Yellowstone Grizzly Bear Investigations 2004

Photo by Chad Dickinson, IGBST

Annual Report of the Interagency Grizzly Bear Study Team

YELLOWSTONE GRIZZLY BEAR INVESTIGATIONS

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2004

U.S. Geological Survey
Wyoming Game and Fish Department
National Park Service
U.S. Fish and Wildlife Service
Montana Fish, Wildlife and Parks
U.S. Forest Service
Idaho Department of Fish and Game
Montana State University

Charles C. Schwartz, Mark A. Haroldson, and Karrie West, Editors

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INTRODUCTION *(Charles C. Schwartz, Interagency Grizzly Bear Study Team; and David Moody, Wyoming Game and Fish Department)*

This Report

The contents of this Annual Report summarize results of monitoring and research from the 2004 field season. The report also contains a summary of nuisance grizzly bear (*Ursus arctos horribilis*) management actions.

The study team continues to work on issues associated with counts of unduplicated females with cubs-of-the-year (COY). These counts are used to establish a minimum population size, which is then used to establish mortality thresholds for the Recovery Plan (U.S. Fish and Wildlife Service [USFWS] 1993). A computer program that defines the rule set used by Knight et al. (1995) to differentiate unique family groups was completed in spring 2005. We will use an improved version of this model to verify the accuracy of the rules using known bears and their telemetry locations in test runs. We hope to have this work complete by the end of 2005.

The grizzly bear recovery plan (USFWS 1993) established mortality quotas at 4% of the minimum population estimate derived from female with COY data and no more than 30% of the 4% (1.2%) could be female bears. Simulation modeling (Harris 1984) established sustainable mortality at around 6% of the population. We used the latest information on reproduction and survival to estimate population trajectory in the same simulation model originally used by Harris. A Wildlife Monograph has been accepted for publication and should be available by summer 2005.

Our project addressing the potential application of stable isotopes and trace elements to quantify consumption rates of whitebark pine (*Pinus albicaulis*) and cutthroat trout (*Oncorhynchus clarki*) by grizzly bears was completed. Our manuscript on consumption rates of whitebark pine has been published (Canadian Journal of Zoology 81:763-770). The manuscript on fish consumption rates was also accepted and is published in the Canadian Journal of Zoology 82:493-501. Both can be found on the Interagency Grizzly Bear Study Team (IGBST) website http://www.nrmsc.usgs.gov/research/igbst-home.htm.

We began a new study in Grand Teton National Park evaluating habitat use both temporally and spatially between grizzly and black (*Ursus americanus*) bears. We will employ a new form of Global Positioning System (GPS) technology that incorporates a spread spectrum communication system. Spread spectrum allows for transfer of stored GPS locations from the collar to a remote receiving station. Results of our first year's field season are summarized in this report.

Whitebark pine (WBP) has been identified as one of the important fall foods of the Yellowstone grizzly bear. Previous efforts to map the distribution of WBP were for the Cumulative Effects Model. Consequently the only coverage of WBP distribution was for the grizzly bear Recovery Zone. We were successful in getting financial support through the U.S. Geological Survey Land Remote Sensing Program and Interdisciplinary Science Support Activities Project to create an ecosystem-wide map of the distribution of WBP. The results of that project are reported in Appendix A. The study team annually estimates WBP cone production on a series of transects. That information is reported annually in our reports. Concern over the long-term health of WBP prompted us to investigate the usefulness of cone counts as an indirect index of WBP health. Results of this analysis (Appendix B) indicated that cone production is too variable to serve this purpose. Consequently, we partnered with several

other agencies and embarked on a program to develop a long-term monitoring program directed specifically at WBP health in the Greater Yellowstone Ecosystem (GYE). Our team (Greater Yellowstone Whitebark Pine Monitoring Working Group) was successful in obtaining funds to develop and implement a WBP health monitoring program. Results of our first years work are presented in Appendix C. We also successfully competed for funds in 2005 and will continue to collect information on WBP health.

Army cutworm moths (*Euxoa auxiliaris*) are also a very important food for a segment of the GYE grizzly bear population. Hillary Robison, graduate student at University of Nevada, Reno, is nearing completion of her program. In this report, we post her annual work summary, and abstracts of her most recently submitted publications. These include one on the levels of pesticides in cutworm moths and their potential affect on grizzly bears (Appendix D), a spatial analysis to identify army cutworm moth habitat (Appendix E), and the results of a preliminary analysis of pollen grains on the mouth parts of moths (Appendix F) to help identify which plant species are commonly fed upon.

Other study team members have also been working on various aspects of grizzly bear science. Study team member Kerry Gunther hosted a workshop on habituated grizzly bears in North America. A copy of the abstract of that report can be found in Appendix G. Additionally, Kerry Gunther and Doug Smith, wolf researcher in Yellowstone National Park (YNP), reported on the interactions between gray wolves (*Canis lupus*) and female grizzly bears with young. They report that of 15 interactions between these 2 carnivores, 8 involved females with COY. They observed 2 incidents where cubs were killed by wolves at ungulate carcasses (Appendix H).

The annual reports of the IGBST summarize annual data collection. Because additional information can be obtained after publication, <u>data summaries are subject to change</u>. For that reason, data analyses and summaries presented in this report supersede all previously published data. The study area and sampling techniques are reported by Blanchard (1985), Mattson et al. (1991*a*), and Haroldson et al. (1998).

History and Purpose of the Study Team

It was recognized as early as 1973, that in order to understand the dynamics of grizzly bears throughout the GYE, there was a need for a centralized research group responsible for collecting, managing, analyzing, and distributing information. To meet this need, agencies formed the IGBST, a cooperative effort among the U.S. Geological Survey, National Park Service, U.S. Forest Service, USFWS, and the States of Idaho, Montana, and Wyoming. The responsibilities of the IGBST are to: (1) conduct both short- and long-term research projects addressing information needs for bear management; (2) monitor the bear population, including status and trend, numbers, reproduction, and mortality; (3) monitor grizzly bear habitats, foods, and impacts of humans; and (4) provide technical support to agencies and other groups responsible for the immediate and long-term management of grizzly bears in the GYE. Additional details can be obtained at our web site (http://www.nrmsc.usgs.gov/research/igbst-home.htm).

Quantitative data on grizzly bear abundance, distribution, survival, mortality, nuisance activity, and bear foods are critical to formulating management strategies and decisions. Moreover, this information is necessary to evaluate the recovery process. The IGBST

coordinates data collection and analysis on an ecosystem scale, prevents overlap of effort, and pools limited economic and personnel resources.

Previous Research

Some of the earliest research on grizzlies within YNP was conducted by John and Frank Craighead. The book, "The Grizzly Bears of Yellowstone" provides a detailed summary of this early research (Craighead et al. 1995). With the closing of open-pit garbage dumps and cessation of the ungulate reduction program in YNP in 1967, bear demographics (Knight and Eberhardt 1985), food habits (Mattson et al. 1991*a*), and growth patterns (Blanchard 1987) for grizzly bears changed. Since 1975, the IGBST has produced annual reports and numerous scientific publications (for a complete list visit our web page http://www.nrmsc.usgs.gov/research/igbst-home.htm) summarizing monitoring and research efforts within the GYE. As a result, we know much about the historic distribution of grizzly bears within the GYE (Basile 1982, Blanchard et al. 1992), movement patterns (Blanchard and Knight 1991), food habits (Mattson et al. 1991*a*), habitat use (Knight et al. 1984), and population dynamics (Knight and Eberhardt 1985, Eberhardt et al. 1994, Eberhardt 1995). Nevertheless, monitoring and updating continues so that status can be reevaluated annually.

This report truly represents a "study team" approach. Many individuals contributed either directly or indirectly to its preparation. To that end, we have identified author(s). We also wish to thank Chad Dickinson, Craig Whitman, Jeremiah Smith, Josh Brown, Matt Neuman, Mark Packila, Meghan Riley, Janissa Balcomb, Bryn Karabensch, Andrew Sorensen, Sabrina Mueller, Doug Blanton, Rich Baerwald, Susan Chin, Tyler Coleman, Colette Daigle-Berg, Bonnie Gaffney, Sarah Dewey, Kan Dhillon, Leslie Frattaroli, Gina Garrett, Rick Guerrieri, Bill Kraegle, Karen Loveless, Mary McKinney, Kathy McFarland, Gina Poulson, Lori Roberts, Doug Smith, Sue Wolff, Kurt Alt, Neil Anderson, Sam Shepard, Shawn Stewart, Scott Becker, Brian DeBolt, Craig Sax, Gary Brown, Max Black, Doug Brimeyer, Dave Edmunds, Tim Fagan, Arian Hampel, Carey Hendrix, Dave Hyde, Andy Johnson, Jordan Kraft, Jerry Longobardi, Doug McWhirter, Eric Shorma, Mike Wegan, Shane Liss, Gregg Losinski, Mark Hinschberger, Brian Aber, Kim Barber, Bill Davis, Pilar Delmolino, Connie King, Linette Otto, Mark Petroni, Andy Pils, Jesse Rawson, Ron Wiseman, Rich Hyatt, Gary Lust, Claude Tyrrel, Roger Stradley, Steve Ard, Dave Stinson, John Martin, Bill Ard, Harley Leach, Dan Stahler, Dave Stradley, Janice Stroud, Greg Anderson, Bart Kroger, Dean Clause, Steve Kilpatrick, Kim Keating, and Steve Cherry for their contributions to data collection, analysis, and other phases of the study. Without the collection efforts of many, the information contained within this report would not be available.

RESULTS AND DISCUSSION

Bear Monitoring and Population Trend

Marked Animals *(Mark A. Haroldson and Chad Dickinson, Interagency Grizzly Bear Study Team; and Dan Bjornlie, Wyoming Game and Fish Department)*

During the 2004 field season, 58 individual grizzly bears were captured on 67 occasions (Table 1), including 22 females (14 adult) and 36 males (25 adult). Thirty-nine individuals were new bears not previously marked.

We conducted research trapping efforts for 904 trap days (1 trap day = 1 trap set for 1 day) in 10 Bear Management units (BMUs) within the Grizzly Bear Recovery Zone (USFWS 1993) and adjacent 10-mile perimeter area. Research trapping efforts were also conducted outside the 10-mile perimeter in Montana and Wyoming. During research trapping operations we captured 29 individual grizzly bears 38 times for a trapping success rate of 1 grizzly capture every 23.8 trap days.

There were 29 management captures of 29 individual bears in the GYE during 2004 (Tables 1 and 2), including 14 females (8 adult) and 15 males (8 adults). Twenty bears (10 females, 10 males), were relocated due to conflicts situations (Table 1). Two male bears (both subadults) were not known to be bears involved in nuisance activity at the time of capture and were released on site. Seven grizzly bears (4 females, 3 males) captured at management trap sites were removed from the population as a result of conflicts with humans.

We radio-monitored 78 individual grizzly bears during the 2004 field season, including 26 adult females (Tables 2 and 3). Fifty-one grizzly bears entered their winter dens wearing active transmitters in the GYE, 2 of these bears are considered missing. Since 1975, 479 individual grizzly bears have been radiomarked.

Table 1. Grizzly bears captured in the Greater Yellowstone Ecosystem during 2004.

Bear	Sex	Age	Date	General location[a]	Capture type	Release site	Trapper/Handler[b]
G92	male	adult	04/12/04	Clark's Fork, Pr-WY	management	removal	WYGF
452	male	subadult	05/20/04	Cedar Cr, Pr-MT	management	Tepee Cr, GNF	MTFWP
453	male	subadult	05/24/04	Pebble Cr, YNP	management	on site	YNP/IGBST
454	male	adult	06/02/04	Mormon Cr, SNF	research	on site	IGBST
243	male	adult	06/02/04	Mormon Cr, SNF	research	on site	IGBST
455	male	adult	06/11/04	Mormon Cr, SNF	research	on site	WYGF
456	male	adult	06/14/04	Long Cr, SNF	research	on site	WYGF
G93	male	yearling	06/23/04	Deer Cr, Pr-WY	management	on site	WYGF
348	male	adult	06/26/04	Deer Cr, Pr-WY	management	removal	WYGF
451	female	subadult	06/29/04	N Fork Shoshone, SNF	management	removal	WYGF
399	female	adult	07/15/04	Pilgrim Cr, GTNP	research	on site	IGBST
			10/02/04	Colter Bay, GTNP	research	on site	IGBST
457	male	adult	07/28/04	Waynes Cr, SNF	research	on site	WYGF
458	female	subadult	08/04/04	Lodgepole Cr, Pr-WY	management	Boone Cr, CTNF	WYGF
459	male	adult	08/06/04	E Fork Wind, SNF	research	on site	WYGF
372	male	adult	08/09/04	Paint Cr, SNF	management	removal	WYGF
460	male	subadult	08/11/04	Lizard Cr, GTNP	research	on site	IGBST
			08/17/04	Lizard Cr, GTNP	research	on site	IGBST
461	female	adult	08/13/04	Lizard Cr, GTNP	research	on site	IGBST
			09/23/04	Lizard Cr, GTNP	research	on site	IGBST
462	male	subadult	08/18/04	Rose Cr, Pr-WY	management	Lost Lake, BTNF	WYGF
273	male	adult	08/19/04	Reef Cr, SNF	research	on site	IGBST
463	male	adult	08/27/04	Gas Cr, SNF	research	on site	IGBST
464	male	adult	08/27/04	Wagon Cr, BTNF	management	Mormon Cr, SNF	WYGF
365	female	adult	08/28/04	Reef Cr, SNF	research	on site	IGBST
337	female	adult	08/30/04	Reef Cr, SNF	research	on site	IGBST
386	female	adult	08/30/04	Paint Cr, SNF	management	Burroughs Cr, SNF	WYGF
377	male	adult	08/31/04	Raspberry Cr, BTNF	management	Mormon Cr, SNF	WYGF
465	male	adult	09/02/04	S Fork Shoshone, Pr-WY	management	Lost Lake, BTNF	WYGF
305	female	adult	09/02/04	Big Cr, Pr-WY	management	removal	WYGF
G94	male	yearling	09/02/04	Big Cr, Pr-WY	management	Fox Cr, SNF	WYGF
G95	female	yearling	09/02/04	Big Cr, Pr-WY	management	Fox Cr, SNF	WYGF
G96	female	subadult	09/07/04	Pebble Cr, YNP	management	removal	YNP
466	male	adult	09/07/04	Mill Cr, Pr-MT	management	Trapper Cr, GNF	WS/MTFWP
467	female	adult	09/09/04	Togwotee Lodge, BTNF	management	Sunlight Cr, SNF	WYGF
468	male	subadult	09/11/04	Waterfalls Canyon, GTNP	research	on site	IGBST
469	male	adult	09/11/04	Elk Tongue Cr, YNP	research	on site	IGBST

Table 1. Continued.

Bear	Sex	Age	Date	General location[a]	Capture type	Release site	Trapper/Handler[b]
470	male	adult	09/12/04	Buffalo Plateau, YNP	research	on site	IGBST
			09/16/04	Buffalo Plateau, YNP	research	on site	IGBST
471	male	adult	09/13/04	Crow Cr, BTNF	management	Mormon Cr, SNF	WYGF
367	female	adult	09/17/04	Sage Cr, Pr-WY	management	Blackrock Cr, BTNF	WYGF
472	female	adult	09/19/04	Sage Cr, Pr-WY	management	Jackson Crk CTNF	WYGF
473	male	subadult	09/22/04	S Fork Shoshone, Pr-WY	management	Horse Cr, SNF	WYGF
474	female	adult	09/25/04	Lizard Cr, GTNP	research	on site	IGBST
475	male	subadult	09/26/04	Jasper Cr, YNP	research	on site	IGBST
			10/06/04	Antelope Cr, YNP	research	on site	IGBST
			10/10/04	Jasper Crk, YNP	research	on site	IGBST
476	female	adult	09/26/04	S Fork Shoshone, Pr-WY	management	Parque Cr, SNF	WYGF
G97	female	subadult	09/28/04	Eagle Cr, SNF	management	removal	WYGF
477	male	adult	09/29/04	Pilgrim Cr, GTNP	research	on site	IGBST
398	male	adult	10/01/04	Pilgrim Cr, GTNP	research	on site	IGBST
478	female	adult	10/02/04	S Fork Shoshone, Pr-WY	management	Hominy Cr, CTNF	WYGF
G98	female	yearling	10/02/04	S Fork Shoshone, Pr-WY	management	Hominy Cr, CTNF	WYGF
G99	male	yearling	10/02/04	S Fork Shoshone, Pr-WY	management	Hominy Cr, CTNF	WYGF
448	female	subadult	10/06/04	Arnica Cr, YNP	research	on site	IGBST
			10/07/04	Arnica Cr, YNP	research	on site	IGBST
479	male	adult	10/06/04	Stephens Cr, YNP	research	on site	IGBST
321	female	adult	10/08/04	Cascade Cr, YNP	research	on site	IGBST
			10/10/04	Cascade Cr, YNP	research	on site	IGBST
480	male	adult	10/08/04	Cascade Cr, YNP	research	on site	IGBST
			10/10/04	Cascade Cr, YNP	research	on site	IGBST
211	male	adult	10/08/04	Antelope Cr, YNP	research	on site	IGBST
441	male	subadult	10/09/04	Jasper Cr, YNP	research	on site	IGBST
481	female	subadult	10/10/04	Arnica Cr, YNP	research	on site	IGBST
155	male	adult	10/11/04	Antelope Cr, YNP	research	on site	IGBST
482	female	adult	10/11/04	Eagle Cr, SNF	management	Lost Lake, BTNF	WYGF
345	male	adult	10/12/04	Stephens Cr, YNP	research	on site	IGBST

[a] BTNF = Bridger-Teton National Forest, CTNF = Caribou-Targhee National Forest, GNF = Gallatin National Forest, GTNP = Grand Teton National Park, SNF = Shoshone National Forest, YNP = Yellowstone National Park, Pr = private.
[b] IGBST = Interagency Grizzly Bear Study Team, USGS; MTFWP = Montana Fish, Wildlife and Parks; WS = Wildlife Services/Animal and Plant Health Inspection Service (APHIS); WYGF = Wyoming Game and Fish.

Table 2. Annual record of grizzly bears monitored, captured, and transported in the Greater Yellowstone Ecosystem since 1980.

| Year | Number monitored | Individuals trapped | Total captures | | Transports |
			Research	Management	
1980	34	28	32	0	0
1981	43	36	30	35	31
1982	46	30	27	25	17
1983	26	14	0	18	13
1984	35	33	20	22	16
1985	21	4	0	5	2
1986	29	36	19	31	19
1987	30	21	15	10	8
1988	46	36	23	21	15
1989	40	15	14	3	3
1990	35	15	4	13	9
1991	42	27	28	3	4
1992	41	16	15	1	0
1993	43	21	13	8	6
1994	60	43	23	31	28
1995	71	39	26	28	22
1996	76	36	25	15	10
1997	70	24	20	8	6
1998	58	35	32	8	5
1999	65	42	31	16	13
2000	84	54	38	27	12
2001	82	63	41	32	15
2002	81	54	50	22	15
2003	80	44	40	14	11
2004	78	58	38	29	20

Table 3. Grizzly bears radio monitored in the Greater Yellowstone Ecosystem during 2004.

