitats on Eleuthera, The Bahamas. *Condor* 112:123-137.

# Fire History and Its Implications for an Endemic Fire-Dependent Herb (*Iliamna corei* Sherff.) of the Appalachian Mountains



**Charles W. Lafon**<sup>1</sup>  
Jennifer A. Hoss<sup>2</sup>  
Henri D. Grissino-Mayer<sup>3</sup>  
Serena R. Aldrich<sup>4</sup>  
Georgina G. DeWeese<sup>5</sup>

<sup>1</sup> Department of Geography, Texas A&M University, 3147 TAMU, College Station, TX 77843-3147, clafon@geog.tamu.edu

<sup>2</sup> 6112 Moose Court, El Paso, TX, 79924, ja.hoss@gmail.com

<sup>3</sup> Department of Geography, University of Tennessee, Knoxville, TN 37996-0925, grissino@utk.edu

<sup>4</sup> Department of Geography, Texas A&M University, 3147 TAMU, College Station, TX 77843-3147, serena@neo.tamu.edu

<sup>5</sup> Department of Geosciences, University of West Georgia, Carrollton, GA, 30118, georgina.deweese@gmail.com

## Abstract

The Peters Mountain mallow is a fire-dependent herbaceous plant species endemic to Peters Mountain, Virginia. Its population declined over the twentieth century, likely because of fire exclusion. We used fire-scarred trees to reconstruct the fire history of Peters Mountain (Hoss et al. 2008) and found that fires occurred frequently in the past, before fire protection became commonly practiced. The mean fire interval for the site was 2.5 years, and most fires occurred during the dormant season (spring or fall). Fire frequency is lower today. In 2005, at the time of our fieldwork, 29 years had elapsed since the last fire. The results suggest the need to reintroduce fire to restore the Peters Mountain mallow and its habitat.

## Introduction

In 1989, the entire known wild population of the Peters Mountain mallow (*Iliamna corei* Sherff.; Figure 1) comprised only three individuals (Baskin and Baskin 1997), making it one of the rarest plant species on earth. Apparently, the species was never abundant. When discovered in 1927, the population of the perennial herb included about 50 plants growing on thin, rocky soils among widely spaced trees (Strausbaugh and Core 1932) on the western end of Peters Mountain, Virginia. About 40 individuals remained in 1962 (Keener and Hardin 1962), and at some point thereafter the population plummeted.



A Peters Mountain mallow in flower. The plants are fenced to protect them from deer browsing.

Remnant of a pine tree at Peters Mountain that was scarred by multiple fires over its life. The ridges on the surface formed when wood grew over each successive fire scar.



ed to the low level recorded in 1989.

Autecological work (Baskin and Baskin 1997) revealed a large bank of dormant, water-impermeable mallow seeds that require heating by fire to germinate. The plants also resprout after burning, and because of their shade intolerance they need fire to maintain an open, well lit habitat. These life-history traits appear to suit the mallow to a

Detail of a fire-scarred cross-section. We sanded the cross-sections to make the rings and scars readily visible for dating. This photograph shows eight fire scars, with the oldest on the right and the most recently formed scar on the left. Photo credit: Jean Wulfson, Division of Research and Graduate Studies, Texas A&M University.



short fire interval, likely on the order of only a few years (Hoss et al. 2008; cf. Rowe 1983). But fire prevention and suppression efforts had begun to reduce the frequency of fire in the Appalachian region by the 1940s and 1950s (Sarvis 1993). Fire exclusion is thought to have contributed to the decline of the mallow population (Caljouw et al. 1994). The Nature Conservancy purchased the habitat of the endangered Peters Mountain mallow in 1992 and began experimental burns, which resulted in a larger mallow population that has fluctuated annually in the number of individuals present (Edwards and Allen 2003). Restoring the Peters Mountain mallow and its habitat requires an understanding of the historic fire regime(s)

under which the plants thrive.

### **A study of fire history on Peters Mountain**

We cut and dated cross-sections from 73 fire-scarred pine trees within a 40 ha area surrounding the mallows (Figures 2, 3). We also cored all the trees growing in two plots that were situated near the mallows to illuminate how changing

fire activity affected tree establishment in the vicinity of the mallows. Hoss et al. (2008) report the study in detail.