Bear	Sex	Age	Offspring[a]	Monitored Out of den	Into den	Current Status
155	M	Adult		no	yes	Active
188	F	Adult	1 2-year-old	yes	no	Cast
193	F	Adult	2 COY	yes	no	Cast
196	F	Adult	2 yearlings	yes	no	Cast
211	M	Adult		no	yes	Active
213	F	Adult	2 COY, lost 1 or both	yes	no	Cast
214	F	Adult	3 COY	yes	yes	Active
243	M	Adult		no	no	Cast
267	F	Adult	Unknown	yes	yes	Active
273	M	Adult		no	yes	Active
295	F	Adult	2 yearlings	yes	yes	Active
321	F	Adult	2 yearlings	no	yes	Active
337	F	Adult	2 COY	no	yes	Active
345	M	Adult		no	yes	Active
349	F	Adult	None	yes	yes	Active
356	M	Adult		no	no	Probable failed battery
365	F	Adult	None	no	yes	Active
372	M	Adult		yes	no	Dead
377	M	Adult		no	yes	Active
386	F	Adult	None	no	yes	Active
399	F	Adult	1 COY, lost	yes	yes	Active
402	F	Adult	Unknown	yes	yes	Active
406	M	Adult		yes	no	Unresolved[b]
408	M	Subadult		yes	no	Cast
412	F	Adult	Unknown	yes	yes	Active
415	M	Subadult		yes	no	Cast
423	F	Adult	1 COY	yes	yes	Active
427	M	Subadult		yes	yes	Active
428	F	Subadult		yes	yes	Active
429	M	Adult		yes	no	Cast
430	M	Subadult		yes	no	Missing
431	M	Subadult		yes	no	Dead
432	M	Adult		yes	no	Cast
433	M	Adult		yes	no	Cast
436	M	Adult		yes	yes	Active
437	M	Adult		yes	yes	Active
439	F	Adult	Unknown	yes	yes	Active
440	M	Adult		yes	no	Cast

Table 3. Continued.

Bear	Sex	Age	Offspring[a]	Monitored Out of den	Into den	Current Status
441	M	Subadult		yes	yes	Active
442	M	Adult		yes	no	Cast
443	M	Adult		yes	no	Cast
445	M	Adult		yes	no	Cast
447	F	Adult	1 2-year-old	yes	yes	Cast
448	F	Subadult		no	yes	Active
449	M	Adult		yes	no	Failed transmitter
450	M	Adult		yes	no	Cast
451	F	Subadult		yes	no	Dead
452	M	Subadult		no	yes	Active
453	M	Subadult		no	yes	Active
454	M	Adult		no	no	Cast
455	M	Adult		no	no	Cast
456	M	Adult		no	no	Dead
457	M	Adult		no	no	Missing
458	F	Subadult		no	yes	Active
459	M	Adult		no	no	Cast
460	M	Subadult		no	yes	Active
461	F	Adult	None	no	yes	Active
462	M	Subadult		no	yes	Active
463	M	Adult		no	yes	Active
464	M	Adult		no	yes	Active
465	M	Adult		no	yes	Active
466	M	Adult		no	no	Cast
467	F	Adult	None	no	yes	Active
468	M	Subadult		no	yes	Active
469	M	Adult		no	yes	Active
470	M	Adult		no	yes	Active
471	M	Adult		no	yes	Active
472	F	Adult	None	no	yes	Active
473	M	Subadult		no	yes	Active
474	F	Adult	None	no	yes	Active
475	M	Subadult		no	yes	Active
476	F	Adult	None	no	yes	Active
477	M	Adult		no	yes	Active
478	F	Adult	2 yearlings	no	yes	Active
479	M	Adult		no	yes	Active

Table 3. Continued.

Bear	Sex	Age	Offspring[a]	Monitored Out of den	Into den	Current status
480	M	Adult		no	yes	Active
481	F	Subadult		no	yes	Active
482	F	Adult	None	no	yes	Active

[a] COY = cub-of-the-year.
[b] Transmitter was not retrieved in 2004, site will be visited as soon as possible in 2005 to determine status.

Unduplicated Females *(Mark A. Haroldson, Interagency Grizzly Bear Study Team)*

Forty-nine unduplicated females with COY were identified using the method described by Knight et al. (1995) in the GYE during 2004 (Fig. 1). Three of the 49 females were observed further than 10 miles from the Recovery Zone (1 in Wyoming, 2 in Montana). Under the rules established by the Grizzly Bear Recovery Plan (Appendix F of USFWS 1993), 46 females will be used in calculation of the minimum population estimates and mortality thresholds in the Yellowstone Grizzly Bear Recovery Zone for the year 2004.
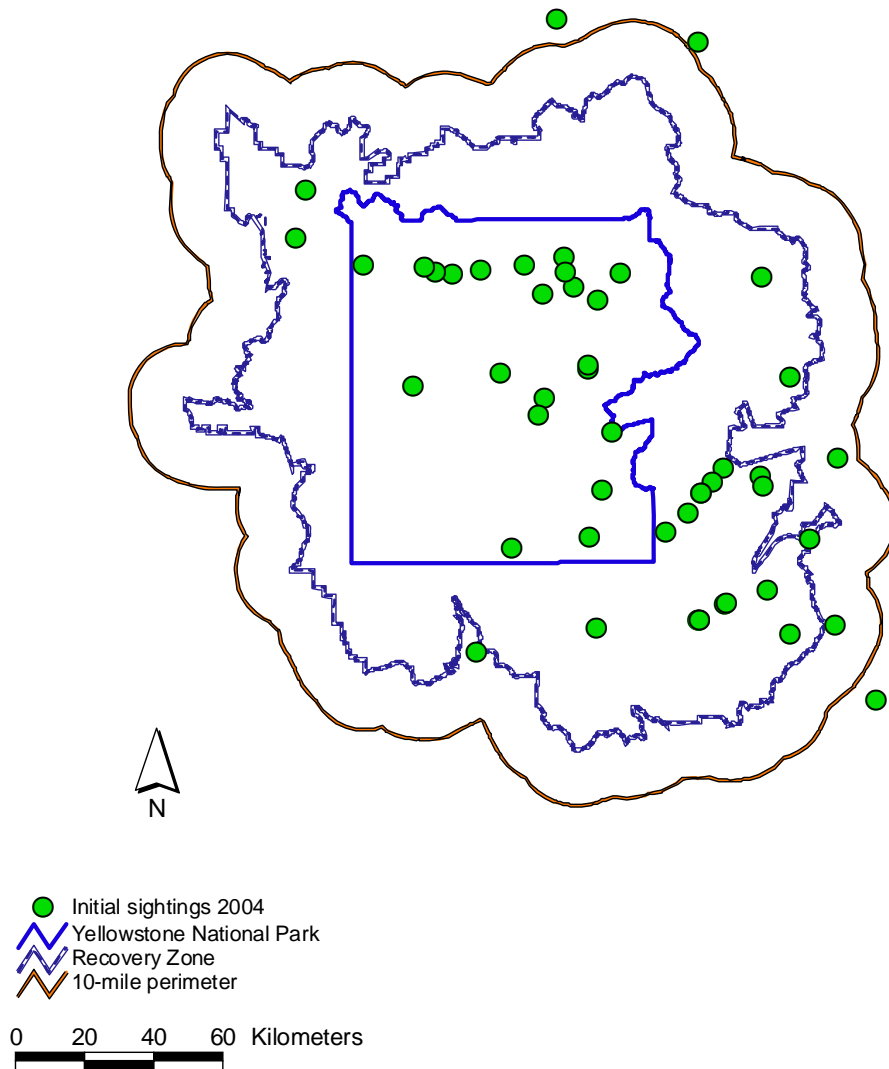
Fig. 1. Distribution of initial sightings for 49 unduplicated females with cubs-of-the-year identified in the Greater Yellowstone Ecosystem during 2004.

We documented 223 verified sightings of females with COY during 2004 (Fig. 2). This was a 271% increase over the number of sightings obtained in 2003 ($n = 60$). A likely explanation for the large increase in number of sightings is that more females were available for breeding during 2003. With the good whitebark pine cone crop during 2003, these females bred and produced cubs. This is support by the relatively high rate of grizzly bear observations per hour of observation, and the increased rate of females with COY observed per hour of flight (Fig. 3) over that observed during 2003. Most (77%) of the sightings occurred in YNP, and most observations (66%) were attributable to ground observers (Table 4). The correlation between the number of sightings obtained and the number of unduplicated females with COY identified annually (Fig. 4) remains strong (Pearsons $r = 0.91$).
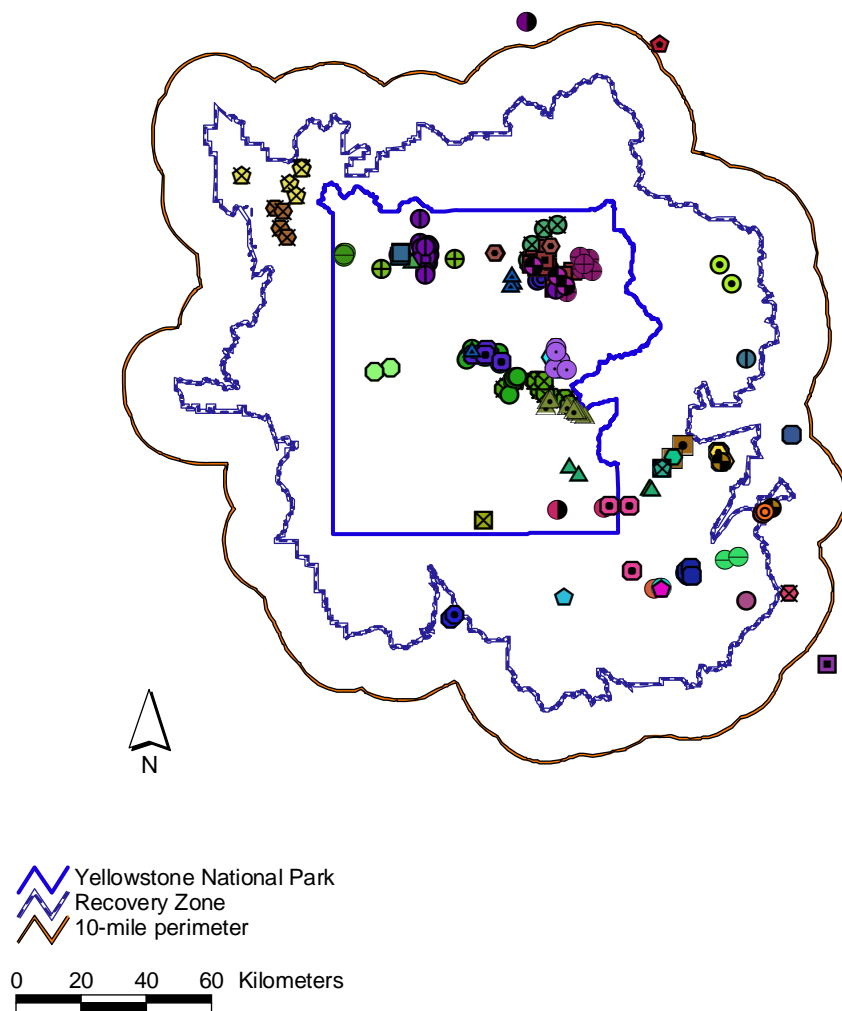

Fig. 2. Distribution of 223 observations of 49 unduplicated females (indicated by unique symbols) with cubs-of-the-year during 2004.
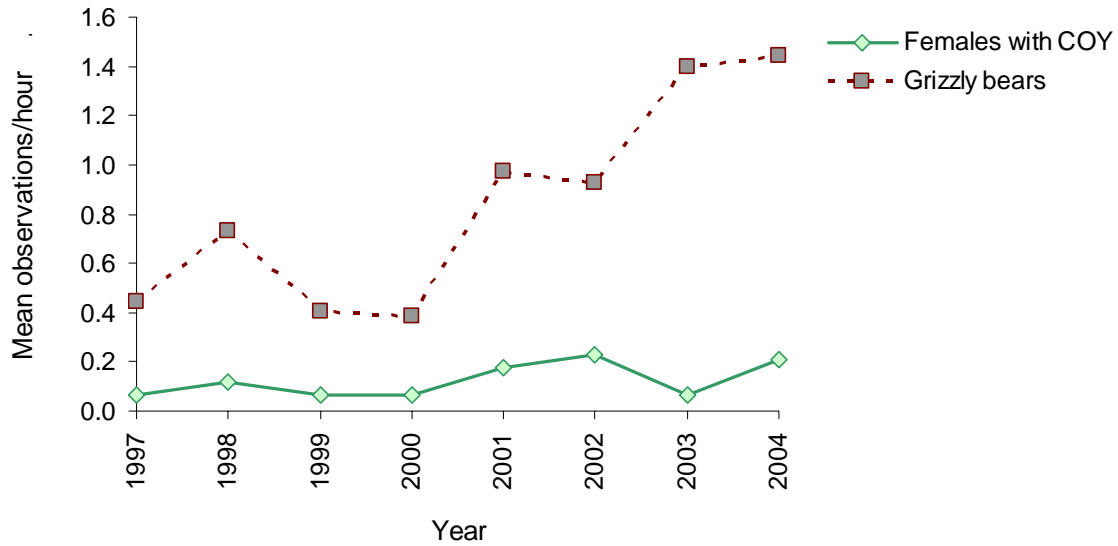
Fig. 3. Average observations/hour for total number of unmarked grizzly bears, and females with cubs-of-the-year (COY), in non-moth Bear Management Units within the Recovery Zone, 1997-2004.

Table 4. Method of observation for sightings of unduplicated females with cubs-of-the-year during 2004.

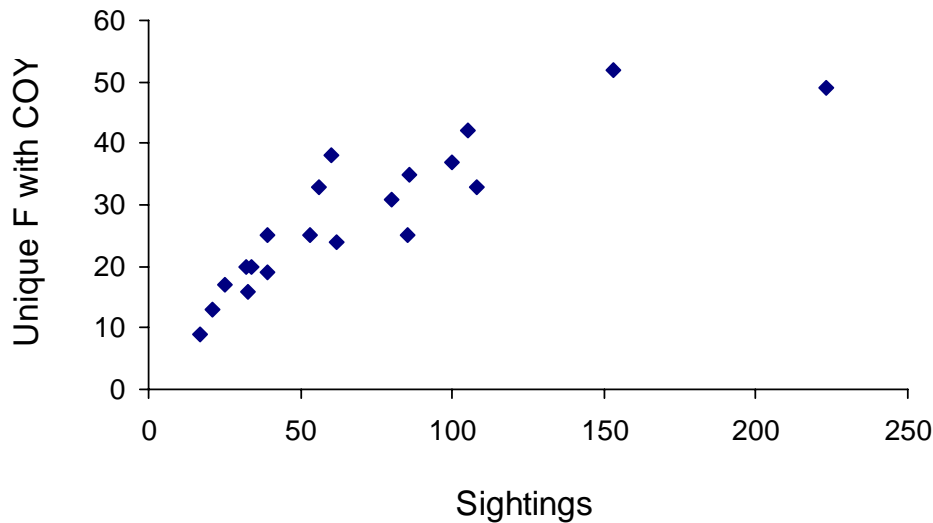| Method of observation	Frequency	Percent	Cumulative percent
Fixed wing - other researcher	11	4.9	4.9
Fixed wing - observation	49	22.0	26.9
Fixed wing - radio flight	12	5.4	32.3
Ground sighting	148	66.4	98.7
Helicopter - other research	2	0.9	99.6
Trap	1	0.4	100.0
Total	223	100.0	

Fig. 4. Relationship between number of sightings and number of unduplicated females (F) with cubs-of-the-year (COY) identified annually during 1985-2004.

Total number of COY observed during initial sighting of unique females was 96 (Table 5). Mean litter size was 1.96 (Table 5). There were 14 single cub litters, 23 litters of twins, and 12 litters of triplets seen during initial observations. The current 6-year average (1999-2004) for counts of unduplicated females with COY within the Recovery Zone and the 10-mile perimeter is 40 (Table 5). The 6-year average for total number of COY and average litter size observed at initial sighting were 77 and 1.9, respectively (Table 5).

Current methodology to determine number of unduplicated females with COY provides a minimum count (Knight et al. 1995). Keating et al. (2002) investigated 7 methods to estimate the total numbers of females with cubs annually using sighting frequencies of randomly observed bears and recommended the second order sample coverage estimator ($\hat{N}_{SC2}$) of Lee and Chao (1994). The Conservation Strategy for the Grizzly Bear in the Greater Yellowstone Area (USFWS 2003) proposes to estimate total grizzly bear population size and set mortality thresholds using estimates of total number of females with COY produced by this methodology. During 2004, we estimated 73 unduplicated females with COY in the GYE using $\hat{N}_{SC2}$ (Table 6). We met the minimum sample size ($n/\hat{N}_{SC2} \geq 1$, Table 6) recommended by Keating et al. (2002), but our estimate for the number of females with COY during 2004 was biased (about 20% high) because the estimated coefficient of variation ($\hat{\gamma}$) among sighting probabilities for individual animals was high (Table 6).

Table 5. Number of unduplicated females with cubs-of-the-year (COY), number of COY, and average litter size at initial observation for the years 1973-2004 in the Greater Yellowstone Ecosystem (GYE). Six-year running averages were calculated using only unduplicated females with COY observed in the Recovery Zone and 10-mile perimeter.

| | GYE | | | Recovery Zone and 10-mile perimeter | | |
| | | | Mean litter | 6-year running averages | | |
Year	Females	COY	size	Females	COY	Litter size
1973	14	26	1.9			
1974	15	26	1.7			
1975	4	6	1.5			
1976	17	32	1.9			
1977	13	25	1.9			
1978	9	19	2.1	12	22	1.8
1979	13	29	2.2	12	23	1.9
1980	12	23	1.9	11	22	1.9
1981	13	24	1.8	13	25	2.0
1982	11	20	1.8	12	23	2.0
1983	13	22	1.7	12	23	1.9
1984	17	31	1.8	13	25	1.9
1985	9	16	1.8	13	23	1.8
1986	25	48	1.9	15	27	1.8
1987	13	29	2.2	15	28	1.9
1988	19	41	2.2	16	31	1.9
1989[a]	16	29	1.8	16	32	1.9
1990	25	58	2.3	18	36	2.0
1991[b]	24	43	1.9	20	41	2.0
1992	25	60	2.4	20	43	2.1
1993[a]	20	41	2.1	21	45	2.1
1994	20	47	2.4	21	46	2.1
1995	17	37	2.2	22	47	2.2
1996	33	72	2.2	23	50	2.2
1997	31	62	2.0	24	53	2.2
1998	35	70	2.0	26	55	2.1
1999[a]	33	63	1.9	28	58	2.1
2000[c]	37	72	2.0	31	62	2.0
2001	42	78	1.9	35	69	2.0
2002[c]	52	102	2.0	38	73	1.9
2003[d]	38	75	2.0	38	74	1.9
2004[d]	49	96	2.0	40	77	1.9

[a] One female with COY was observed outside the 10-mile perimeter.
[b] One female with unknown number of COY. Average litter size was calculated using 23 females.
[c] Two females with COY were observed outside the 10-mile perimeter.
[d] Three females with COY were observed outside the 10-mile perimeter.

Table 6. Estimates of annual numbers ($\hat{N}_{Obs}$) of females with cubs-of-the-year (F_{Cub}) in the Greater Yellowstone Ecosystem grizzly bear population, 1986–2004. $\hat{N}_{Obs}$ gives the number of unique F_{Cub} differentiated, including those located using radiotelemetry; m gives the number of unique F_{Cub} observed using random sightings only; and $\hat{N}_{SC2}$ gives the second-order sample coverage estimates, per Lee and Chao (1994; Eqs. 3–5). Lower, 1-tailed confidence bounds are for $\hat{N}_{SC2}$ and were calculated using Efron and Tibshirani's (1993) percentile bootstrap method. Also included are annual estimates of relative sample size ($n / \hat{N}_{SC2}$, where n is the total number of observations of F_{Cub}) and of the coefficient of variation among sighting probabilities for individual animals ($\hat{\gamma}$, Eq. 5). Estimates differ in some years from those in Table 5 of Keating et al. (2002) because values presented here are for the entire GYE, not the just the Recovery Zone plus 10-mile perimeter.

| Year | $\hat{N}_{Obs}$ | m | $\hat{N}_{SC2}$ | Lower 1-tailed confidence bounds | | | | $n / \hat{N}_{SC2}$ | $\hat{\gamma}$ |
				70%	80%	90%	95%		
1986	25	24	31.9	28.3	26.9	25.3	23.7	2.6	0.9
1987	13	12	19.5	17.0	15.4	13.6	11.8	1.0	0.4
1988	19	17	21.5	20.1	19.1	17.7	16.7	1.7	0.3
1989	16	14	23.4	19.3	17.3	15.4	14.0	1.2	0.7
1990	25	22	25.5	24.4	23.6	22.2	21.3	1.9	0.0
1991	24	24	34.5	31.2	29.2	26.6	25.1	1.8	0.6
1992	25	23	47.6	39.9	36.3	32.5	29.2	0.8	0.6
1993	20	18	23.9	22.0	20.8	19.6	18.0	1.3	0.0
1994	20	18	25.5	23.2	22.1	19.9	18.8	1.1	0.0
1995	17	17	54.9	40.6	35.3	28.6	24.5	0.5	0.9
1996	33	28	41.4	38.6	36.4	33.9	31.5	1.1	0.0
1997	31	29	41.3	37.4	35.5	33.2	31.2	1.6	0.6
1998	35	33	40.9	38.4	37.0	35.1	33.7	1.8	0.4
1999	33	30	36.7	34.3	33.0	31.2	29.9	2.6	0.6
2000	37	34	62.6	54.5	50.9	45.9	42.9	1.2	0.9
2001	42	39	54.6	49.7	47.7	44.6	42.7	1.5	0.6
2002	52	49	72.4	66.1	63.4	59.3	56.3	2.0	0.9
2003	38	35	53.2	49.9	47.1	44.1	41.5	1.0	0.0
2004	49	48	72.6[a]	65.6	62.5	59.4	56.1	2.8	1.1

[a] This estimate is likely biased high by approximately 20%.

Occupancy of Bear Management Units by Females with Young *(Shannon Podruzny, Interagency Grizzly Bear Study Team)*

Dispersion of reproductive females throughout the ecosystem is represented by verified reports of female grizzly bears with young (COY, yearlings, 2-year-olds, and/or young of unknown age) by BMU. The population recovery requirements (USFWS 1993) include occupancy of 16 of the 18 BMUs by females with young on a running 6-year sum with no 2 adjacent BMUs unoccupied. Seventeen of 18 BMUs had verified observations of female grizzly bears with young during 2004 (Table 7). The BMU that did not contain verified documentation of females with young was the Hellroaring/Bear unit. Eighteen of 18 BMUs contained verified observations of females with young in at least 4 years of the last 6-year period.

Table 7. Bear Management Units in the Greater Yellowstone Ecosystem occupied by females with young (cubs-of-the-year, yearlings, 2-year-olds, or young of unknown age), as determined by verified reports, 1999-2004.

Bear Management Unit	1999	2000	2001	2002	2003	2004	Years occupied
1) Hilgard	X	X	X	X	X	X	6
2) Gallatin	X	X	X	X	X	X	6
3) Hellroaring/Bear	X	X	X	X	X		5
4) Boulder/Slough	X	X	X	X	X	X	6
5) Lamar	X	X	X	X	X	X	6
6) Crandall/Sunlight	X	X	X	X	X	X	6
7) Shoshone	X	X	X	X	X	X	6
8) Pelican/Clear	X	X	X	X	X	X	6
9) Washburn	X	X	X	X	X	X	6
10) Firehole/Hayden	X	X	X	X	X	X	6
11) Madison	X	X	X	X		X	5
12) Henry's Lake		X	X	X		X	4
13) Plateau	X	X	X	X	X	X	6
14) Two Ocean/Lake	X	X	X	X	X	X	6
15) Thorofare	X	X	X	X	X	X	6
16) South Absaroka	X	X	X	X	X	X	6
17) Buffalo/Spread Creek	X	X	X	X	X	X	6
18) Bechler/Teton	X	X	X	X	X	X	6
Totals	17	18	18	18	16	17	

Observation Flights *(Karrie West, Interagency Grizzly Bear Study Team)*

Two rounds of observation flights were conducted during 2004. The 37 Bear Observation Areas (BOA; Fig. 5) were surveyed once during each round (Round 1: 12 June-26 July; Round 2: 3 July-28 August). Observation time was 84 hours for Round 1 and 77 hours for Round 2; average duration of flights for both rounds combined was 2.2 hours per BOA (Table 8). One hundred ninety bear sightings, excluding dependent young, were recorded during observation flights. This included 1 solitary radio-marked bear, 3 marked females with young, 137 solitary unmarked bears, and 49 unmarked females with young (Table 8). Observation rates were 1.18 bears/hour for all bears or 0.32 females with young/hour. Ninety-seven young (74 COY, 21 yearlings, and 2 of unknown age) were observed (Table 9). Observation rate was 0.23 females with COY/hour.
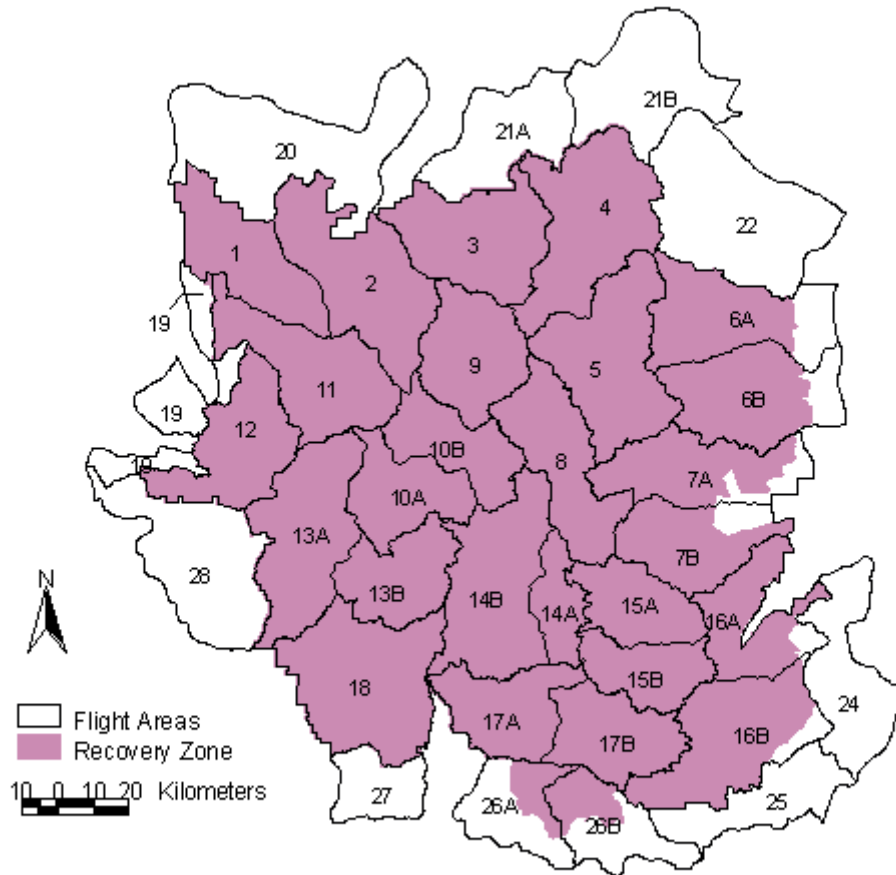
Fig. 5. Observation flight areas within the Greater Yellowstone Ecosystem, 2004. The numbers represent the 27 bear observation areas. Those units too large to search during a single flight were further subdivided into 2 units. Consequently, there were 37 search areas.

Table 8. Annual summary statistics for observation flights conducted in the Greater Yellowstone Ecosystem, 1987-2004.

Date	Observation period	Total hours	Number of flights	Average hours/flight	Marked Lone	Marked With young	Unmarked Lone	Unmarked With young	Total number of groups	Observation rate All groups	Observation rate With young	Observation rate With COY[a]
1987	Total	50.6	21	2.4					26[b]	0.51	0.16	0.12
1988	Total	34.8	17	2.0					30[b]	0.86	0.43	0.23
1989	Total	91.9	39	2.4					60[b]	0.65	0.16	0.09
1990	Total	88.1	41	2.1					48[b]	0.54	0.19	0.15
1991	Total	101.3	46	2.2					134[b]	1.32	0.52	0.34
1992	Total	61.1	30	2.0					113[b]	1.85	0.54	0.29
1993[c]	Total	56.4	28	2.0					32[b]	0.57	0.10	0.05
1994	Total	80.1	37	2.2					67[b]	0.84	0.30	0.19
1995	Total	70.3	33	2.1					62[b]	0.88	0.14	0.09
1996	Total	88.6	40	2.2					70[b]	0.79	0.27	0.23
1997[d]	Round 1	55.5	26	2.1	1	1	38	19	59	1.08		
	Round 2	59.3	24	2.5	1	1	30	17	49	0.83		
	Total	114.8	50	2.3	2	2	68	36	108	0.94	0.33	0.16
1998[d]	Round 1	73.6	37	2.0	1	2	54	26	83	1.13		
	Round 2	75.4	37	2.0	2	0	68	18	88	1.17		
	Total	149.0	74	2.0	3	2	122	44	171	1.15	0.31	0.19
1999[d]	Round 1	79.7	37	2.2	0	0	13	8	21	0.26		
	Round 2	74.1	37	2.0	0	1	21	8	30	0.39		
	Total	153.8	74	2.1	0	1	34	16	51	0.33	0.11	0.05
2000[d]	Round 1	48.7	23	2.1	0	0	8	2	10	0.21		
	Round 2	83.6	36	2.3	3	0	51	20	74	0.89		
	Total	132.3	59	2.2	3	0	59	22	84	0.63	0.17	0.12
2001[d]	Round 1	72.3	32	2.3	0	0	37	12	49	0.68		
	Round 2	72.4	32	2.3	2	4	85	29	120	1.66		
	Total	144.7	64	2.3	2	4	122	41	169	1.17	0.31	0.25
2002[d]	Round 1	84.0	36	2.3	3	0	88	34	125	1.49		
	Round 2	79.3	35	2.3	6	0	117	46	169	2.13		
	Total	163.3	71	2.3	9	0	205	80	294	1.80	0.49	0.40

Table 8. Continued.

Date	Observation period	Total hours	Number of flights	Average hours/flight	Marked Lone	Marked With young	Unmarked Lone	Unmarked With young	Total number of groups	Observation rate (bears/hour) All groups	With young	With COY[a]
2003[d]	Round 1	78.2	36	2.2	2	0	75	32	109	1.39		
	Round 2	75.8	36	2.1	1	1	72	19	93	1.23		
	Total	154.0	72	2.1	3	1	147	51	202	1.31	0.34	0.17
2004[d]	Round 1	84.1	37	2.3	0	0	43	12	55	0.65		
	Round 2	76.6	37	2.1	1	3	94	37	135	1.76		
	Total	160.8	74	2.2	1	3	137	49	190	1.18	0.32	0.23

[a] COY = Cub-of-the-year.

[b] Only includes unmarked bears. Checking for radio-marks on observed bears was added to the protocol starting in 1997.

[c] Three flights were excluded from the 1993 data because they were not flown as part of the 16 observation flight areas.

[d] Dates of flights (Round 1, Round 2): 1997 (24 Jul–17 Aug, 25 Aug-13 Sep); 1998 (15 Jul-6 Aug, 3-27 Aug); 1999 (7-28 Jun, 8 Jul–4 Aug); 2000 (5-26 Jun, 17 Jul–4 Aug); 2001 (19 Jun–11 Jul, 16 Jul–5 Aug); 2002 (12 Jun–22 Jul, 13 Jul–28 Aug); 2003 (12 Jun-28 Jul, 11 July-13 Sep); 2004 (12 Jun-26 Jul, 3 Jul-28 Aug).

Table 9. Size and age composition of family groups seen during observation flights in the Greater Yellowstone Ecosystem, 1998-2004.

Date	Females with cubs-of-the-year (number of cubs)			Females with yearlings (number of yearlings)			Females with young of unknown age (number of young)		
	1	2	3	1	2	3	1	2	3
1998[a]									
Round 1	4	10	4	0	4	2	1	2	1
Round 2	0	7	3	2	4	1	0	1	0
Total	4	17	7	2	8	3	1	3	1
1999[a]									
Round 1	2	1	1	0	1	2	1	0	0
Round 2	1	2	0	0	3	1	0	1	0
Total	3	3	1	0	4	3	1	1	0
2000[a]									
Round 1	1	0	0	0	0	0	0	1	0
Round 2	3	11	1	1	2	0	0	2	0
Total	4	11	1	1	2	0	0	3	0
2001[a]									
Round 1	1	8	1	1	0	0	0	0	1
Round 2	14	10	2	4	2	1	0	0	0
Total	15	18	3	5	2	1	0	0	1
2002[a]									
Round 1	8	15	5	3	2	0	0	0	1
Round 2	9	19	9	2	4	2	0	1	0
Total	17	34	14	5	6	2	0	1	1
2003[a]									
Round 1	2	12	2	2	6	2	3	3	0
Round 2	2	5	3	2	5	0	2	0	1
Total	4	17	5	4	11	2	5	3	1
2004[a]									
Round 1	4	1	3	1	1	0	2	0	0
Round 2	6	16	7	4	7	0	0	0	0
Total	10	17	10	5	8	0	2	0	0

[a] Dates of flights (Round 1, Round 2): 1998 (15 Jul-6 Aug, 3-27 Aug); 1999 (7-28 Jun, 8 Jul–4 Aug); 2000 (5-26 Jun, 17 Jul–4 Aug); 2001 (19 Jun–11 Jul, 16 Jul–5 Aug); 2002 (12 Jun–22 Jul, 13 Jul–28 Aug); 2003 (12 Jun-28 Jul, 11 Jul-13 Sep); 2004 (12 Jun-26 Jul, 3 Jul-28 Aug).

Telemetry Relocation Flights *(Karrie West, Interagency Grizzly Bear Study Team)*

Ninety-five telemetry relocation flights were conducted during 2004, resulting in 375.2 hours of search time (ferry time to and from airports excluded) (Table 10). Flights were conducted at least once during all months, with 74% occurring May-November. During telemetry flights, 649 locations of bears equipped with radio transmitters were collected, 45 (7%) of which included a visual sighting. Forty-one sightings of unmarked bears were also obtained during telemetry flights, including 33 solitary bears and 8 females with COY. Rate of observation for all unmarked bears during telemetry flights was 0.11 bears/hour. Rate of observing females with COY was 0.02/hour, which was considerably less than during observation flights (0.23/hour) in 2004.

Table 10. Summary statistics for radio-telemetry relocation flights in the Greater Yellowstone Ecosystem, 2004.

| Month | Hours | Number of flights | Mean hours per flight | Radioed bears | | | Lone bears | Unmarked bears observed | | | Observation rate (groups/hour) | |
| | | | | Number of locations | Number seen | Observation rate (groups/hour) | | Females | | | | |
								With COY[a]	With yearlings	With young	All groups	Females with COY
January	11.32	4	2.83	25	0	0.00	0	0	0	0	-----	-----
February	8.60	2	4.30	16	0	0.00	0	0	0	0	-----	-----
March	9.85	3	3.28	26	2	0.20	0	0	0	0	-----	-----
April	34.80	10	3.48	66	13	0.37	7	1	0	0	0.23	0.03
May	35.28	11	3.21	53	7	0.20	3	0	0	0	0.09	0.00
June	40.13	9	4.46	38	4	0.10	0	0	0	0	-----	-----
July	30.23	8	3.78	35	2	0.07	0	0	0	0	-----	-----
August	56.41	13	4.34	75	6	0.11	19	5	0	0	0.43	0.09
September	34.99	8	4.37	66	3	0.09	0	1	0	0	0.03	0.03
October	46.57	11	4.23	76	5	0.11	4	1	0	0	0.11	0.02
November	49.58	11	4.51	127	3	0.06	0	0	0	0	-----	-----
December	17.43	6	2.91	46	0	0.00	0	0	0	0	-----	-----
Total	375.19	96	3.91	649	45	0.12	33	8	0	0	0.11	0.02

[a] COY = cub-of-the-year.

Grizzly Bear Mortalities *(Mark A. Haroldson, Interagency Grizzly Bear Study Team; and Kevin Frey, Montana Fish, Wildlife and Parks)*

We continue to use the definitions provided in Craighead et al. (1988) to classify grizzly bear mortalities in the GYE relative to the degree of certainty regarding each event. Those cases in which a carcass is physically inspected or when a management removal occurs are classified as "known" mortalities. Those instances where evidence strongly suggests a mortality has occurred but no carcass is recovered are classified as "probable" mortalities. When evidence is circumstantial, with no prospect for additional information, a "possible" mortality is designated.

We documented 26 grizzly bear mortalities during 2004 (Table 11). Nineteen (including 9 females and 10 males) were known human-caused bear deaths. Two of these occurred >10 miles outside the Recovery Zone in Wyoming (Tables 11 and 12). Both of these instances involved male grizzly bears that were misidentified and mistakenly killed by black bear hunters over bait. Seven of the human-caused mortalities were management removals (Table 11) resulting from a variety of causes, including anthropogenic food rewards ($n = 2$), property damage ($n = 3$), livestock depredation ($n = 1$), and nuisance activity in a campground ($n = 1$). In addition to the mistaken identity kill outside the 10-mile perimeter, we documented 7 other known hunting-related mortalities (Table 11). Five of the hunting-related bear mortalities were the result of chance encounters between bears and hunters; 4 of these involved females with yearlings (Table 11). The 2 remaining hunting-related mortalities resulted from conflicts at hunter killed ungulate carcasses; 1 of these involved a female with yearlings (Table 11). The 3 remaining human-caused mortalities resulted from 1 road kill, 1 defense of life at a backcountry camp, and 1 potential poisoning that remains under investigation (Table 11).

Five natural mortalities were documented during 2004 (Table 11). Two were known COY deaths, 2 were probable COY losses from 2 radiomarked females. We also considered the humane removal of a subadult male bear that was mortally wounded by another bear a natural mortality.

Cause of death could not be determined for 2 mortalities documented during 2004 (Table 11). During June, an outfitter found the remains of a subadult male bear that likely died during the fall of 2003. Cause of death for this bear could not be determined. Bear #456 was handled and fitted with a GPS collar on 14 June 2004. Data on the collar indicated that #456 died approximately 15 days after being handled. The carcass had been mostly scavenged by the time it was discovered on 12 July and cause of death could not be determined.

The Grizzly Bear Recovery Plan (USFWS 1993:41-44) provides criteria for determining if human-caused grizzly bear mortalities have exceeded annual thresholds established in the plan. Appendix F of the Grizzly Bear Recovery Plan (USFWS 1993) intended that known mortalities occurring within the Yellowstone Grizzly Bear Recovery Zone **and** a 10-mile perimeter area be counted against mortality quotas. The U.S. Fish and Wildlife Service clarified this with an amendment to the Recovery Plan. In addition, beginning in 2000, probable mortalities were included in the calculation of mortality thresholds, and COY orphaned as a result of human causes will be designated as probable mortalities (see Appendix A in Schwartz and Haroldson 2001). Prior to these changes, COY orphaned after 1 July were designated possible mortalities (Craighead et al. 1988). Sex of probable mortalities will be randomly assigned as described in Appendix A in Schwartz and Haroldson (2001). Under these criteria, 17 known human-caused grizzly bear mortalities, including 6 adult females and 9 total females, were applied to the calculation of mortality threshold (USFWS 1993) for 2004. None of the females killed in

encounters with hunters had COY, in all instances, investigators determined that offspring were yearlings. Using these results, total human-caused mortality was under, but female mortalities exceeded the annual mortality thresholds during 2004 (Table 13). The female mortality threshold had not been exceeded since 1997.

Table 11. Grizzly bear mortalities documented in the Greater Yellowstone Ecosystem during 2004.