The trees recorded 53 fires during the period 1794–2005. Between 1867 (the first year with two or more scarred trees) and 1976 (the last year recording a fire), the mean fire interval was 2.5 years. That



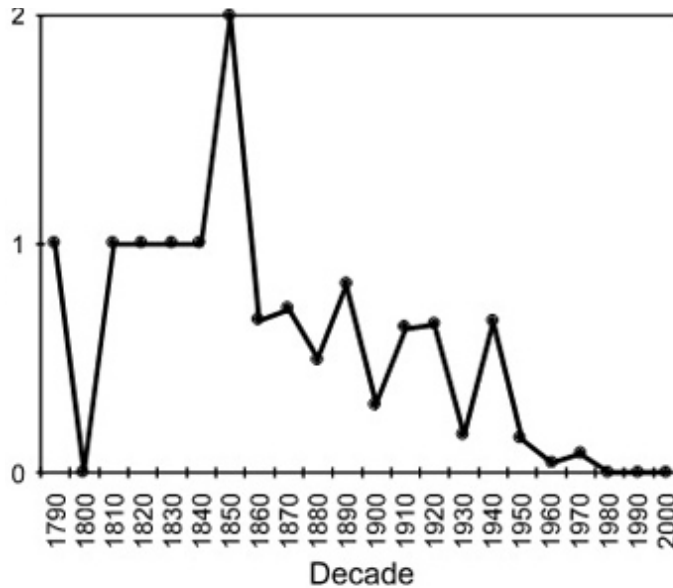
is, at least part (but not necessarily all) of the area burned every 2–3 years. We also calculated a more conservative estimate of fire frequency: large fires that scarred at least 25% of the trees across the study area occurred about every 12.5 years.

Most (93.6%) of the fire scars formed while the trees were dormant (i.e., scars are positioned between the annual rings), suggesting that the fires occurred during the spring or fall; fires rarely burn in the Appalachian Mountains during winter (Lafon et al. 2005).

Fire frequency remained high through the 1940s, then declined (Figure 4), probably because of fire prevention/suppression. Most of the trees we cored became established during the 1950s and 1960s, following the cessation of frequent fire. Only the fire-resistant chestnut oaks (*Quercus montana* Willd.) and pines (*Pinus* spp.) were able to establish under frequent burning before the 1950s. These results suggest that the mallows thrived historically in a more open, fire-maintained woodland. With reduced burning after the 1940s, the forests grew denser and the mallows encountered increasing competition for light and other resources. The lack of fire also would have precluded reproduction by seed, hence the small population remaining in 1989.

### Implications of the study

Our study supports the need to manage the Peters Mountain mallow with fire. It would appear that the most prudent restoration strategy would entail burning at a frequency and seasonality



**Figure 1:** Temporal trend in fire frequency at Peters Mountain. For each decade, the number of fire scars was divided by the number of trees available to record fire during that decade. This denominator differed among decades because not all trees covered the same range of dates. The sample size was low in the early portion of the record.

similar to that which maintained the mallow historically. Mechanical clearing of the forest around the mallows could be important, too, for promoting the establishment, growth, and reproduction of the mallows. More broadly, this research underscores the important historical and ecological role of fire in the temperate forest region of eastern North America.

### Acknowledgements

This research was funded by The Nature Conservancy and by the National Interagency Fire Center's Joint Fire Science Program.

### Literature Cited

Baskin, J.M. and Baskin, C.C. 1997. Methods of breaking seed dormancy in the endangered *Ilamna corei* (Sherff) (Malvaceae), with special attention to heating. *Natural Areas Journal* 17:313–323.

Caljouw, C.A., Lipscomb, M.V., Adams, H.S., and Hobbs, M.S. 1994. Prescribed Burn and Disturbance History Studies at the Narrows:

- Habitat Studies for the Endangered Peters Mountain Mallow. Richmond, VA: Virginia Department of Conservation and Recreation, Natural Heritage Tech. Rep. 94-8.
- Edwards, R. and Allen, J.C. 2003. Population Monitoring and Management of Peters Mountain Mallow (*Iliamna corei*), Annual Report 2002. Charlottesville, VA: The Nature Conservancy of Virginia Tech. Rep. TNC-VA 03-01.
- Hoss, J.A., Lafon, C.W., Grissino-Mayer, H.D., Aldrich, S.R., and DeWeese, G.G. 2008. Fire history of a temperate forest with an endemic fire-dependent herb. *Physical Geography* 29:424–441.
- Keener, C.S. and Hardin, J.W. 1962. *Iliamna corei* revisited. *Castanea* 27:176–178.
- Lafon, C.W., Hoss, J.A., and Grissino-Mayer, H.D. 2005. The contemporary fire regime of the central Appalachian Mountains and its relation to climate. *Physical Geography* 26:126–146.
- Rowe, J.S. 1983. Concepts of fire effects on plant individuals and species. In: Wein, R.W. and MacLean, D.A. *The Role of Fire in Northern Circumpolar Ecosystems*. Chichester, UK: John Wiley and Sons, pp. 135–154.
- Sarvis, W. 1993. An Appalachian forest: creation of the Jefferson National Forest and its effects on the local community. *Forest and Conservation History* 37:169–178.
- Strausbaugh, P.D. and Core, E.L. 1932. *Phymosia remota*. *Rhodora* 34:142–146.

# Lion predation on the African Black Rhinoceros and its potential effect on management.



## Felix Patton

Division of Biology, Chemistry and Health Science  
Manchest Metropolitan University, Manchester, M1 5GD, UK

felixp@africaonline.co.ke

## Introduction

Apart from humans, lions are the only other main predator of rhinos. Despite this, there is very little published information describing the effect of lion predation on rhino populations. This paper brings together the references that directly indicate a problem between lions and rhinos, identifies the indications of lion predation, adds new information and considers the effect of lion predation on one population of African black rhinoceros.

Lions are generally thought to prey on medium to large ungulates within the weight range 190-550kg with buffalo, giraffe and zebra among the preferred species but will also prey on smaller species such as warthogs (Hayward & Kerley 2005). There are relatively few references which directly indicate a predation problem between lions and rhinos and this paper reviews those published, adds new information and considers the effect on population performance.

## Chronological Review of Recent Literature

Ritchie (1963) stated that young rhinos are sometimes killed by lions. He had also heard of an old bull being killed by two lions but conversely had seen two rhinos drive three lionesses off an ostrich kill.

Lions were seen moving away from rhinos which had deliberately advanced when they had become aware of the presence of the pride (Goddard 1967). Goddard further reported that, in August 1967, a sub-adult lion attempted to attack an 11 month old rhino calf. The mother was close at hand and engaged the lion. The lion bit the females hock and clawed its thigh but was gored twice by the rhino in the centre of the ribs and then in the centre of the neck followed by a blow through the base of the jaw that killed it.

Joubert & Eloff (1971) reported that rhino cows hide their small calves and were only able to find one record, in 1963, of a rhino calf being killed by a lion.

In Amboseli National Park, between 1971 and 1977, a sub-adult black rhino was killed by lions after having separated naturally from its mother (Western 1982). In another three cases the calves of poached females were too young to defend themselves and were killed by predators. Western reported that few calves less than 3 years of age survive predators if separated from their mothers.

In the Hluhluwe/Corridor/Umfolozzi complex of Natal, South Africa, Hitchins and Anderson (1983) reported that the only predators capable of killing rhino were lion, hyena and crocodile. However, there were no record-



ings of lion killing black rhinos, only white rhinos.

A freshly killed, black rhino carcass was found with an adult male lion by Elliot (1987) in Umfolozi Game Reserve. There were signs of a struggle and well defined claw and tooth marks on the neck of the rhino which had a horn length of 18-20 cm making it probably two year old. It was concluded that there was strong circumstantial evidence that the lion had killed the rhino.

In 1986/7, the Salient of the Aberdares National Park, Kenya, was home to 12 lions (Sillero-Zukiri and Gottelli, 1991). They reported that rhinos can be killed by lions even when adult up to the age of four months.