Bear[a]	Sex	Age[b]	Date	Location[c]	Certainty	Cause
unm	M	subadult	Fall/2003	Fishhawk Cr, SNF	Known	Undetermined cause, remains found by outfitter on 6/17/04.
G92	M	adult	4/12/04	Clark's Fork, Pr-WY	Known	Human-caused, management removal for repeated nuisance activity at human habitations and breaking into secured buildings.
unm	M	subadult	5/1/04	Grass Cr, State-WY	Known	Human-caused, mistaken identity, illegal, shot over black bear bait. Outside 10-mile perimeter.
unm	M	adult	5/7/04	Timber Cr, BLM-WY	Known	Human-caused, self-defense, bear was shot as it came into horn-hunter camp at night.
431	M	subadult	5/10/04	Sweeney Cr, BLM-WY	Known	Human-caused, mistaken identity, illegal, bear was shot over black bear bait. Outside 10-mile perimeter.
unm	M	COY	5/22/04	Lamar River, YNP	Known	Natural mortality, skull crushed by large predator.
unm	F	COY	6/7/04	Lamar River, YNP	Known	Natural mortality, specific cause undetermined, necropsy report was inconclusive, but likely not predation.
348	M	adult	6/26/04	Deer Cr, Pr-WY	Known	Human-caused, management removal for repeated livestock depredation.
451	F	subadult	6/29/04	N Fork Shoshone River, SNF	Known	Human-caused, management removal for numerous food rewards from garbage and repeated property damage.
456	M	adult	6/29/04	W DuNoir Cr, SNF	Known	Undetermined cause, died 6/29, was captured 6/14. Carcass was approximately 12 km from capture site.
unm	Unk	COY	5/3-7/15/04	Pilgrim Cr, GTNP	Probable	Natural mortality, specific cause unknown, single cub of bear #399 lost between 5/3 and 7/15, location is approximate, estimate is an average location for the interval.
unm	Unk	COY	7/18-8/10/04	Wapiti Cr, GNF	Probable	Natural mortality, specific cause unknown, 1 of 2 cubs of bear #213 lost between 7/18 and 8/10, location is approximate, estimated as average location for the interval.
372	M	adult	8/9/04	Paint Cr, SNF	Known	Human-caused, management removal for breaking into buildings.
305	F	adult	9/2/04	Big Cr, Pr-WY	Known	Human-caused, management removal for repeatedly breaking into buildings, 2 yearlings (G94 and G95) were relocated to Fox Cr, SNF. Bear had recently been shot with #8 bird shot.
unm	F	adult	9/3/04	Falls Cr, BTNF	Known	Human-caused, self-defense, hunting related at elk carcass, female with yearling(s), minor injuries to hunter and guide.
unm	F	subadult	9/7/04	Pebble Cr, YNP	Known	Human-caused, management removal for campground depredation.

Table 11. Continued.

Bear[a]	Sex	Age[b]	Date	Location[c]	Certainty	Cause
G80	M	adult	9/8/04	Nez Perce Cr, YNP	Known	Human-caused, road kill, bear was hit and killed by a vehicle.
Unm	F	Adult	9/11/04	Fishhawk Cr, SNF	Known	Human-caused, self-defense, hunting related, chance encounter, female with yearling.
unm	M	subadult	9/25/04	S Fork Shoshone, Pr-WY	Known	Natural, humane removal, bear was badly injured by another bear and would have died, was removed due to extent of injuries. Bear had recently been shot with #2 steel shot.
G97	F	subadult	9/28/04	Eagle Cr, SNF	Known	Human-caused, management removal for numerous food rewards.
417	M	adult	10/3/04	N Fork Fish Cr, BTNF	Known	Human-caused, self-defense, hunting related, chance encounter while hunters were in pursuit of game, human injuries.
unm	F	adult	10/9/04	Crandall Cr, SNF	Known	Human-caused, self-defense, hunting related, chance encounter, female with 3 yearlings charged hunter.
unm	F	adult	10/9/04	Silver Cr, SNF	Known	Human-caused, self-defense, hunting related, chance encounter, female with 2 yearlings charged hunter.
220	M	adult	10/19/04	Open Cr, BTNF	Known	Human-caused, self-defense, hunting related, bear came in on carcass with hunter and guide.
mkd	M	adult	10/31/04	Spanish Cr, GNF	Known	Human-caused, likely accidental poisoning, bear raided hunting camp tent that was unused for a period of time. Bear was marked, but unable to determine number.
172	F	adult	11/11/04	Pacific Cr, BTNF	Known	Human-caused, self-defense, hunting related, chance encounter on trail, female with 2 yearlings charged hunter.

[a] Unm = unmarked bear; mkd = marked bear, number indicates bear number .

[b] COY = cub-of-the-year. Unk = unknown age

[c] BLM = Bureau of Land Management, BTNF = Bridger-Teton National Forest, GNF = Gallatin National Forest, GTNP = Grand Teton National Park, SNF = Shoshone National Forest, YNP = Yellowstone National Park, Pr = private.

[d] Occurred >10 miles outside the Recovery Zone.

Table 12. Known and probable grizzly bear deaths in the Greater Yellowstone Ecosystem, 1983-2004.

	All bears				Adult females			
	Human-caused		Other[a]		Human-caused		Other	
	In[b]	Out[b]	In	Out	In	Out	In	Out
1983	6	0	1	0	2	0	0	0
1984	9	0	2	0	2	0	0	0
1985	5	1	7	0	2	0	0	0
1986	5	4	2	0	1	1	0	0
1987	3	0	0	0	2	0	0	0
1988	5	0	7	0	0	0	2	0
1989	2	0	1	0	0	0	0	0
1990	9	0	0	0	4	0	0	0
1991	0	0	0	0	0	0	0	0
1992	4	0	4	0	0	0	0	0
1993	3	0	2	0	2	0	1	0
1994	11	1	1	0	4	0	0	0
1995	17	0	1	0	3	0	0	0
1996	10[c]	0	4	1	3	0	0	0
1997	8	2	10[d]	0	3	0	0	0
1998	1	2	3	0	1	0	0	0
1999	7[e]	1	7	0	1	0	0	0
2000[f]	16	6	10	0	3	1	0	0
2001	19	1	12[g]	0	6	0	1	0
2002	15	2	8[h]	0	4	0	3[g]	0
2003	11	1	4	0	3	0	0	0
2004	17	2	7[i]	0	6	0	0	0

[a] Includes deaths from natural and unknown causes.
[b] In refers to inside the Recovery Zone or within a 10-mile perimeter of the Recovery Zone. Out refers to >10 miles outside the Recovery Zone.
[c] Includes 1 known human-caused mortality from 1996 discovered during 1999.
[d] Includes 1 mortality from the fall of 1997 discovered in 1998.
[e] Includes 1 probable human-caused mortality from 1999 discovered in 2000.
[f] Starting in 2000, includes human-caused orphaned cubs-of-the-year (Appendix A in Schwartz and Haroldson 2001).
[g] Includes 1 known mortality from fall of 2001 discovered in 2002.
[h] Includes 1 known mortality from 2002 discovered in 2003.
[i] Includes 1 known mortality from 2003 discovered in 2004.

Table 13. Annual count of unduplicated females with cubs-of-the-year (COY), known and probable[a] human-caused grizzly bear mortalities within the Recovery Zone and the 10-mile perimeter, 1993-2004. Calculations of mortality thresholds (USFWS 1993) do not include mortalities or unduplicated females with COY documented outside the 10-mile perimeter.

| | | Human-caused mortality | | | Human-caused mortality 6-year running averages | | | | U.S. Fish and Wildlife Service Grizzly Bear Recovery Plan mortality thresholds | | | | |
| | | | | | | | | | Total human-caused mortality | | | Total female mortality | |
Year	Unduplicated females with COY	Total	Female	Adult female	Total	Female	Adult female	Minimum population estimate	4% of minimum population	Year result	30% of total mortality	Year result
1993	19	3	2	2	3.8	1.8	1.0	241	9.6	Under	2.9	Under
1994	20	10	3	3	4.7	2.0	1.5	215	8.6	Under	2.6	Under
1995	17	17	7	3	7.2	3.2	2.0	175	7.0	Exceeded	2.1	Exceeded
1996	33	10	4	3	7.3	2.8	1.8	223	8.9	Under	2.7	Exceeded
1997	31	7	3	2	8.5	3.3	2.2	266	10.7	Under	3.2	Exceeded
1998	35	1	1	1	8.0	3.3	2.3	339	13.6	Under	4.1	Under
1999	32	5	1	1	8.3	3.2	2.2	343	13.7	Under	4.1	Under
2000[a]	35	16	5	3	9.3	3.5	2.2	354	14.2	Under	4.2	Under
2001	42	17	8	6	9.3	3.7	2.7	361	14.5	Under	4.3	Under
2002	50	15	7	4	10.2	4.2	2.8	416	16.6	Under	5.0	Under
2003	35	10	6	3	10.7	4.7	3.0	416	16.6	Under	5.0	Under
2004	46	17	9	6	13.3	6.0	3.8	431	17.2	Under	5.2	Exceeded

[a] Beginning in 2000, probable human-caused mortalities are used in calculation of annual mortality thresholds.

Key Foods Monitoring

Spring Ungulate Availability and Use by Grizzly Bears in Yellowstone National Park.
(Shannon Podruzny, Interagency Grizzly Bear Study Team; and Kerry Gunther, Yellowstone National Park)

       It is well documented that grizzly bears use ungulates as carrion (Mealey 1980, Henry and Mattson 1988, Green 1994, Blanchard and Knight 1996, Mattson 1997) in YNP. Competition with recently reintroduced wolves for carrion and changes in bison (*Bison bison*) and elk (*Cervus elaphus*) management policies in the GYE have the potential to affect carcass availability and use by grizzly bears. For these and other reasons, we continue to survey historic carcass transects in YNP. In 2004, we surveyed routes in ungulate winter ranges to monitor the relative abundance of spring ungulate carcasses (Fig. 6).
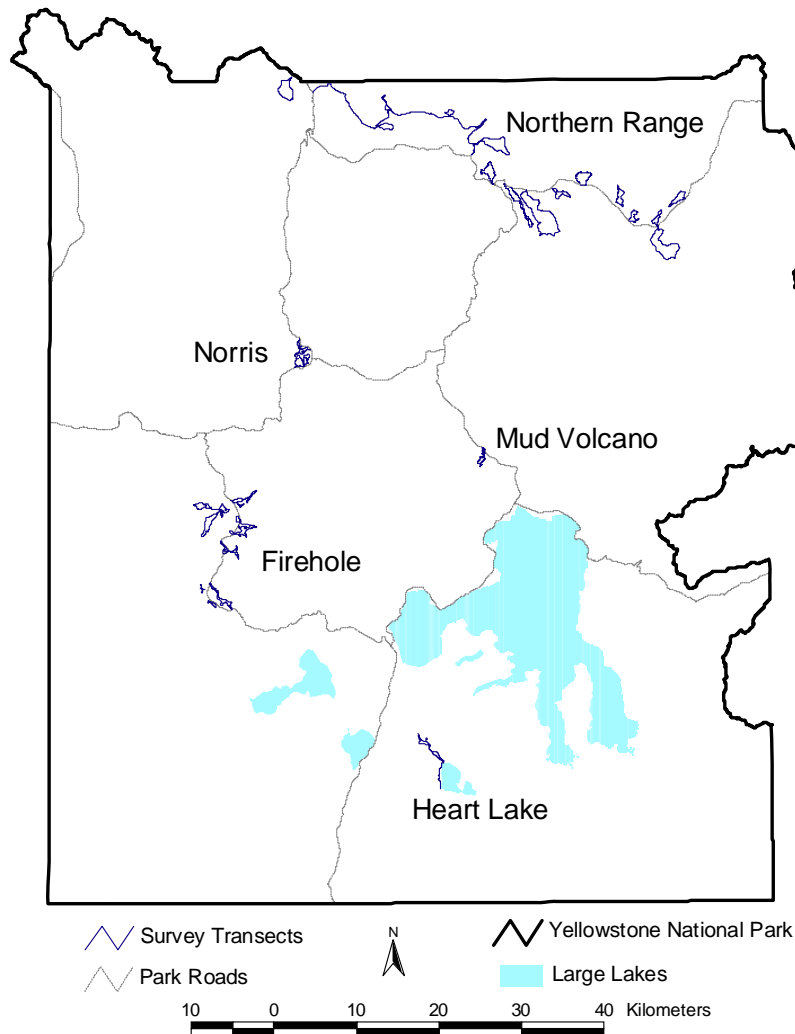
Fig. 6. Spring ungulate carcass survey transects in 5 areas of Yellowstone National Park.

We surveyed each route once for carcasses between April and early-May. At each carcass, we collected a site description (i.e., location, aspect, slope, elevation, distance to road, distance to forest edge), carcass data (i.e., species, age, sex, cause of death), and information about animals using the carcasses (i.e., species, percent of carcass consumed, scats present). We were unable to calculate the biomass consumed by bears, wolves, or other unknown large scavengers with our survey methodology.

We are interested in relating the changes in ungulate carcass numbers to potential independent measures of winter die-off. Such measures include weather, winter severity, and forage availability. All are considered limiting factors to ungulate survival during winter (Cole 1971, Houston 1982). Long-term changes in weather and winter severity monitoring may be useful in predicting potential carcass availability. The Winter Severity Index (WSI) developed for elk (Farnes 1991), tracks winter severity, monthly, within a winter and is useful to compare among years. WSI uses a weight of 40% of minimum daily winter temperature below 0° F, 40% of current winter's snow pack (in snow water equivalent), and 20% of June and July precipitation as surrogate for forage production (Farnes 1991).

Northern Range

We surveyed 12 routes on Yellowstone's Northern Range totaling 153.7 km traveled. In 2004, we used a GPS to more accurately measure the actual distance traveled on most of the routes. We counted 15 carcasses, including 3 bison and 12 elk, which equated to 0.098 carcasses/km (Table 14). Sex and age of carcasses found are shown in Table 15. All carcasses were almost completely consumed by scavengers, evidence of use by bears could not be determined at any of the carcasses. Two elk may have been killed by wolves, but none of the carcasses showed definitive evidence of cause-of-death. Grizzly bear sign (e.g., tracks, scats, or feeding activity) was observed along 1 of the routes, black bear sign was observed along 2 routes. Bear sign of an undetermined species was identified on 2 additional routes

Table 14. Carcasses found and visitation of carcasses by bears, wolves, and unknown large scavengers along surveyed routes in Yellowstone National Park during spring 2004.

Survey area (# routes)	Elk				Bison				Total Carcasses/km
	Number of carcasses	# Visited by species			Number of carcasses	# Visited by species			
		Bear	Wolf	Unknown		Bear	Wolf	Unknown	
Northern Range (12)	12	0	0	12	3	1	0	2	0.098
Firehole (8)	7	0	4	4	7	5	1	1	0.204
Norris (4)	1	0	0	1	7	6	1	1	0.323
Heart Lake (1)	0	0	0	0	0	0	0	0	0
Mud Volcano (1)	0	0	0	0	0	0	0	0	0

Table 15. Age classes and sex of elk and bison carcasses found, by area, along surveyed routes in Yellowstone National Park during spring 2004.

| | Elk (n =20) | | | | | | Bison (n =17) | | | | | |
	Northern Range	Firehole	Norris	Heart Lake	Mud Volcano	Total	Northern Range	Firehole	Norris	Heart Lake	Mud Volcano	Total
<u>Age</u>												
Adult	5	2	1	0	0	8	3	7	2	0	0	12
Yearling	0	0	0	0	0	0	0	0	2	0	0	2
Calf	2	0	0	0	0	2	0	0	3	0	0	3
Unknown	5	5	0	0	0	10	0	0	0	0	0	0
<u>Sex</u>												
Male	3	2	0	0	0	5	1	4	1	0	0	6
Female	2	0	0	0	0	2	2	1	4	0	0	7
Unknown	7	5	1	0	0	13	0	2	2	0	0	4

Firehole River Area

We surveyed 8 routes in the Firehole drainage totaling 68.6 km. We found the remains of 7 bison and 7 elk, which equated to 0.204 carcasses/km traveled (Table 14). Evidence of use by wolves was found at 1 bison carcass. Definitive evidence of use by bears was found at 5 bison carcasses. Grizzly bear sign was found along 5 of the routes, and black bear tracks were found on 1 route. One bison was probably winter-killed and 1 other bison may have been killed by a bear. Wolves likely killed 2 of the elk, cause of death could not be determined for the 5 other elk carcasses.

Norris Geyser Basin

We surveyed 4 routes in the Norris Geyser Basin totaling 24.8 km traveled. We found 7 bison and 1 elk carcass, which equated to 0.323 carcasses/km (Table 14). Five of the bison appear to have died due to hydrogen-sulfide poisoning as a result of weather-induced concentration of gases from nearby thermal vents. Cause of death could not be determined for the elk carcass as well as for 1 calf bison and 1 yearling bison. Evidence of use by bears was observed at 6 bison carcasses. Wolf sign was found at 1 bison and 4 elk carcasses. We observed grizzly bear tracks along all 4 routes.

Heart Lake

We surveyed 1 route in the Heart Lake thermal basin covering 6.6 km. We observed no carcasses. Grizzly and black bear sign was observed along the route.

Mud Volcano

We surveyed 1 route in the Mud Volcano area covering 7.4 km. No carcasses were observed this spring, but grizzly bear tracks were abundant.

According to the WSI, the winter of 2003-2004 presented milder-than-average conditions (Fig. 7). There were fewer carcasses observed than in previous years, and our index of carcass abundance was lower in 2003-2004 compared to the relatively severe winter of 1996-1997 (Fig. 8). We found a significant correlation between the WSI and numbers of carcasses observed on the Northern Range ($R^2 = 0.81$, $n = 12$, $F = 42.7$, $P < 0.001$), and in the Firehole/Norris basins ($R^2 = 0.68$, $n = 17$, $F = 31.6$, $P < 0.001$).
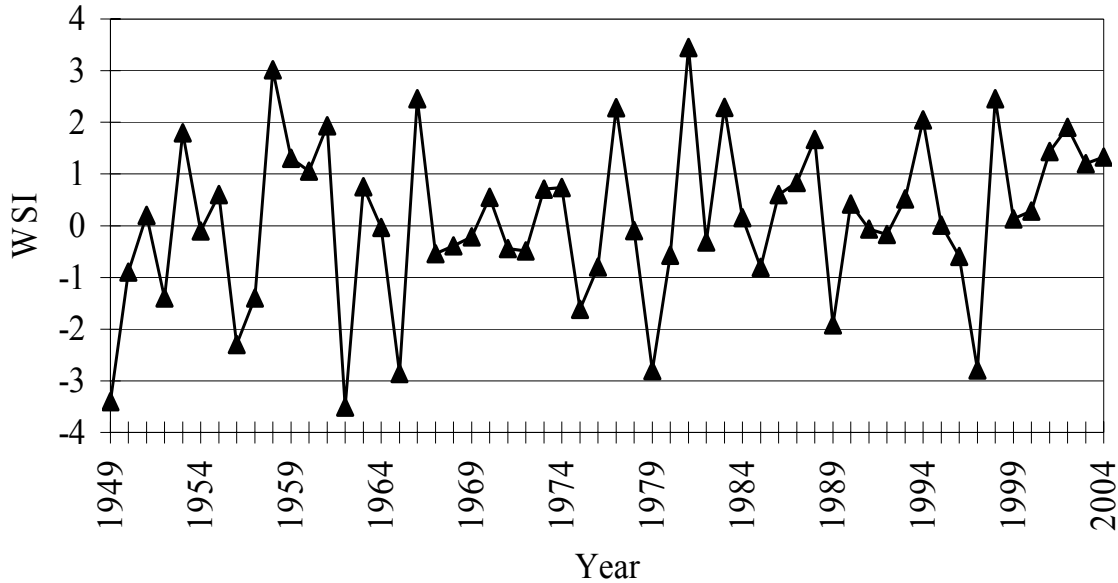
Fig. 7. Winter Severity Index (WSI) for the Northern Range of Yellowstone National Park, 1949-2004.
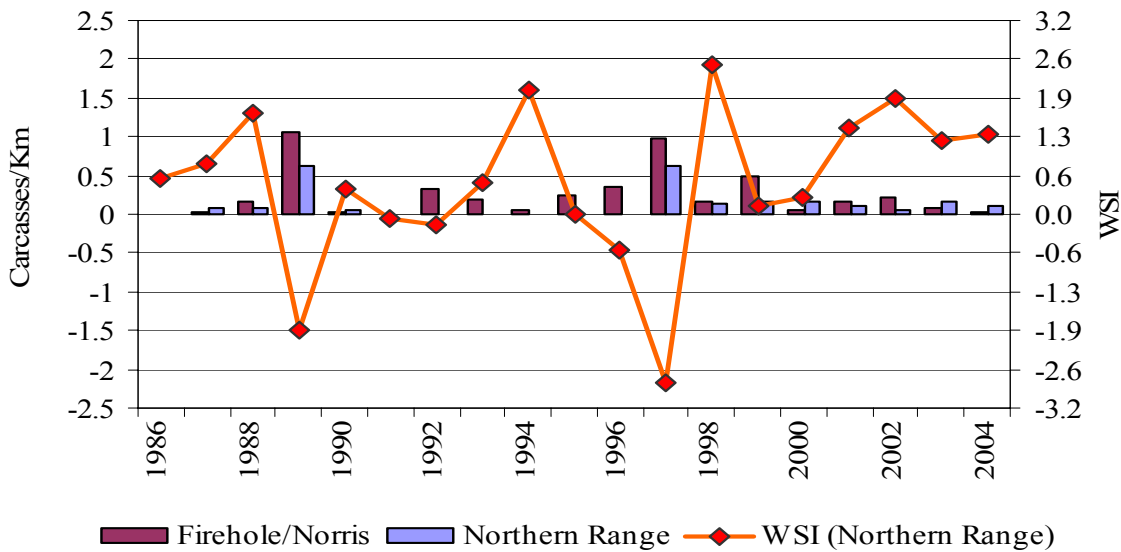
Fig. 8. Winter Severity Index (WSI) for elk on the Northern Range of Yellowstone National Park and ungulate carcasses per kilometer surveyed, 1986-2004.