In September 1995, three rhinos aged between 3 and 4 years old, were killed by three male lions in the central area of Etosha National Park, Namibia, all by strangulation illustrated by marks on the necks (Brain et al, 1998). On two occasions the lions fed on the dead rhino immediately after killing them. On one occasion the lions left the rhino uneaten. The three rhinos killed were of similar size and age having just left their mothers but were still rela-

tively small. The attacks occurred near waterholes which other rhinos, particularly mothers and calves, frequent. That the lions took the sub-adults rather than the smaller calves suggested a substantial maternal deterrent effect.

Lions and leopards were responsible for the loss of two emaciated and diarrhoea-ridden calves, aged between four and seven months in Matusadona National Park, Zimbabwe (Matipano 2004). The mothers of the calves were also in poor body condition: one cow had diarrhoea and struggled to stand, and the other died five weeks after losing her calf.

A rhino calf was killed by lions in the Ngorongoro Crater, Tanzania in May 2000 and the mother died four months later (Maige 2001). This followed a translocation of two rhinos from the Addo Elephant Park in South Africa where there are no predators. Having calved, the mother had no knowledge of how to defend itself or the calf from lions.

In February 2002 at Lake Nakuru National Park, Kenya, a rhino calf was killed by a lion despite the mother putting up a considerable fight. This was

Female	Calf	Birth	Death	Age
Mama Ngina	Luck	11/07	04/08	5m
Classic	May	05/08	08/08	3m
Yasa	Ijumaa	07/08	01/09	5m
UID	UID	05/09	09/09	4m
Kolio	Gachembe	07/09	10/09	3m
Lamuria	Pati *	06/08	10/09	16m
* injured 02/09 at 8m, died of injury at 16m				
Classic	age 37 yrs, died 2 months after death of calf May			

**Table 1:** Black rhino deaths attributed to lions at Solio Game Reserve in 2008 and 2009. The average age of calves of under six months old killed was 4 months (n=5, range 3-5 months).

BENCHMARKS	2007 Before translocation		2007 After translocation (i)		2008		2009	
	Growth rate per annum (1)	7.10%	Mod - good	12.10%	Good - excellent	7.5% *	Mod - good	10.0% *
% cows with calves of that year (2)	23.30%	V poor - poor	40.90%	Good - excellent	27.8% **	V poor - poor	45.0% **	Good - excellent
ICI (3) (estimated months)	39.5m	Poor - mod	38.6m	Poor - mod	45.8m	V. poor	34.9m	Mod - good

**Table 2:** Key benchmarks of the development of the Solio black rhino population

(i) for details of the translocation see Patton et al (2008); the after translocation figures show an exceptional improvement due to the removal of non-breeding females to a new location to attempt to motivate breeding

(1) growth rate per annum is shown as birth rate less natural deaths (not including poached rhinos or those killed by predators)

\* the actual growth rate 2008, due to 3 deaths, is only 4.5% but 2 deaths were unnatural (lions) so have been excluded, births only growth rate is 9.0%; the actual growth rate 2009, due to 5 deaths, is only 3.0% but 5 deaths were unnatural (lions and poaching) so have been excluded. Births only growth rate is 10.0%

(2) cows with calves of that year include calves killed by predators

\*\* the actual rate in 2009 including 2 deaths by lions was 35%

(3) ICI stands for intercalving interval; of the 7 calves born – two intervals are estimates and five are more accurately known

exemplified by savage wounds on the shoulder and breast of the rhino and scratch marks on the flanks. The mother, Suzie, was considerably distressed by the incident. Despite the wounds being treated, it died a few months later.

In July 2002 at Sweetwaters Game Reserve, Laikipia, four lion cubs were observed “fighting” with a rhino at around 7pm for one hour. There appeared to be no actual contact, with the lions charging but withdrawing at the

last minute. Further problems with lion predation were experienced at Sweetwaters Game Reserve when a pride of some nine lions was considered responsible for the death of an old female rhino in poor condition in March 2007.

Plotz and Linklater (2009) reported the death of an eight month old black rhino calf from injuries caused by lions in the Hluhluwe-iMfolozi Park, South Africa in August 2008. There were puncture and tear wounds on the neck of the calf, tissue trauma, severed tail and claw marks on the anogenital region. The calf's tail was amputated at around one third of its original length.

### **New Data**

Following the introduction of a black rhino monitoring system, based on the photo-identification of individuals, at Solio Game Reserve in 2007, it was possible to subsequently analyse births and deaths (Patton et al 2007).

Lion censuses were carried out in 2008 and 2009 based on individual identification using muzzle spot patterns. Prior to this, it was believed that there were five to seven individuals. The census showed that births in 2007/8 had led to at least 14 individuals.

Table 1 shows the number of black rhino deaths attributed to lions at Solio Game Reserve in 2008 and 2009.

The key measure of population performance is the 'biological growth rate' defined as the natural increase in population size from births and mortalities as a percentage of the population size at the start of the year (Emslie and Brooks 1999) The strategy of the Kenya Wildlife Service is to manage all black rhinoceros populations in Kenya for rapid population growth while preventing overstocking and resultant sub-optimal population performance (Okita-Ouma et al 2007). The growth rate of a population as defined in the strategy is "the natural increase in a population size being the net result of additions from

breeding and losses from natural mortalities expressed as a percentage of the population size at the start of the year.

Table 2 shows the development of the black rhino population at Solio between 2007 and 2009 and considers the effect of the predation of lions on population growth rates which remains one of the best indicators of population performance (Knight 2001). Benchmarks are those proposed by Du Toit (2001).

## **Discussion**

### ***Detecting lion predation***

According to Plotz and Linklater (2009), predation on black rhinoceros juveniles might be under reported because both births and carcasses are rarely detected. New born calves are small and mothers tend to calve in dense bush remaining secretive until the calf has grown and strengthened at around 3 months. Mortality during this period is hard to detect and often, if a carcass is found, it has deteriorated to such an extent that the cause of death cannot be determined.

The carcass of Ijumaa was found mostly whole but only skin and bones remained. The unidentified calf carcass was around 50% whole with a long cut in the skin down the back which is considered typical of a lion attack. Some meat was missing from the rump and remains of the two rear legs were found nearby suggesting there had been more than one predator. The carcass of Gachembe was mostly whole with some of the meat of the rump having been eaten. This suggests that lion predation is mostly opportunistic and not due to hunger. The carcasses of Luck and May were not found which was probably due to scavengers, such as striped hyena and jackals, removing the remains. Scavenging of the small carcasses of rhino calves explains why lion predation may often go undetected.

In October 2009, the author ob-



served a female lion catch and kill a warthog about the same size as a three month old black rhino calf. The lioness eventually ate a small amount from the rump of the warthog but left the majority of the carcass untouched. On inspecting the carcass, there were no signs – such as scratches, tears or puncture marks - that the warthog had been killed by a lion apart from the bites in the rear. This outcome might also explain how lion predation might not be wholly obvious on rhino calf carcasses.

The injuries to Pati included the tail being bitten off at the base and some damage to the rear stomach which led to a swelling. No scratch marks were visible on either the calf or the mother suggesting there was no significant fight or defence from the predator. The absence of signs such as claw scratch marks or bite marks on bodies of the females, suggests that, in all cases, the calves had not been defended by their mothers. While black rhino mothers have killed lions in defence of their

calves (Goddard 1967; Owen-Smith 1988, p126-127), maternal inexperience may result in failure to defend calves. In the case of Solio rhinos, there had not previously been an observed problem with what had historically been a low density of lions of less than seven individuals (E. Parfet, personal communication) suggesting that the females were ill prepared to defend their offspring when the density increased.

The 37+ years old female Classic was distressed after its calf was killed by lions. Its health deteriorated rapidly over four months and it drowned in a river without the strength to get out. This could be considered as indirect predation by lions.

### *The Effect on Population Performance*

Poor population performance in black rhino populations has largely been attributed to density dependence (Plotz and Linklater 2009). It should be remembered that individual year figures for small populations (around 50 indi-

viduals) should be viewed with caution and three year rolling averages used. When considering the three benchmarks for 2007 before translocation (for further details see Patton et al 2008), 2008 and 2009, there is clear evidence of a general improvement following the removal of 26 poor breeding and other individuals and reducing the density of rhinos from 92 to 65. This represented a reduction in density, with Solio Game Reserve being 69 sq km, of 1.33/sq km to 0.94/sq km rising to 1.04/sq km in 2009.