Spawning Cutthroat Trout *(Kerry A. Gunther, C. Travis Wyman, Todd M. Koel, Patrick Perrotti, and Eric Reinertson, Yellowstone National Park)*

Spawning cutthroat trout are one of the highest sources of energy available to grizzly bears in YNP (Mealey 1975, Pritchard and Robbins 1990), and influence the distribution of bears over a large geographic area (Mattson and Reinhart 1995). Grizzly bears are known to prey on cutthroat trout in at least 36 different tributary streams to Yellowstone Lake (Hoskins 1975, Reinhart and Mattson 1990). Haroldson et al. (2005) estimated that approximately 60 grizzly bears likely fished Yellowstone Lake tributary streams annually. Male grizzly bears appear to dominate the spawning streams around Yellowstone Lake and consume greater quantities of trout than female bears (Felicetti et al. 2004). Bears also occasionally prey on cutthroat trout in the Trout Lake inlet in the northwest section of the park.

The cutthroat trout population in Yellowstone Lake is now threatened by the introduction of exotic lake trout (*Salvelinus namaycush*) and whirling disease (*Myxobolus cerebralis*) (Koel et al. 2003). Lake trout and whirling disease could depress the native cutthroat trout population and associated bear fishing activity (Haroldson et al. 2005). There is evidence that the number of spawning cutthroat trout in Yellowstone Lake is declining. Reinhart et al. (1995) reported a decline in the number of spawning cutthroat trout in North Shore and West Thumb spawning streams during the period 1989-1995, as compared to the period 1985-1987. The downward trend has generally continued in all monitored streams during the period 1996-2004. Non-native lake trout were discovered in Yellowstone Lake in 1994 (Kaeding et al. 1996) and have probably been present in the lake since 1988 (Munro et al. 2005). Lake trout are highly predatory on cutthroat trout and have significantly reduced native trout populations in other lakes where they have been introduced (Gerstung 1988, Donald and Alger 1993). Younger age classes of lake trout compete with cutthroat trout for macroinvertebrates consumed by both species (Elrod and O'Gorman 1991). Older lake trout are highly predatory on cutthroat trout and may consume at least 41 cutthroat trout per year (Ruzycki et al. 2003). Without control, lake trout could reduce the cutthroat trout population in Yellowstone Lake by as much as 90% (McIntyre 1996).

Whirling disease was discovered in Yellowstone Lake in 1998 (Koel et al. 2003). Whirling disease primarily affects young cutthroat trout by destroying head cartilage, resulting in loss of equilibrium, skeletal deformities, and inability to feed normally and avoid predators (Yellowstone Center for Resources 2002). Whirling disease has devastated wild trout populations in other waters of the Intermountain West (Nickum 1999). In addition to lake trout and whirling disease, wildfire, and drought may also be contributing to the decline of the Yellowstone Lake cutthroat trout population. Due to the importance of cutthroat trout to grizzly bears and the potential threats from lake trout and whirling disease, monitoring of the cutthroat trout population is specified under the Yellowstone grizzly bear Conservation Strategy (USFWS 2003). The cutthroat trout population is currently monitored annually using counts at fish traps and during stream surveys (Koel 2001, USFWS 2003).

Yellowstone Lake

Fish trap surveys.--The number of spawning cutthroat trout migrating upstream are counted annually from weirs with fish traps at the mouths of Clear Creek and Bridge Creek on the east and north sides of Yellowstone Lake, respectively (Koel 2001). The fish traps are generally installed in May, the exact date depending on winter snow accumulation, weather

conditions, and spring snow melt (Koel 2001). Fish are counted by dip netting trout that enter the upstream trap box and/or visually counting trout as they swim through wooden chutes attached to the traps (Koel 2001). An electronic fish counter is also periodically used (Koel 2001).

In 2004, 1,438 upstream migrating cutthroat trout were counted at Clear Creek (Koel et al. in press), this represents a 58% decrease from the total of 3,432 trout counted in 2003 (Koel et al. 2004), and a 98% decrease since the peak upstream spawner count of 70,105 in 1978. The 1,438 spawners counted in 2004, was the lowest count since monitoring began in 1945 (Koel et al. in press). Lake trout are thought to have been illegally introduced into Yellowstone Lake in the mid-1980s (Munro et al. 2001). The number of cutthroat trout counted at Clear Creek has generally declined (Fig. 9) since the mid-1980s (Koel et al. 2003). The number of spawning cutthroat trout migrating up Bridge Creek has also declined in recent years (Koel et al. 2003). In 2004, only 1 cutthroat trout was counted at the Bridge Creek weir (Koel et al. in press). This represents an 89% decrease from the 86 upstream migrants counted in 2003 (Koel et al. 2004), and a decrease of >99% since monitoring began in 1999 (Fig. 10) (Koel et al. in press).
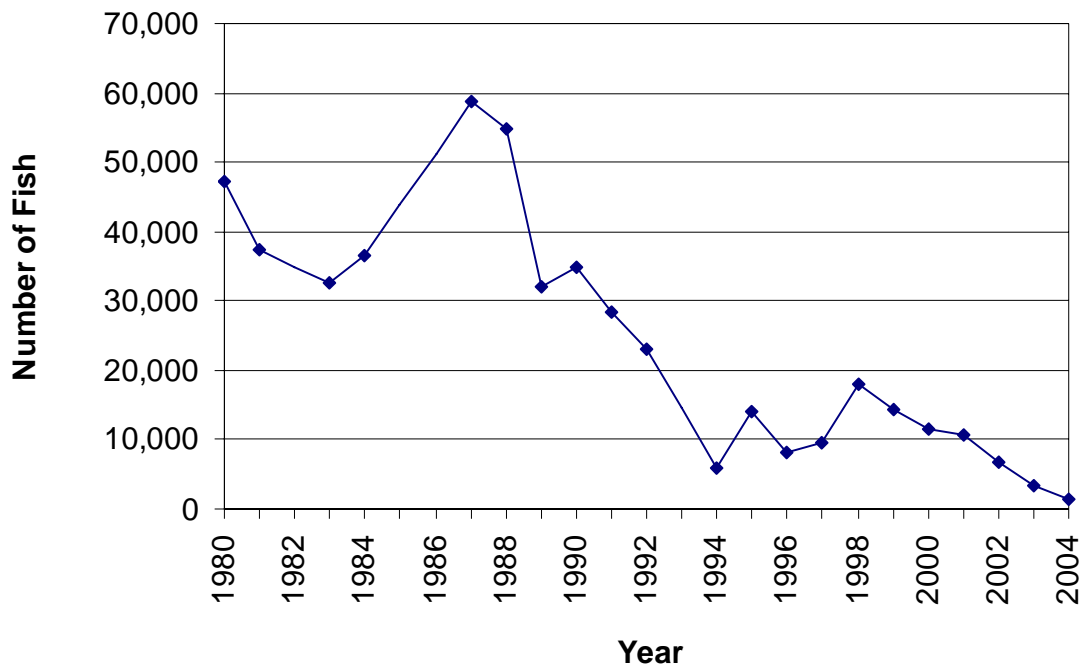
Fig. 9. Number of upstream migrating spawning cutthroat trout counted at the Clear Creek fish trap on the east shore of Yellowstone Lake, Yellowstone National Park, 1980-2004.
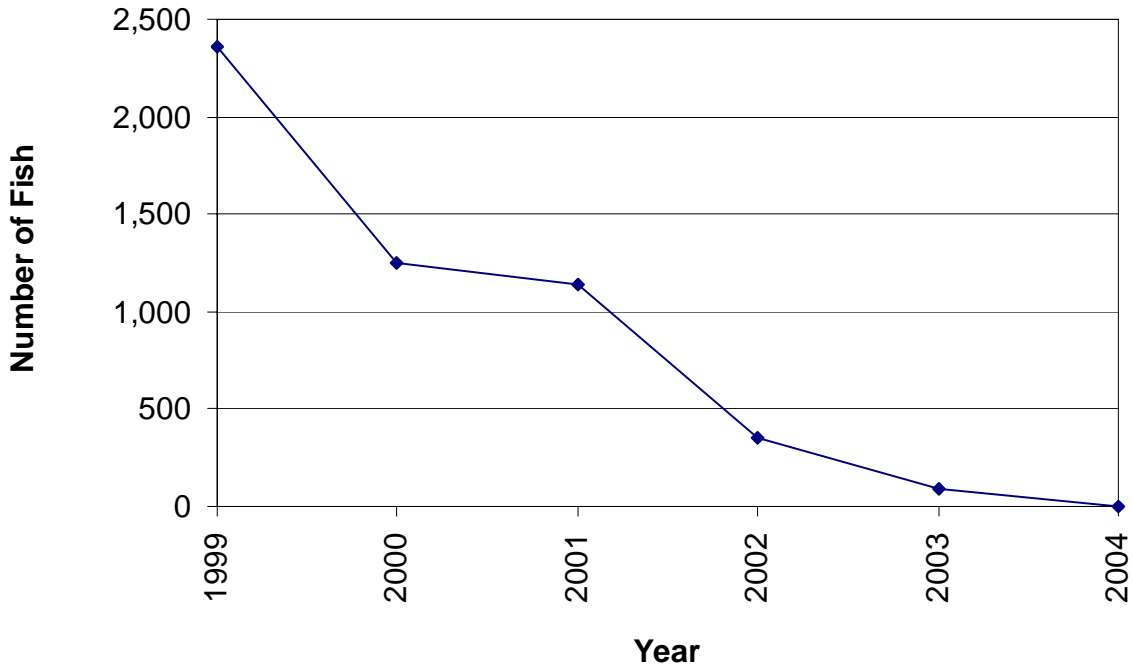
Fig. 10. Number of upstream migrating spawning cutthroat trout counted at the Bridge Creek fish trap on the north shore of Yellowstone Lake, Yellowstone National Park, 1999-2004.

Spawning stream surveys.--Beginning 1 May each year, several streams including Lodge, Hatchery, Incinerator, Wells, and Bridge Creeks on the north shore of Yellowstone Lake, and Sandy, Sewer, Little Thumb, and 1167 Creeks in the West Thumb area are checked daily to detect the presence of adult cutthroat trout (Andrascik 1992, Olliff 1992). Once adult trout are found (i.e., onset of spawning), weekly surveys of cutthroat trout in these streams are conducted. Sample methods follow Reinhart (1990), as modified by Andrascik (1992) and Olliff (1992). In each stream on each sample day, 2 people walk upstream from the stream mouth and record the number of adult trout observed. Sampling continues 1 day per week until most adult trout return to the lake (i.e., end of spawning). The peak spawner count (the number of fish counted during the peak week) is used to identify annual trends in the number of cutthroat trout spawning in the Trout Lake inlet.

    Data collected in 2004 continued to show low numbers of spawning cutthroat on North Shore and West Thumb streams. On North Shore streams only 2 spawning cutthroat trout were counted in Bridge Creek, 1 in Lodge Creek, and none in Hatchery, Incinerator, or Wells Creeks. On West Thumb streams 8 spawning cutthroat trout were counted in Sandy Creek, 4 in Little Thumb Creek, and 3 in Sewer Creek. No spawning cutthroat trout were counted in 1167 Creek. The number of spawners counted in the North Shore and West Thumb streams have decreased noticeably since 1989 (Figs. 11 and 12).
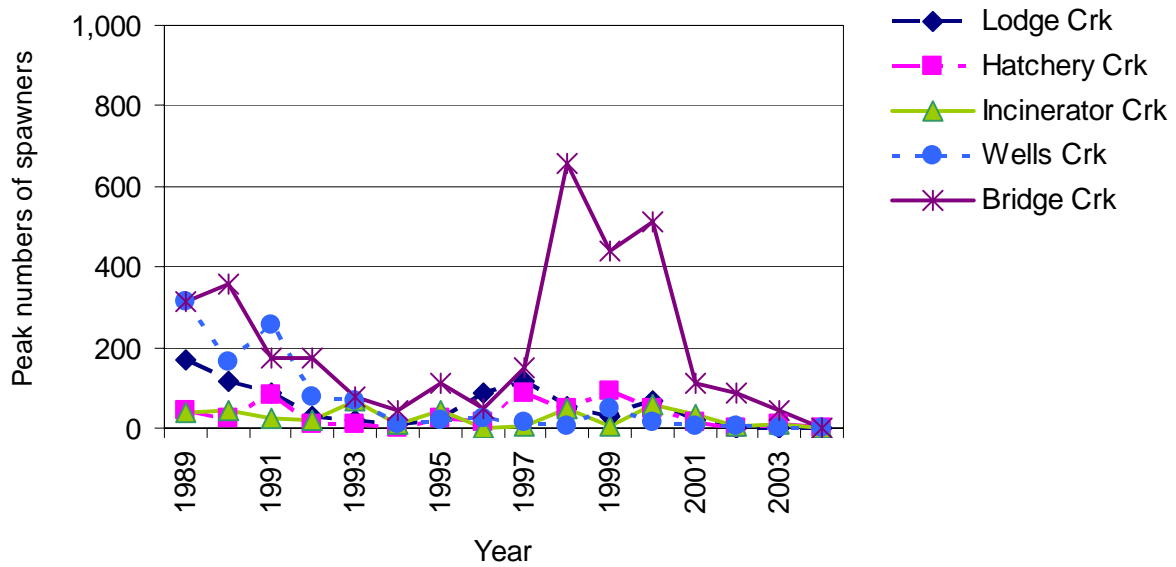
Fig. 11. Number of spawning cutthroat trout counted during the peak week in North Shore spawning streams (Lodge, Hatchery, Incinerator, Wells, and Bridge Creeks) tributary to Yellowstone Lake, Yellowstone National Park, 1989-2004.
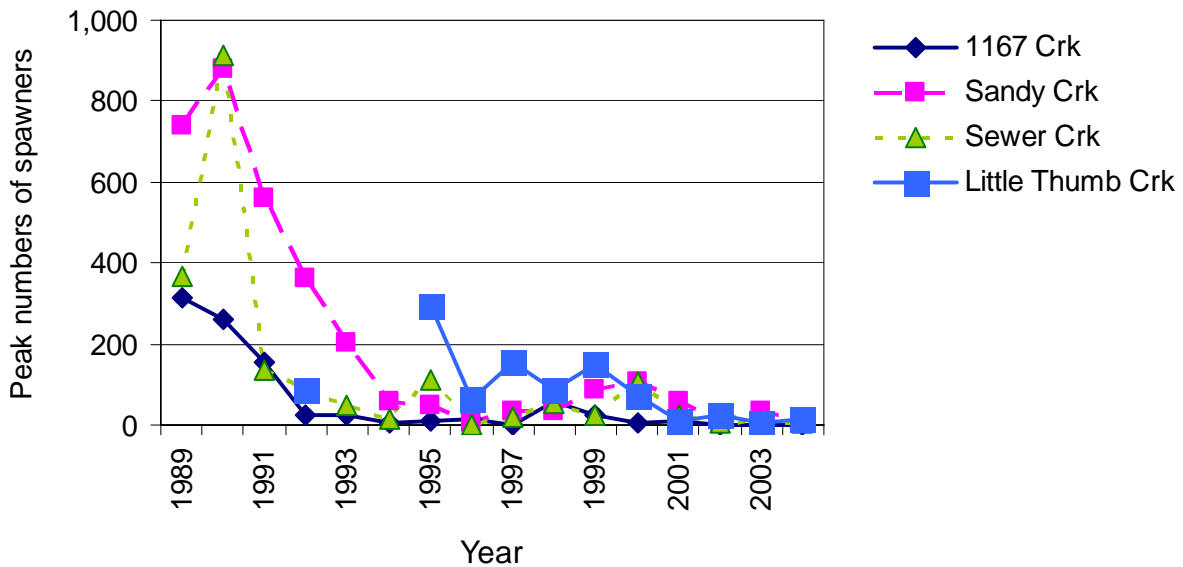
Fig. 12. Number of spawning cutthroat trout counted during the peak week in West Thumb spawning streams (1167, Sandy, Sewer, and Little Thumb Creeks) tributary to Yellowstone Lake, Yellowstone National Park, 1989-2004.

Trout Lake

Spawning stream surveys.--Beginning the first week of June each year, the number of spawning cutthroat trout migrating up the Trout Lake inlet are counted once per week. On each sample day, 2 people walk upstream from the stream mouth and record the number of adult trout observed. Sampling continues 1 day per week until most adult trout return to Trout Lake (i.e., end of spawning). The peak spawner count (the number of fish counted during the peak week) is used to identify annual trends in the number of cutthroat trout spawning in the Trout Lake inlet.

In 2004, 94 cutthroat trout were counted during the peak week of spawning activity in the Trout Lake inlet. This represents a slight increase from the 45 spawners counted during the peak week in 2003, but well below the high of 448 spawners counted during the peak week in 1999 (Fig. 13).
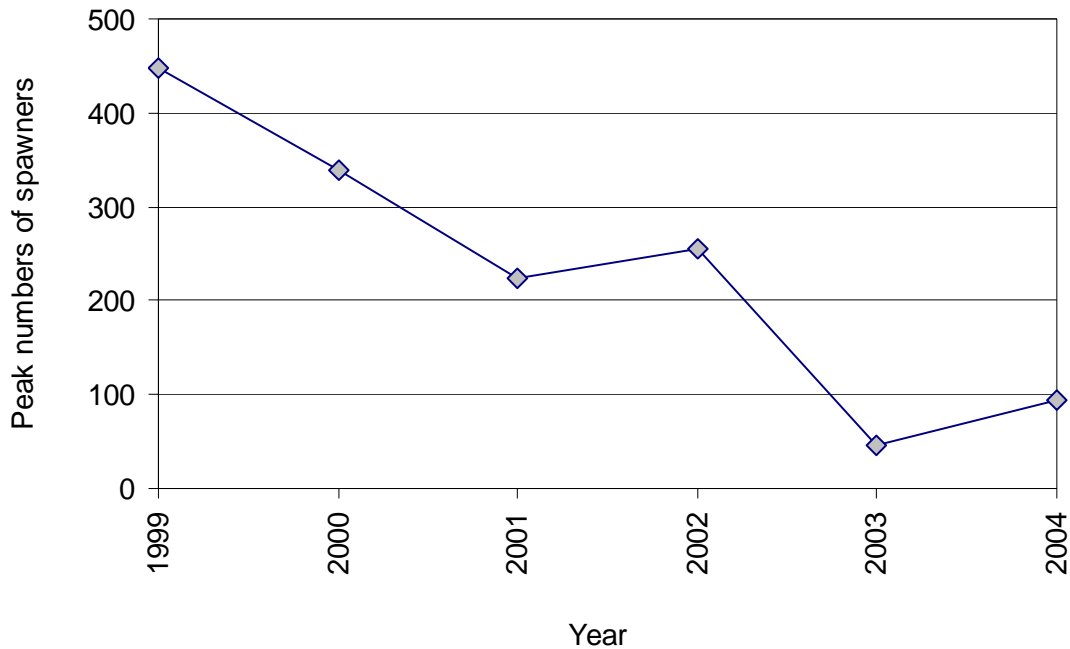
Fig. 13. Number of spawning cutthroat trout counted during the peak week in the Trout Lake inlet, Yellowstone National Park, 1999-2004.