Brain et al (1998) concluded that it might become increasingly important to consider the impact of large predators on breeding rhinoceros populations and that it would also be desirable to be familiar with both predator and prey and to make certain of the causes of each death in a rhinoceros so that the appropriate management actions can be made. The new data and analyses supports this conclusion.

The growth rate before predation was 2008 – 7.5% and 2009 – 10.0%. When predations is included the figures are dramatically reduced to 2008 – 4.5% and 2009 – 3.0%. The latter figures are those used officially and are significantly below the Kenya Wildlife Service target for private rhino reserves of 6.0%. If predation is taken into account, the effect of reducing the density of the rhinos is largely hidden and could even be interpreted as having had a negative effect.

The benchmarking description for the data would show the 'with-predation' growth rate in 2008 as 'poor to moderate' compared to the 'without-predation' growth rate of 'moderate to good'. Similarly, for 2009, the change is from 'poor to moderate' to 'good to excellent'. For the percentage of females with calves of that year, the benchmarking description changes from the with-predation 'moderate to good' to the without-predation 'good to excellent'. This further demonstrates that the

population performance of the rhinos was better than the "official" record including predation, which cannot be attributed directly to the breeding success of the population.

An important implication of this analysis is that the sub-optimal growth rate recorded with predation included could lead managers to conclude that the Carrying Capacity of the reserve had been reached or exceeded thereby unnecessarily undertaking costly and stressful population reduction through translocating individuals to other sanctuaries.

Despite the high density of rhinos in 2009, the population growth rate before predation was in the top benchmarking category suggesting that the breeding performance of the Solio black rhino population was good even though the actual growth rate was not. It may therefore be appropriate to add breeding performance as a benchmark rather than simply using growth rate. Where any natural calf deaths occur due to disease, malnutrition and injury, these should be included in the growth rate calculation while unnatural deaths due to poaching should be considered as predation.

### Literature Cited

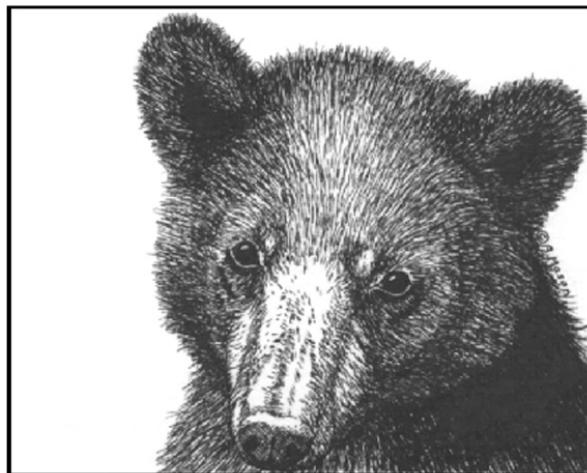
- Brain, C., Forge, O. & Erb, P. (1999) Lion predation on black rhinoceros (*Diceros bicornis*) in Etosha National Park. *African Journal of Ecology* 37, 107-109
- Du Toit RF. 2001. Benchmarks for rhino population performance in the wild. In: Emslie R, ed. SADC rhino management group (RMG) workshop on biological management to meet continental and national black rhino conservation goals: proceedings. 25-26 South African Development Community, Harare
- Elliot, W. M. (1987) Possible Predation of a Black Rhinoceros Calf by a Lion. *Lammergeyer* 38, 68
- Emslie, R. & Brooks, M., (1999) African Rhino, Status Survey and Conservation Action Plan. IUCN/SSC African rhino specialist group. IUCN Gland, Switzerland and Cambridge, UK.
- Goddard, J. (1967) Home Range, Behaviour and Recruitment of Rhinos East African Wildlife