Grizzly Bear Use of Insect Aggregation Sites Documented from Aerial Telemetry and
Observations *(Dan Bjornlie, Wyoming Game and Fish Department; and Mark A. Haroldson,*
Interagency Grizzly Bear Study Team)

Army cutworm moths were first recognized as an important food source for grizzly bears in the GYE during the mid 1980s (Mattson et al. 1991*b*, French et al. 1994). Early observations indicated that moths, and subsequently bears, showed specific site fidelity. These sites are generally high alpine areas dominated by talus and scree adjacent to areas with abundant alpine flowers. Such areas are referred to as "insect aggregation sites." Since their discovery, numerous bears have been counted on or near these aggregation sites due to excellent sightability from a lack of trees and simultaneous use by multiple bears.

Complete tabulation of grizzly presence at insect sites is nearly impossible. Only a few sites have been investigated by ground reconnaissance and the boundaries of sites are not clearly known. In addition, it is likely that the size and location of insect aggregation sites fluctuate from year to year with moth abundance and variation in environmental factors such as snow cover.

Since 1986, when insect aggregation sites were initially included in aerial observation surveys, our knowledge of these sites has increased annually. Our techniques for monitoring grizzly bear use of these sites have changed in response to this increase in knowledge. Prior to 1997, we delineated insect aggregation sites with convex polygons drawn around locations of bears seen feeding on moths and buffered these polygons by 500 m. The problem with this technique was that small sites were overlooked due to the inability to create polygons around sites with 2 or fewer locations. From 1997-1999, the method for defining insect aggregation sites was to inscribe a 1-km circle around the center of clusters of observations in which bears were seen feeding on insects in talus/scree habitats (Ternent and Haroldson 2000). This method allowed trend in bear use of sites to be annually monitored by recording the number of bears documented in each circle (i.e., site).

A new technique was developed in 2000. Using this technique, sites were delineated by buffering only the locations of bears observed actively feeding at insect aggregation sites by 500 m to account for error in aerial telemetry locations. The borders of the overlapping buffers at individual insect sites were dissolved to produce a single polygon for each site. These sites are identified as "confirmed" sites. Locations from the grizzly bear location database from 1 July through 30 September of each year were then overlaid on these polygons and enumerated. The new technique to delineate confirmed sites in 2000 substantially decreased the number of sites described compared to past years in which locations from both feeding and non-feeding bears were used. Therefore, annual analysis for this report is completed for all years using this technique. Areas suspected as insect aggregation sites but dropped from the confirmed sites list using this technique, as well as sites with only 1 observation of an actively feeding bear or multiple observations in a single year, are termed "possible" sites and will be monitored in subsequent years for additional observations of actively feeding bears. These sites may then be added to the confirmed sites list. When possible sites are changed to confirmed sites, analysis is done on all data back to 1986 to determine the historic use of that site. Therefore, the number of bears using insect aggregation sites in past years may change as new sites are added, and data from this annual report may not match that of past reports. In addition, as new actively feeding bear observations are added to existing sites, the polygons defining these sites increase in size and, thus, more overlaid locations fall within the site. This retrospective analysis brings us closer each year to the "true" number of bears using insect aggregation sites in past years.

In 2004, actively feeding grizzly bears were observed on 4 sites classified as possible in past years. Therefore, these sites were reclassified to confirmed and analysis was done on those sites back to 1986. In addition, an observation of a grizzly bear actively feeding in 1 new area resulted in the classification of a new possible insect aggregation site. Some previously known sites were also combined into 1 site because locations demonstrated that they were 1 large site without topographical isolation between them. Therefore, a combination of reclassified sites, a new possible site, and grouping some sites into 1, produced 29 confirmed sites and 21 possible sites for 2004.

The percentage of confirmed sites with documented use by bears varies from year to year, suggesting that some years have higher moth activity than others (Fig. 14). For example, the years 1993-1995 were probably poor moth years because the percentage of confirmed sites used by bears (Fig. 14) and the number of observations recorded at insect sites (Table 16) were low. Overall, insect aggregation site use by grizzly bears decreased in 2004. The total number of observations or telemetry relocations at sites decreased by 27% from 2003 (Table 16). The number of insect aggregation sites used by bears in 2004 decreased to 21 (Table 16) but was above the 5-year average of 19.2 sites/year from 1999-2003. The percent use of insect aggregation sites also decreased in 2004 (Fig. 14).
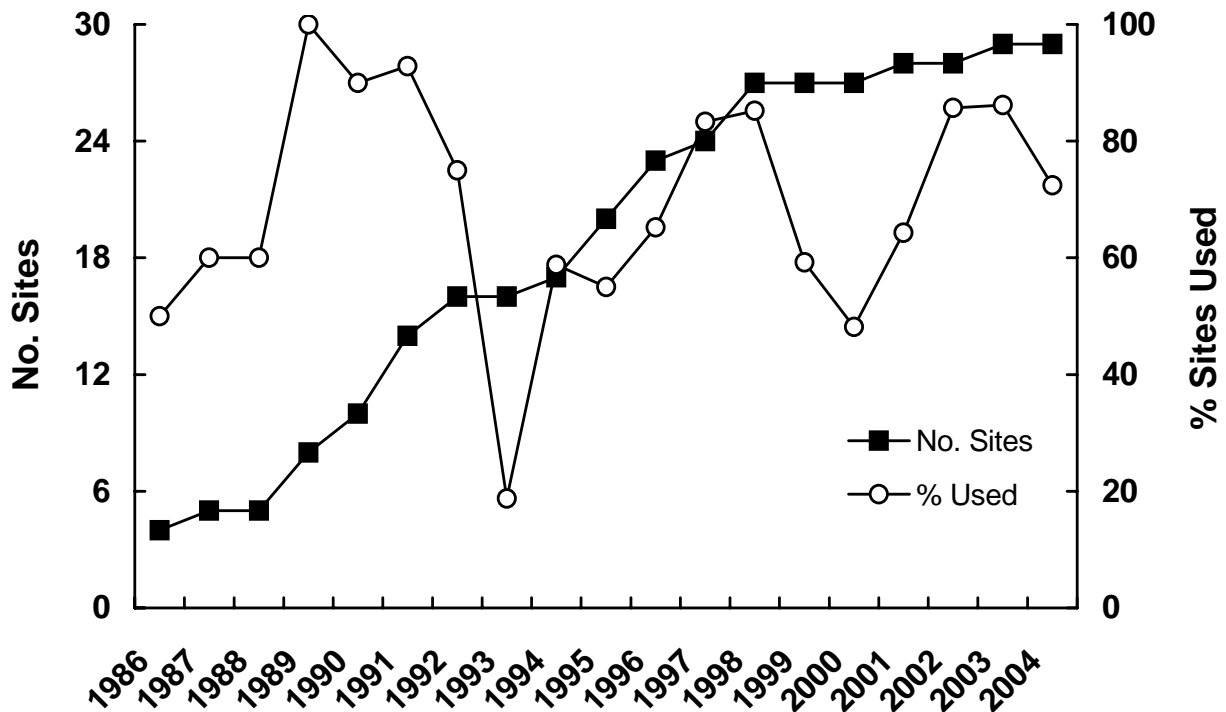
Fig. 14. Annual number of confirmed insect aggregation sites and percent of those sites at which either telemetry relocations of marked bears or visual observations of unmarked bears were recorded, Greater Yellowstone Ecosystem, 1986-2004.

Table 16. The number of confirmed insect aggregation sites in the Greater Yellowstone Ecosystem annually, the number actually used by bears, and the total number of telemetry relocations or aerial observations of bears recorded at each site during 1986-2004.

Year	Number of confirmed moth sites[a]	Number of sites used[b]	Number of locations or observations[c]
1986	4	2	10
1987	5	3	14
1988	5	3	42
1989	8	8	50
1990	10	9	80
1991	14	13	172
1992	16	12	97
1993	16	3	2
1994	17	10	25
1995	20	11	33
1996	23	15	74
1997	24	20	90
1998	27	23	177
1999	27	16	167
2000	27	13	112
2001	28	18	137
2002	28	24	266
2003	29	25	161
2004	29	21	117
Total			1,826

[a] The year of discovery was considered the first year a telemetry location or aerial observation was documented at a site. Sites were considered confirmed every year thereafter regardless of whether or not additional locations were documented.
[b] A site was considered used if ≥1 location or observation was documented within the site that year.
[c] May include replicate sightings or telemetry relocations.

The IGBST maintains an annual list of unduplicated females observed with COY (see Table 4). Since 1986, 554 initial sightings of unduplicated females with COY have been recorded, of which 152 (27%) have occurred at (within 500 m, $n = 131$) or near (within 1,500 m, $n = 21$) insect aggregation sites (Table 17). Peaks in the number of initial sightings recorded at sites correlate with annual trends in the total number of locations at sites ($r^2 = 0.72, P = 0.000$) (Table 16). In 2004, there were 15 unduplicated females with COY observed at insect

aggregation sites, a decrease of 4 from 2003 (Table 17). Of the total observations of unduplicated females with COY, 30.6% (15 of 49) were recorded at insect aggregation sites in 2004, a slight increase from the 5-year average of 28% from 1999-2003.

Table 17. Number of initial sightings of unduplicated females with cubs-of-the-year (COY) that occurred on or near insect aggregation sites, number of sites where such sightings were documented, and the mean number of sightings per site in the Greater Yellowstone Ecosystem, 1986-2004.

Year	Unduplicated females with COY[a]	Number of moth sites with an initial sighting[b]	Initial sightings			
			Within 500 m[b]		Within 1,500 m[c]	
			N	%	N	%
1986	25	0	0	0.0	0	0.0
1987	13	0	0	0.0	0	0.0
1988	19	1	2	10.5	2	10.5
1989	16	1	1	6.3	1	6.3
1990	25	3	3	12.0	4	16.0
1991	24	7	11	45.8	14	58.3
1992	25	4	6	24.0	9	36.0
1993	20	1	1	5.0	1	5.0
1994	20	3	5	25.0	5	25.0
1995	17	2	2	11.8	2	11.8
1996	33	4	4	12.1	7	21.2
1997	31	8	11	35.5	11	35.5
1998	35	11	13	37.1	13	37.1
1999	33	3	6	18.2	7	21.2
2000	37	6	7	18.9	10	27.0
2001	42	6	11	26.2	13	31.0
2002	52	10	14	26.9	17	32.7
2003	38	11	19	50.0	20	52.6
2004	49	10	15	30.6	16	32.7
Total	554		131		152	
Mean	29.2	4.8	6.9	21.0	8.0	23.7

[a] Initial sightings of unduplicated females with COY; see Table 4.
[b] Insect aggregation site is defined as a 500-m buffer drawn around a cluster of observations of bears actively feeding.
[c] This distance is 3 times what is defined as a insect aggregation site for this analysis, since some observations could be made of bears traveling to and from insect aggregation sites.

Survey flights at insect aggregation sites contribute to the count of unduplicated females with COY; however, it is typically low, ranging from 0 to 20 initial sightings/year since 1986 (Table 17). If these sightings are excluded, an increasing trend in the annual number of unduplicated sightings of female with COY is still evident (Fig. 15). This suggests that some other factor besides observation effort at insect aggregation sites is responsible for the increase in sightings of females with cubs.
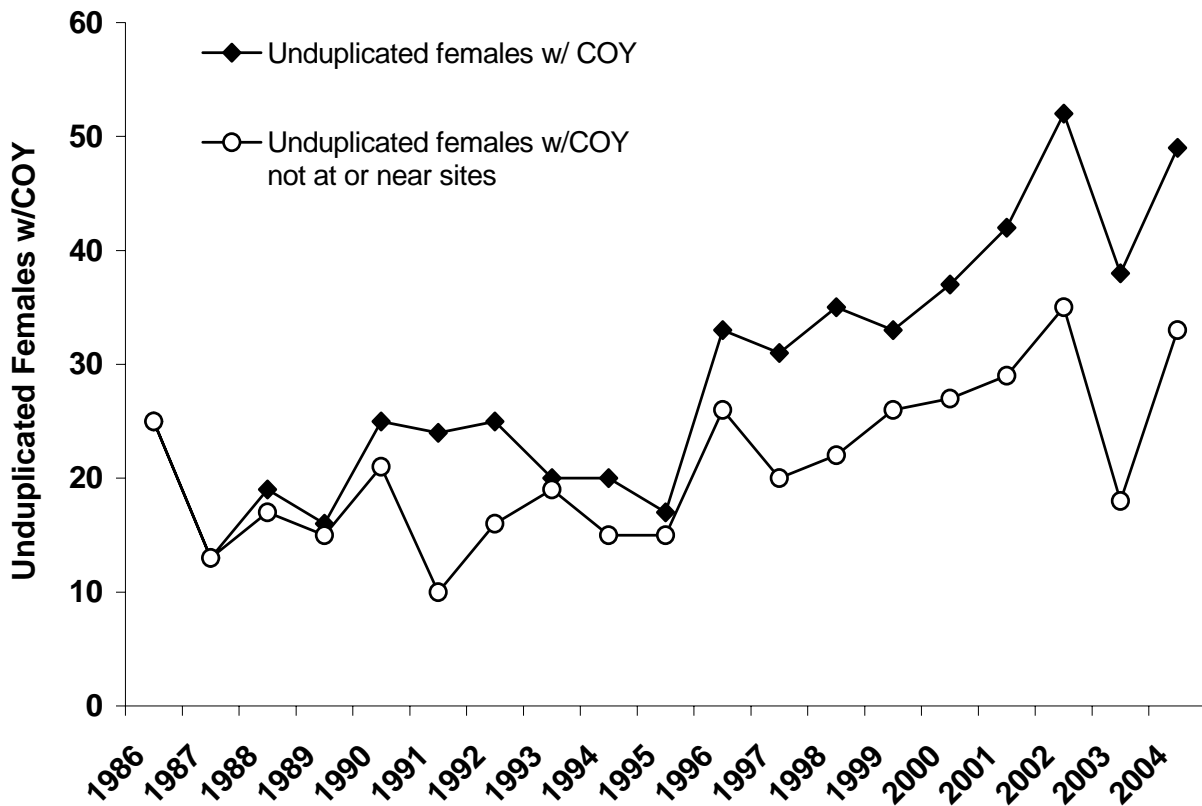
Fig. 15. The total number of unduplicated females with cubs-of-the-year (COY) observed annually in the Greater Yellowstone Ecosystem and the number of unduplicated females with COY not found within 1,500 m of known insect aggregation sites, 1986-2004.

The Ecological Relationship between a Rocky Mountain Threatened Species and a Great Plains Agricultural Pest *(Hillary Robison, Ph.D. candidate, University of Nevada, Reno)*

Project Summary

       Army cutworm moth (ACM) adults migrate from low elevations in the Great Plains and Intermountain West (hereafter low elevations) to the Rocky Mountains and aggregate in high-elevation talus slopes. These ACM aggregations provide an important food resource for grizzly bears. Much is known about the agricultural aspect of the life history of ACMs. However, relatively little is known about their alpine and migratory ecology and their population genetics.

       This study was designed to understand how ACM ecology and population genetics might impact grizzly bear conservation in the GYE. Fieldwork was conducted in high-elevation areas from late June through September and in low-elevation areas from August through October in 1999, 2000, and 2001.

       This study addresses the following: the scale at which ACMs migrate to high-elevation areas; whether ACMs harbor pesticides which could biomagnify in bears; and identification of sites where ACMs may aggregate and bears may feed on them. The results of this study will provide groundwork for further investigations of the affects of moth variability and abundance on grizzly bear fecundity and mortality, as well as provide insights to biologists that may help them make management decisions.

Background and Significance

       Army cutworm moth migration and grizzly bear conservation.--Grizzly bears were first found feeding on ACMs aggregated in talus slopes in the Mission Mountains in 1952 (Chapman et al. 1955). Since this discovery, grizzly bears have been observed feeding on ACMs at several high-elevation sites in Montana and Wyoming (Craighead et al. 1982, Servheen 1983, Mattson et al. 1991*b*, French et al. 1994, White 1996).

       ACMs are an important summer and fall food source for grizzly bears. Grizzly bears excavate the moths from the talus and consume millions of them from July through September (Pruess 1967, Chapman et al. 1955, Mattson et al. 1991*b*, French et al. 1994, White 1996). When compared to other food sources in the GYE, ACMs are the richest food available to grizzly bears (Mealey 1975, Pritchard and Robbins 1990, French et al. 1994, Craighead et al. 1995, White 1996). In 30 days, a grizzly bear feeding extensively on ACMs can consume 47% of its annual energy needs (White 1996).

       When ACMs and whitebark pine nuts (WBPNs) are abundant in the summer and fall, grizzly bears move to high elevations to forage on these rich foods, and in doing so, the bears geographically separate themselves from areas of human activity. Due to this geographic separation, fewer grizzly bear management situations and grizzly bear mortalities are recorded during years when WBPNs and ACMs are abundant or present than during years when they are scarce or absent (Gunther et al. 1993, 1994, 1995, 1996, 1997). WBPN abundance positively correlates with increased grizzly bear fecundity (Mattson et al. 1992). Cyclic crashes in the WBPN crop and damage to whitebark pine from white pine blister rust (*Cronartium ribicola*) increase the importance of understanding the factors influencing ACM presence and abundance at grizzly bear foraging sites.

In 1991 and 1992, researchers estimated that an average of 44% of GYE grizzly bears foraged at ACM aggregation sites in the Absaroka Mountains and that female grizzly bears comprised 40% of these bears (O'Brien and Lindzey 1994).

Female grizzly bear survivorship and reproduction is important to grizzly bear population persistence (Bunnell and Tait 1981, Eberhardt 1990, Craighead and Vyse 1996). Female reproduction depends on adequate pre-hibernation weight gain and fat deposition (Rogers 1987) and is influenced by the quantity and quality of available food (Stringham 1990, McLellan 1994).

The goal of the Endangered Species Act is to recover species and ensure their persistence through time. ACMs and WBPNs are likely important to grizzly bear recovery in the GYE because presence and abundance of these foods influence grizzly bear survival, reproduction, and, in turn, persistence.

Biology of the army cutworm moth.--The ACM is native to North America and ranges from California to Kansas and from Alberta, Canada, to New Mexico. When agriculture began to dominate ACM habitat at the turn of the 20th century, the ACM became an agricultural pest. Adult moths oviposit in loose soil in the fall (Strickland 1916, Burton et al. 1980), and the larvae develop underground. In spring, the larvae surface and feed on emergent plants (e.g., native plants as well as sugar beets, small grains, and alfalfa). The larvae pupate underground, and the adult moths emerge in June and migrate to high-elevation talus slopes in the Rocky Mountains (Pruess 1967). Once ACMs reach the mountains, they remain there from July through September and forage on alpine flower nectar at night and hide in talus during the day (Pruess 1967, French et al. 1994, O'Brien and Lindzey 1994, White 1996). From late August through the beginning of October, the moths migrate back to low elevations and oviposit into soil (Pruess 1967, Burton et al. 1980).

Project Objectives

The main objectives of this study are to determine the scale of ACM origins and, hence, the scale at which factors may influence ACM migration to high-elevation areas where they are fed on by bears; to determine whether ACMs harbor pesticides that could biomagnify in bears; and to identify sites where moths may aggregate and bears may feed on them.