- Journal 5 133-150
- Hayward, M. W & Kerley, I. H. (2005) Prey preferences of the lion (*Panthera leo*) Journal of Zoology 267, 309-322
- Hitchins, P. M. & Anderson, J. L. (1983) Reproduction, Population Characteristics and Management of the Black Rhinoceros in the Hluhluwe-Umfolozi Game Reserve. South African Journal of Wildlife Research 13 (3) 78-85
- Joubert, E. & Eloff (1971) Notes on the ecology and behaviour of the black rhinoceros in South West Africa. Madoqua 1, 5-53
- Knight, M. H. (2001) Current and future population performance indicators for black rhinos. In: Emslie R, ed. SADC rhino management group (RMG) workshop on biological management to meet continental and national black rhino conservation goals: proceedings. 49-56 South African Development Community, Harare
- Maige, M (2001) Deaths in Ngorongoro Crater, Tanzania. Pachyderm 30, (January-June) 96
- Matipano, G (2004) Black rhinoceros mortality in Matusadona National Park, Zimbabwe: 1992-2003 Pachyderm 36 (January-June) 109-112
- Okita-Ouma, B., Amin, R & Kock, R (2007) Conservation and Management Strategy for the Black Rhino and Management Guidelines for the White Rhino in Kenya (2007-2011) Kenya Wildlife Service
- Owen-Smith, R.H.N. (1988) Megaherbivores: The Influence of Very Large Body Size on Ecology, (eds) R.S.K. Barnes, H.J.B. Birks, E.F. Connor, J.L. Harper & R.T. Paine. Cambridge University Press, Cambridge
- Patton, F., Campbell, P & Parfet, E (2007) Establishing a monitoring system for black rhinos in the Solio Game Reserve, central Kenya Pachyderm 43, p87-95
- Patton, F., Campbell, P & Parfet, E (2008) Biological management of the high density black rhino population in Solio Game Reserve, central Kenya. Pachyderm 44, p72-79
- Plotz, R. D & Linklater, W. L. (2009) Black rhinoceros (*Diceros bicornis*) calf succumbs after lion predation attempt: implications for conservation management African Zoology 44 (2): 283-287
- Ritchie, A. T. A., (1963) The Black Rhinoceros (*Diceros bicornis* L.) East African Wildlife Journal 1, 54-62
- Sillero-Zubiri, C. & Gottelli, D. (1991) Threat to Aberdares Rhinos: Predation versus Poaching Pachyderm 14, 37-38
- Western, D (1982) Patterns of depletion in a Kenya rhino population and the conservation implications Biological Conservation 24 147-156

## FOCUS ON NATURE<sup>®</sup>

*Insight into the lives of animals*



With food availability being so important to the **LOUISIANA BLACK BEAR** (*Ursus americanus luteolus*), the sow knows she needs to quickly build up her fat stores. If they're not sufficient, the embryos, fertilized weeks ago, will not implant. Of the three cubs she gave birth to a couple years ago, two survived the perils of childhood and went off to establish their own territories. The sow taught them how to find edible tubers, herbs, nuts, fruits and insect larvae. When these natural foods are scarce, even over a large range, the bear's keen sense of smell oftentimes leads her to people food and scraps. Quite intelligent and secretive, she is careful to not be seen and makes sure she can dash back to the surrounding hardwood forests and swamps. This subspecies of American black bear is supposedly the original "Teddy bear". *Artwork and text by Rochelle Mason*  
*Copyright 2000-2011 www.Rmasonfinearts.com (808)769-0756*



School of Natural Resources and Environment  
The University of Michigan  
440 Church Street  
Ann Arbor, MI 48109-1041

Non-Profit  
Organization  
U.S. POSTAGE  
PAID  
Ann Arbor, MI  
Permit 144

---

## ***Subscribe to the UPDATE today!***

In its 23 years of publication, the Endangered Species UPDATE, published by the School of Natural Resources and Environment at the University of Michigan, has established itself as the primary forum for government agencies, conservation organizations, private consulting and law firms, zoos, museums, educational institutions, and others to exchange ideas and information on species conservation issues.

Subscription rates are:

institutions	\$78
individual subscriptions	\$33
student/senior (65+)	\$25
address outside the US	add \$5
electronic subscription	\$20

To subscribe, make check payable to the University of Michigan and mail to:

Endangered Species UPDATE  
School of Natural Resources and Environment  
The University of Michigan  
440 Church Street  
Ann Arbor, MI 48109-1041

Visit our website: <http://www.umich.edu/~esupdate> !