Determining the scale of ACM origins and if ACMs exhibit site fidelity is important because pressures on ACMs in natal areas, whether natural (e.g., weather patterns) or human-caused (e.g., pesticides or habitat loss), may affect moth recruitment and the numbers of adults reaching high-elevation sites used by bears. Genetic techniques can be used to determine the origins of species and to differentiate populations (Bolten et al. 1997, Palsboll et al. 1997, Rankin-Baransky et al. 1997, Eldridge et al. 2001). Because ACMs are small, wide-ranging insects that are not amenable to physical tagging, genetic techniques are well-suited to determining the scale of their origins.

Because grizzly bears eat millions of ACMs and the moths are agricultural pests that are controlled with pesticides, concern exists about whether ACMs contain pesticides that could be toxic to bears (French et al. 1994). Hence, we aimed to analyze ACMs for pesticides and estimate risk to bears.

The conservation strategy for the Yellowstone grizzly bear (USFWS 2003) allows the population to expand into biologically suitable and socially acceptable areas beyond the Primary Conservation Area. The conservation strategy requires use of georeferenced habitat data to aid

in monitoring the 4 major Yellowstone grizzly bear foods (ACMs, cutthroat trout, whitebark pine seeds, and winter-killed ungulates) and to identify habitats into which bears may expand. To this end, we aimed to develop models of high-elevation ACM habitat in the GYE with the purpose of creating a tool with which bear biologists and managers can identify potential ACM habitats into which grizzly bears may expand.

Field Sampling

High elevation.--From mid-July through September 1999-2001 crews used black-light traps at moth aggregation sites to collect ACMs for genetic and pesticide analyses.

ACMs were collected from 6, 9, and 5 sites in 1999, 2000, and 2001, respectively. In total, ACMs were collected from 11 different high-elevation sites, including 9 sites in Wyoming, 1 site in Washington, and 1 site in New Mexico.

Low elevation.--In the late summer and early fall, field crews trapped ACMs with pheromone traps in agricultural lands in Wyoming and Idaho. These efforts were coordinated with the ACM trapping programs of university agricultural extension services in Nebraska, Montana, and South Dakota who sent ACM samples.

Fifteen sites were sampled in 1999 and were re-sampled along with 24 new sites in 2000. All 39 sites were re-sampled in 2001 along with 2 new sites. The sampling effort was expanded in 2000 and 2001 in order to sample a 360-degree radius around the high-elevation study areas.

Methods

The ACM samples collected in 1999 were analyzed by the U.S. Geological Survey's Columbia Environmental Research Center (CERC), in Colombia, Missouri. Samples were analyzed using gas chromatography with electron capture (GCE). A detailed protocol is contained in Lebo et al. (2000). ACMs were not collected for pesticide residue analysis during the 2000 field season. In winter 2000, a question arose as to whether the method used in 1999 was sensitive enough to detect traces of certain pesticides in the ACMs. In 2001, a sample of ACMs was submitted to the Agricultural Experiment Station (AES) Analytical Laboratory at Montana State University-Bozeman, for pesticide screening with GC with tandem mass spectrometry (GC-MS/MS) according to the methods described in Sheridan and Meola (1999).

The genetic data are being analyzed in the Laboratory for Ecological and Evolutionary Genetics and the Nevada Genomics Center at the University of Nevada, Reno. Each of the several thousand moths that have been collected must be individually keyed to species, and the DNA of moths identified as ACMs is extracted. A microsatellite DNA library was developed for the ACM. Eight microsatellite loci (hereafter loci) have been isolated from this library, and polymerase chain reactions (PCRs) are being optimized to amplify these loci. Analyses of the variability at these loci are performed using an Applied Biosystems (ABI) 3730 microsatellite fragment analysis machine and GeneMapper software.

Models are being developed of high-elevation ACM habitat in the GYE using attribute data extracted from GIS layers at bear locations ($n = 490$) that were collected during aerial surveys from 1986-2002.

Results to date

The CERC lab found only non-significant traces of pesticides in the samples analyzed with GCE. The sample analyzed with GC-MS/MS by the Montana State University AES lab came back negative for traces of pesticides (see Appendix D).

Analyses indicate loci are variable within and among populations. Because the genetic data will be influenced by when and where ACMs mate, I am analyzing ACM reproduction.

I am developing presence/random models of ACM habitat. To date, these models indicate elevation, aspect, rate of change in slope, and a few Thematic Mapper bands are important. These models will be displayed as maps showing probabilities of moth habitat in the GYE. Models were generated using a subset of bear locations and are being tested with locations not used in model development (see Appendix E).

As an additional project, I am examining pollen from ACMs to identify which high elevation plants they feed on (Appendix F). Determining plants used by ACMs is important because changes in climate and plant composition may influence the availability of ACM nectar sources.

Project Products

The results of this research will be written as manuscripts and submitted to peer-reviewed journals. A Ph.D. dissertation will be submitted to the University of Nevada, Reno and research results will be presented in a public defense.

Funding sources

Rob and Bessie Welder Wildlife Foundation; Yellowstone Park Foundation; International Bear Association – Bevins Fund; The Wyoming Chapter of the Wildlife Society Memorial Bear Fund; Sigma Xi; American Museum of Natural History; U.S. Forest Service Region 1; Yellowstone National Park Bear Management Office; Greater Yellowstone Coordinating Committee; Wyoming Game and Fish Department; and the U.S. Geological Survey, Northern Rocky Mountain Science Center, Interagency Grizzly Bear Study Team. Donations from the Turner Foundation, Camp Fire Conservation Fund, Bernice Barbour Foundation, Earth Friends, and National Park Foundation were contributed through the Yellowstone Park Foundation.

Cooperators

U.S. Geological Survey, Northern Rocky Mountain Science Center, Interagency Grizzly Bear Study Team; Yellowstone National Park Bear Management Office; U.S. Forest Service Region 1; Montana State University-Bozeman Agricultural Extension agents; and the Wyoming Game and Fish Department.

Whitebark Pine Cone Production *(Mark A. Haroldson and Shannon Podruzny, Interagency Grizzly Bear Study Team; and Roy Renkin, Yellowstone National Park)*

Whitebark pine cone production averaged 6.9 cones/tree on 18 transects with surviving trees read during 2004 (Table 18). Although the ecosystem average indicated poor cone production, good cone crops were evident on some southern transects (Fig. 16). Transect results were consistent with qualitative reports by observers throughout the ecosystem, i.e., poor cone production except in the south.

Table 18. Summary statistics for the 2004 whitebark pine cone production transects in the Greater Yellowstone Ecosystem.

Total			Trees				Transect			
Cones	Trees	Transects	Mean cones	SD	Min	Max	Mean cones	SD	Min	Max
1,002	145	18	6.9	16.0	0	98	55.7	121.0	0	536
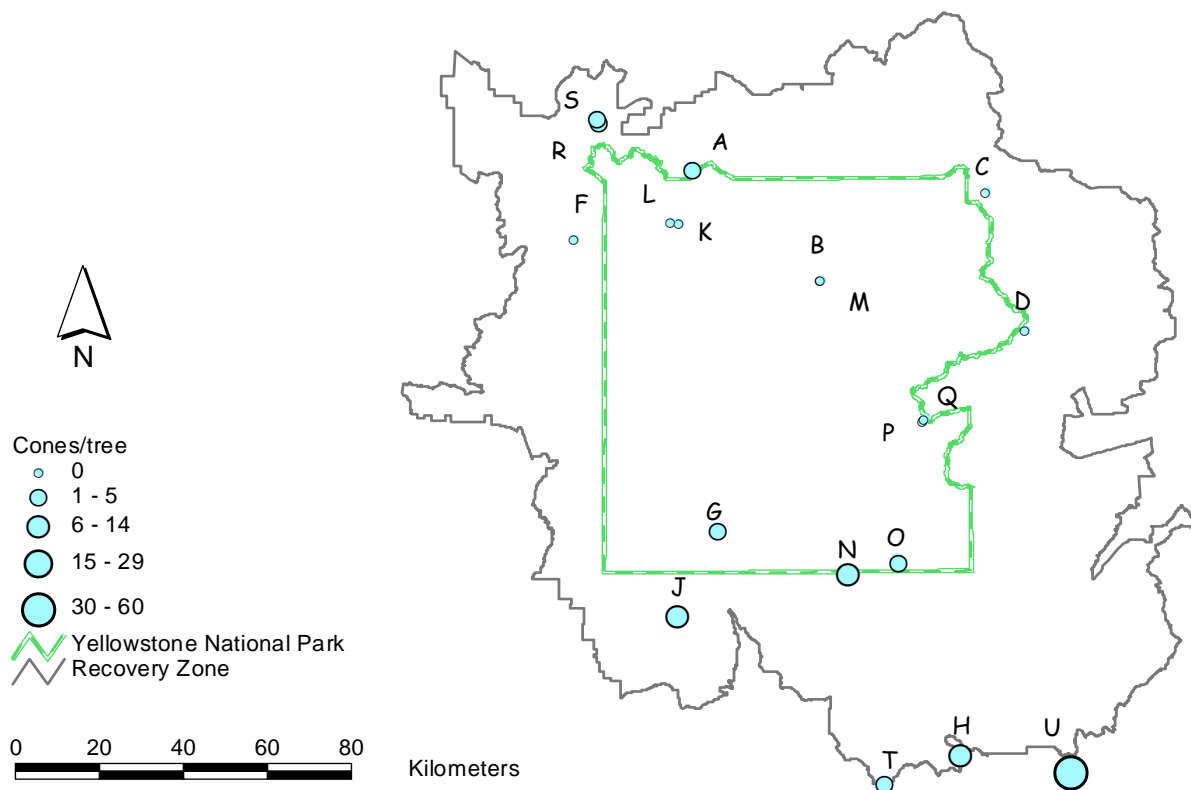
Fig. 16. Average cone production (mean cones/tree) for 18 whitebark pine transects that contained live trees surveyed during 2004 in the Greater Yellowstone Ecosystem. All trees on transect Q were dead from pine beetle and no replacement trees were substituted.

Near exclusive use of whitebark pine seeds occurs during years in which mean cone production on transects exceeds 20 cones/tree (Blanchard 1990, Mattson et al. 1992). Typically, there is a corresponding reduction in numbers of management actions during years of abundant cone availability (Fig. 17). During August-October of 2004, 19 management captures of bears 2 years of age or older (independent) resulted in 15 transports and 4 removals.
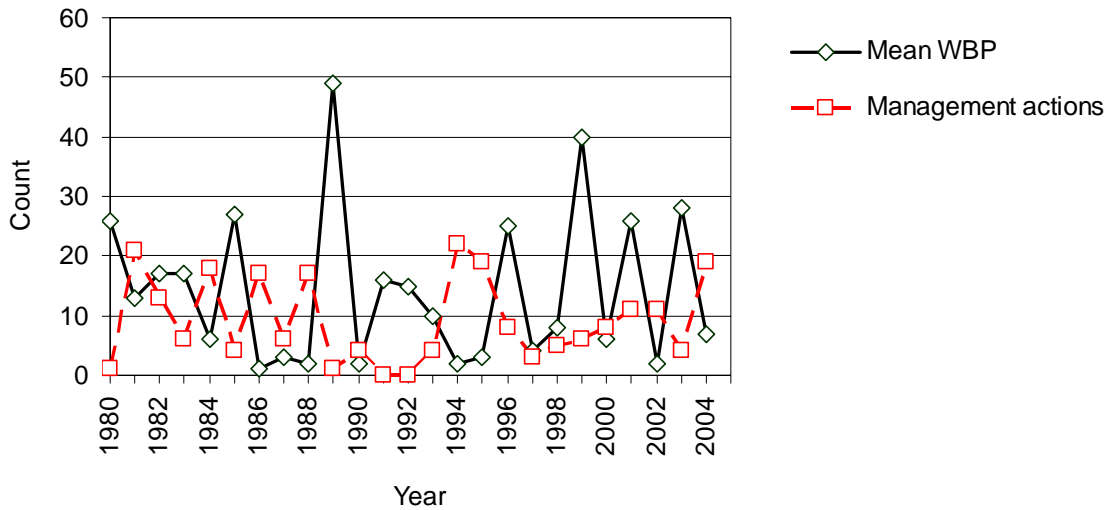
Fig. 17. Mean whitebark pine (WBP) cone production and the number of management actions of grizzly bears older than yearlings during August through October in the Greater Yellowstone Ecosystem, 1980-2004.

Mountain pine beetle (*Dendroctonus ponderosae*) activity continues at high levels throughout the GYE. Eight trees on transect Q (Fig. 16) died from mountain pine beetles between the 2003 and 2004 surveys, resulting in the loss of all trees on this transect since 2002. Other transects hit hard by beetles between the 2003 and 2004 surveys were F (6 dead trees), K (4 dead trees), O (5 dead trees), and P (4 dead trees). Overall, there was 17.6% (31/176) mortality in transect trees between 2003 and 2004, primarily attributed to beetles. Dead trees were generally not replaced during 2003-04, which has resulted in a decline in the total number of trees read from 190 in 2002, to 145 this year. Overall, approximately 8% of whitebark pine stands in YNP have been affected by pine beetle activity. We have no estimates of the extent of whitebark pine mortality on forests surrounding YNP.

Habitat Monitoring

Grand Teton National Park Recreational Use (Steve Cain, Grand Teton National Park)

In 2004, total visitation in Grand Teton National Park was 4,000,697 people, including recreational, commercial (e.g. Jackson Hole Airport), and incidental (e.g. traveling through the Park on U.S. Highway 191 but not recreating) use. Recreational visits alone totaled 2,360,373. Backcountry user nights totaled 28,801. Long-term trends of total visitation and backcountry user nights by decade are shown in Table 19.

Table 19. Average annual visitation and average annual backcountry use nights in Grand Teton National Park by decade from 1951 through 2004.

Decade	Average annual parkwide visitation[a]	Average annual backcountry use nights
1950s	1,104,357	Data not available
1960s	2,326,584	Data not available
1970s	3,357,718	25,267
1980s	2,659,852	23,420
1990s	2,662,940	20,663
2000s[b]	2,490,781	31,727

[a] In 1983 a change in the method of calculation for parkwide visitation resulted in decreased numbers. Another change in 1992 increased numbers. Thus, parkwide visitation data for the 1980s and 1990s are not strictly comparable.
[b] Data for 2000-2004 only.

Yellowstone National Park Recreational Use *(Kerry Gunther, Yellowstone National Park)*

In 2004, total visitation to YNP including non-recreational use was 3,805,797 people. Recreational visits alone totaled 2,868,316. These visitors spent 609,262 user nights camping in developed area roadside campgrounds and 38,268 user nights camping in backcountry campsites. Average annual recreational visitation increased each decade from an average of 333,835 visitors/year in the 1930s to an average of 3,018,624 visitors/year in the 1990s (Table 20). Average annual recreational visitation has decreased slightly the first 5 years (2000-2004) of the current decade, to an average of 2,901,303 visitors/year. Average annual backcountry use nights have been less variable between decades than total park visitation, ranging from 38,268 to 47,395 user nights/year (Table 20). The number of backcountry user nights is limited by both the number and capacity of designated backcountry campsites in the park.

Table 20. Average annual visitation, auto campground user nights, and backcountry user nights in Yellowstone National Park by decade from 1931 through 2003.

Decade	Average annual parkwide total recreational visitation	Average annual auto campground user nights	Average annual backcountry user nights
1931-39	333,835	82,331[a]	Data not available
1940s	552,227	139,659[b]	Data not available
1950s	1,355,559	331,360	Data not available
1960s	1,958,924	681,303[c]	Data not available
1970s	2,243,737	686,594[d]	47,395[e]
1980s	2,381,258	656,093	39,280
1990s	3,018,624	690,044	43,702
2000s[f]	2,901,303	652,102	42,026

[a] Data from 1930-1934.
[b] Average does not include data from 1940 and 1942.
[c] Data from 1960-1964.
[d] Data from 1975-1979.
[e] Backcountry use data available for the years 1973-1979.
[f] Data for the years 2000-2004.

Trends in elk hunter numbers within the Grizzly Bear Recovery Zone plus the 10-mile perimeter area *(Dave Moody, Wyoming Game and Fish Department; Lauri Hanauska-Brown, Idaho Department of Fish and Game; and Kevin Frey, Montana Department of Fish, Wildlife and Parks)*

State wildlife agencies in Idaho, Montana, and Wyoming annually estimate the number of people hunting most major game species. We used state estimates for the number of elk hunters by hunt area as an index of hunter numbers for the Grizzly Bear Recovery Zone plus the 10-mile perimeter area. Because some hunt area boundaries did not conform exactly to the Recovery Zone and 10-mile perimeter area, field personnel familiar with each area were queried to estimate hunter numbers within the Recovery Zone plus the 10-mile perimeter area. Elk hunters were used because they represent the largest cohort of hunters for individual species. While there are sheep, moose, and deer hunters using the Recovery Zone and 10-mile perimeter area, their numbers are fairly small and many hunt in conjunction with elk, especially in Wyoming, where seasons overlap. Elk hunter numbers represent a reasonably accurate index of total hunter numbers within areas occupied by grizzly bears in the GYE.

We generated a data set from all states from 1994 to 2004 (Table 21). Complete data only exists from 1994-1996, 1999, and 2001-2003. Due to the timing of the hunter/harvest survey in Montana (survey, analysis, and final reporting), year specific findings will not available until July in the following year. Overall, hunter numbers have decreased since 1994, especially in Montana and Wyoming. Elk seasons were liberalized the late 1980s through most of the 1990s to reduce elk herds toward their population objective. In the late 1990s, as elk populations reached objective, the number of elk hunters decreased to reduce total harvest, primarily on females. It is felt that hunter numbers in Idaho have not fluctuated significantly over the last 10 years. The increase in hunters starting in 2002 is the result of a new method of calculating hunter numbers.

Table 21. Estimated numbers of elk hunters within the Grizzly Bear Recovery Zone plus a 10-mile perimeter in Idaho, Montana, and Wyoming, for the years 1993-2004.

State	Year										
	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
Idaho[a]	2,682	2,366	3,102	2,869	2,785	2,883	[b]	2,914	3,262	3,285	3,454
Montana	20,942	18,783	18,044	[b]	[b]	16,254	17,329	15,407	17,908	16489	[b]
Wyoming	17,053	17,464	16,283	17,458	15,439	15,727	12,812	13,591	13,709	11,771	10,828
Total	39,777	38,613	37,429			34,864		31,912	34,879	31,905	

[a] Idaho has recalculated hunter numbers. As such, they differ from previous reports.
[b] Hunter number estimates not currently available.

Habitat Use by Grizzly and Black Bears in Grand Teton National Park: First Year Progress Report. (*Charles C. Schwartz, Interagency Grizzly Bear Study Team; Steven Cain, Grand Teton National Park; and Shannon Podruzny, Interagency Grizzly Bear Study Team*)

In May of 2004, the IGBST and National Park Service, Grand Teton National Park (GTNP) initiated a study of grizzly bear-black bear interactions in GTNP. The objectives of the study are to determine habitat use and food habits of grizzly and black bears, evaluate the habitat partitioning of the 2 species, evaluate inter-specific competition between black and grizzly bears for food resources in GTNP, and to examine movements and activity patterns of both species in relation to human activities and the availability of major food resources. This report reviews the progress of location and habitat use data collection efforts for the 2004 field season.

Our general approach to field data collection was to combine the use of advanced GPS technology with traditional field survey methods. We instrumented bears of both species with the latest generation of GPS collars equipped with Spread Spectrum Technology (SST; Podruzny and Schwartz 2004). SST allows for interrogation of the collars to collect stored GPS fixes on demand, which in turn allows for timely investigation of bear-used sites by field crews. This approach allowed us to collect large quantities of spatial data relative to bears' movements, as well as detailed information about the habitat use and feeding activities present at a representative sample of GPS locations.

Study Area

The study is located in the southern part of the GYE, focused within GTNP. This includes the portion of GTNP north of Leigh Canyon and Spread Creek, and adjacent areas of the John D. Rockefeller, Jr. Memorial Parkway and Bridger-Teton and Targhee National Forests. Movements of bears captured in GTNP for this study will determine the final extent of the study area. The terrain and vegetation of the study area are quite variable. The lower elevations included the riparian bottom land of the Snake River and sagebrush (*Artemesia* sp.) covered moraines of the valley floor. Surrounding mountains included subalpine forests and meadows, forest burns of various ages, shrub fields, rocky canyons, and exposed ridgelines. The highest elevations were typified by steep slopes, glaciated peaks, and alpine tundra.

Methods

Capture operations were conducted throughout the field season in GTNP to outfit adult bears of both species with SST collars. Each collar was equipped with a VHF beacon, a store-on-board GPS receiver, a SST transmitter, and a programmable collar release mechanism. The GPS receivers attempted to fix locations at regular intervals. The inter-fix interval was preset for each collar, and was calculated to maximize battery life according to transmitter weight and the amount of time a bear was expected to wear the collar. Intervals ranged from 35 minutes between fixes for adult male collars to 105 minutes for female black bear collars. Male collars were programmed to drop off at the end of the first season; female collars were programmed to release at the end of the following season.

All fix attempts were permanently stored in the collar's receiver, and the SST transmitters were available for downloading copies of the data during 2 mornings each week. We attempted to download location data from each collar via a fixed-wing aircraft once per week. When

conditions did not allow flying, we occasionally downloaded data using a high-gain antenna on the ground if bears were close enough to accessible areas. The downloaded data were imported into a database, and the locations translated into Universal Trans-Mercator (UTM) Zone 12N NAD83 coordinates.

From these data, we selected locations on which to perform field reconnaissance. At first opportunistically, then randomly as the number of marks permitted, we chose the order of bears to sample. Field crews would attempt to visit all successful fixes recorded for each bear in a 24 hour period. Location data were uploaded into personal GPS units for navigation to the sites. We attempted to follow 2-7 days behind the bears to maximize detectability of sign without disturbing the animals. We would leave a survey area if VHF signals indicated that the bear was still present.

At each UTM site, we performed a detailed reconnaissance within a 15 m radius. We recorded site visit data in 3 levels of detail depending upon what we found at the site. For all sites, we recorded descriptive and quantitative data on the physical and vegetal characteristics, including habitat type and forest cover information. We recorded presence or absence of bear sign and made general notes about the site. If bear sign was found, we completed a more detailed "Level 2" plot. This included specific measurements of daybeds, rub trees, and feeding activity as well as percentages of ground cover (foliage, shrubs, deadfall, etc.) as determined by 4 10-m point-line intercept transects. If the bear had been consuming plant foods, we went on to complete a "Level 3" plot. This consisted of measuring vegetation and specific bear foods within 10 0.1-m^2 Daubenmire plot frames laid out along the cover transect tapes.

We collected samples of scat at visited sites for food habits analysis. A small portion of each scat was collected for species determination via mitochondrial DNA (mtDNA) analysis. When multiple scats occurred at daybed sites, only 1 mtDNA sample was collected for that group of scats. In areas near used sites, we collected samples of bear foods for stable isotope and nutritional analysis (Robbins et al. 2004).

Preliminary Results

Capture crews deployed collars on 3 adult female grizzly bears, 5 adult female black bears, and 3 adult male black bears during the field season. None of the females were accompanied by young. Two of the grizzly bears were not captured until late in the summer. One female black bear was killed by an unmarked grizzly bear 3 weeks following her capture. One male black bear shed his collar mid-way through the season. We tracked the remaining bears throughout the field season. The collars attempted to collect 24,087 fixes while on active bears. From these attempts, 18,701 locations of active bears were determined. GPS fix success rates are detailed in Table 22. Collars deployed on female grizzlies had the highest rates of successful fixes (82.5%) and the highest proportion of 3D fixes (61.4%; where elevation was not estimated from previous fixes). Male black bears had the lowest success rates, 75.8% and 43.9%, respectively.

Table 22. Global Positioning System fixes attempted and success rates from Spread Spectrum Technology collars deployed on 8 black and 3 grizzly bears in Grand Teton National Park, 2004. Attempted fixes reported only for active, not denned bears.

	Attempted fixes	Successful fixes		3D fixes	
	n	n	%	n	%
Female black	7,672	5,855	76.32	2,641	45.07
Male black	10,431	7,909	75.82	3,474	43.92
Female grizzly	5,984	4,937	82.50	3,031	61.39
All collars	24,087	18,701	77.64	9,146	48.91

Field crews visited 1,148 bear locations, encompassing 6% of successful fixes and 88 bear/date combinations. Bear sign was found at 826 (72%) of these locations. Sign included feeding activity, daybeds, scats, and tracks. Evidence of feeding activity was found at 565 locations (Table 23). Grizzly bears were most commonly feeding on carcasses, digging roots, and grazing on vegetation (Table 24). Black bears were most often feeding on insects, browsing berries, or grazing (Table 24). Whitebark pine cone production was poor in 2004 (see *Whitebark Pine Cone Production*). We expect food habits of both species to vary in years of good whitebark pine cone production.

Table 23. Feeding activities observed at 1,148 GPS locations of black and grizzly bear locations visited in and near Grand Teton National Park, 2004. More than 1 type of feeding activity may have been found at any location.

Feeding activity	Black bears		Grizzly bears		Total	
	n	%	n	%	n	%
Carcasses	12	2.13	38	26.95	50	8.85
Roots	0	0.00	42	29.79	42	7.43
Whitebark pine	1	0.18	3	2.13	4	0.71
Rodent caches	0	0.00	2	1.42	2	0.35
Grazing	126	22.38	38	26.95	164	29.03
Insects	222	39.43	11	7.80	233	41.24
Berries	187	33.21	6	4.26	193	34.16
Cambium	10	1.78	1	0.71	11	1.95
Other	5	0.89	0	0.00	5	0.88

We collected 187 scats, including 139 at black bear locations, 47 at grizzly bear locations, and 1 unique scat found en route to black bear locations. Analysis of scat contents and mtDNA are in progress. Twenty-seven samples of bear foods were collected. We also collected 52 samples of shed hair at bear locations.

Table 24. Most common species fed upon by black and grizzly bears at 565 GPS locations, Grand Teton National Park, 2004.

Type of feeding activity	Common name of species used	Genus
Carcasses	Elk	*Cervus*
	Mule deer	*Odocoileus*
	Domestic cow[a]	*Bos*
Roots	Yampa	*Perideridia*
	Oniongrass	*Melica*
	Biscuitroot	*Lomatium*
	Angelica	*Angelica*
Caches	Various roots	
Whitebark pine	Whitebark pine	*Pinus*
Grazing	Grasses and sedges	various
	Fern-leaved lovage	*Ligusticum*
	Sticky geranium	*Geranium*
	Bracted lousewort	*Pedicularis*
	Dandelion	*Taraxacum*
	Cow parsnip	*Heracleum*
	Fireweed	*Epilobium*
Insects	Ants	
	Other insects	
Berries	Huckleberry	*Vaccinium*
	Serviceberry	*Amalanchier*
	Buffaloberry	*Sherpherdia*
	Grouse whortleberry	*Vaccinium*
	Rose	*Rosa*
	Chokecherry	*Prunus*
Cambium	Lodgepole pine	*Pinus*
	Englemann spruce	*Picea*

[a]Cattle died of larkspur poisoning.

Future Directions

We will conduct 2 more field seasons following the protocols established in 2004. Results of scat content analysis will help guide food sampling efforts in subsequent field seasons. Data will continue to be maintained in Geographic Information System and Microsoft Office databases. Final analyses and reports will be completed in 2007.

Grizzly Bear-Human Conflicts in the Greater Yellowstone Ecosystem *(Kerry A. Gunther, Yellowstone National Park; Mark T. Bruscino, Wyoming Game and Fish Department; Steven L. Cain, Grand Teton National Park; Kevin Frey, Montana Fish, Wildlife and Parks; Lauri Hanauska-Brown, Idaho Department of Fish and Game; and Mark A. Haroldson and Charles C. Schwartz, Interagency Grizzly Bear Study Team)*

Conservation of grizzly bears in the GYE requires protecting sufficient habitat and maintaining sustainable levels of human-caused mortality. Most human-caused grizzly bear mortalities are directly related to grizzly bear-human conflicts (Gunther et al. 2004). Grizzly bear-human conflicts also erode public support for grizzly bear conservation. To effectively allocate resources for implementing management actions designed to prevent grizzly bear-human conflicts from occurring, land and wildlife managers need baseline information as to the types, causes, locations, and trends of conflict incidents. To address this need, we record all grizzly bear-human conflicts reported in the GYE annually. We group conflicts into 6 broad categories using standard definitions described by Gunther et al. (2000, 2001). To identify areas with concentrations of conflicts, we calculated the 80% isopleth for the distribution of conflicts using the fixed kernel estimator in the Animal Movements (Hooge and Eichenlaub 1997) extension for ArcView GIS (Environmental Systems Research Institute 1999).

The frequency of grizzly bear-human conflicts is inversely associated with the abundance of natural bear foods (Gunther et al. 2004). When native bear foods are of average or above average abundance there tend to be few grizzly bear-human conflicts. When the abundance of native bear foods is below average, grizzly bear-human conflicts increase, especially during the season when bears are hyperphagic (Gunther et al. 2004). In 2004, the availability of high quality, concentrated bear foods was poor during the spring, estrus, and late hyperphagia seasons, but good during early hyperphagia. During spring, the availability of winter-killed ungulate carcasses was below average in thermally influenced ungulate winter ranges (see *Spring Ungulate Availability)*. During estrus, the numbers of spawning cutthroat trout were below average in Yellowstone Lake tributaries (see *Spawning Cutthroat Trout*). However, in the Montana and YNP portions of the ecosystem, biscuit root (*Lomatium cous*) was abundant during estrus and grizzly bears made extensive use of it. Early hyperphagia was characterized by wet rainy conditions that kept vegetal bear foods succulent late into the season. During early hyperphagia, army cutworm moths were also present and attracted large numbers of grizzly bears to high-elevation insect aggregation sites on the east side of the ecosystem. The abundance of whitebark pine seeds during late hyperphagia was lower than average (see *Whitebark Pine Cone Production*) which caused bears to seek yampa (*Perideridia gairdneri*) roots and false truffles (*Rhizopogon* spp.) at lower elevations.

There were 144 grizzly bear-human conflicts reported in the GYE in 2004 (Table 25, Fig. 18). These incidents included bears obtaining anthropogenic foods (40%, *n* = 58), damaging property (29%, *n* = 41), killing livestock (24%, *n* = 35), injuring people (6%, *n* = 8) and obtaining fruits and vegetables from gardens and orchards (1%, *n* = 2). Most (55%, *n* = 79) of the conflicts occurred on public land administered by the U.S. Forest Service (51%, *n* = 73), National Park Service (3%, *n* = 4), and the state of Wyoming (1%, *n* = 2). Forty-five percent (*n* = 65) of the conflicts occurred on private land in the states of Wyoming (35%, *n* = 50) and Montana (10%, *n* = 15). Fifty-seven percent (*n* = 82) of the conflicts occurred inside and 43% (*n* = 62) outside of the Recovery Zone. The conflict distribution map constructed using the fixed kernel 80% conflict distribution isopleth identified 2 areas where most grizzly bear-human

conflicts in the GYE occurred in 2004 (Fig. 18). These 2 areas contained 123 (85%) of the 144 conflicts. The 2 areas where most conflicts occurred included: (1) the Wyoming portion of the ecosystem where bears ate garbage, human foods, livestock feed, and pet foods, and killed cattle and sheep, and (2) the Yellowstone River/Gardiner Montana area where bears killed chickens and ate vegetables from gardens.

The below average abundance of most concentrated high-quality bear foods during the spring, estrus, and late hyperphagia seasons in 2004, was likely partially off-set by the good abundance of biscuit root, yampa root, and truffles in portions of the ecosystem. In 2004, property damages and bear-inflicted human injuries were higher than the long-term averages recorded from 1992-2003 (Table 26). The numbers of incidents of livestock depredation, bears obtaining anthropogenic foods, and damage to gardens, orchards, and beehives were similar to the long-term averages recorded from 1992-2003 (Table 26).

Table 25. Number of incidents of grizzly bear-human conflicts reported within different land ownership areas in the Greater Yellowstone Ecosystem, 2004.

Land owner[a]	Total conflicts	Human injuries	Property damages	Anthropogenic foods	Gardens/ orchards	Beehives	Livestock depredations
BLM	0	0	0	0	0	0	0
BDNF	0	0	0	0	0	0	0
BTNF	19	1	7	4	0	0	7
CNF	0	0	0	0	0	0	0
CTNF	0	0	0	0	0	0	0
GNF	5	1	3	1	0	0	0
GTNP/JDR	0	0	0	0	0	0	0
ID-private	0	0	0	0	0	0	0
ID-state	0	0	0	0	0	0	0
MT-private	15	0	4	4	2	0	5
MT-state	0	0	0	0	0	0	0
SNF	49	4	15	20	0	0	10
WY-private	50	0	10	27	0	0	13
WY-state	2	0	1	1	0	0	0
YNP	4	2	1	1	0	0	0
Total	144	8	41	58	2	0	35

[a] BLM = Bureau of Land Management, BDNF = Beaverhead-Deerlodge National Forest, BTNF = Bridger-Teton National Forest, CNF = Custer National Forest, CTNF = Caribou-Targhee National Forest, GNF = Gallatin National Forest, GTNP/JDR = Grand Teton National Park/John D. Rockefeller, Jr. Memorial Parkway, ID = Idaho, MT = Montana, SNF = Shoshone National Forest, WY = Wyoming, YNP = Yellowstone National Park.
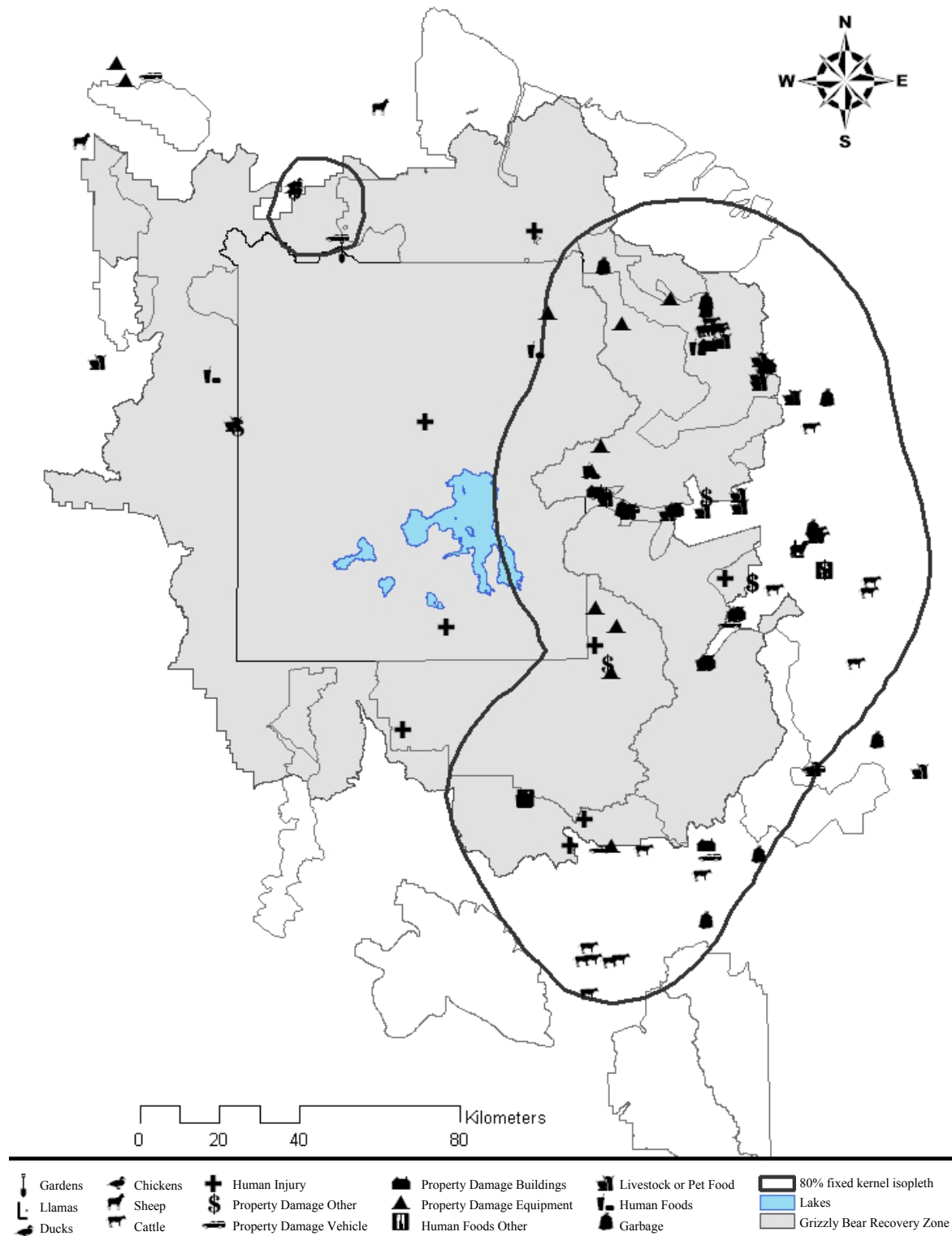
Fig. 18. Locations of different types of grizzly bear-human conflicts reported in the Greater Yellowstone Ecosystem in 2004. Polygons represent concentrations of conflicts identified using the 80% fixed kernel isopleth. The shaded area represents the Yellowstone Grizzly Bear Recovery Zone.

Table 26. Comparison between the number of incidents of different types of grizzly bear-human conflicts in 2004 and the annual average number of conflicts recorded from 1992-2003 in the Greater Yellowstone Ecosystem.

Type of conflict	1992-2003 Mean ± SD	2004
Human injury	4 ± 3	8
Property damage	18 ± 12	41
Anthropogenic foods	54 ± 42	58
Gardens/orchards	5 ± 3	2
Beehives	3 ± 4	0
Livestock depredations	51 ± 20	35
Total conflicts	135 ± 59	144

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APPENDIX A

FINAL REPORT
For Stage 1

**Mapping Whitebark Pine Distribution
Throughout the Greater Yellowstone Ecosystem**

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Greater Yellowstone Network, Inventory and Monitoring Program
Rocky Mountains Cooperative Ecosystem Studies Unit (RM-CESU)
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Background

In the fall of 2003, the U.S. Geological Survey's Northern Rocky Mountain Science Center (NRMSC) initiated an effort to map the distribution of whitebark pine (*Pinus albicaulis* Engelm.) throughout the Greater Yellowstone Ecosystem (GYE). This research initiative was sponsored by the USGS Interdisciplinary Science Support Activities (ISSA) program with the larger aim of promoting the use of remote sensing throughout the USGS. This funding opportunity allowed the NRMSC to purchase necessary software and to compile existing ground data generated by U.S. Forest Service and National Park Service units throughout the ecosystem. It also provided an impetus to forge a constructive collaboration between the NRMSC and Remote Sensing Laboratory at Montana State University. The purpose of this report is to document the results of this project and its potential ramifications to the health and viability of the grizzly bear (*Ursus arctos*) population within the GYE.

Justification and Rationale

Whitebark pine seeds have long been identified as the most significant vegetative food source for grizzly bears in the GYE and are, hence, a crucial element of suitable grizzly bear habitat (Lanner and Gilbert 1994). Whitebark pine also serves as a keystone species in that its presence increases the biodiversity of both plant and animal communities throughout the ecosystem (Tomback and Kendall 2001). The overall health and status of whitebark pine is currently threatened by infestation by mountain pine beetle (*Dendroctonus ponderosae*) and the spread of white pine blister rust (*Cronartium ribicola*). The mapping of whitebark pine distribution is integral to the success of the long-term monitoring of whitebark pine since, before we can study, understand, and mitigate the mechanisms driving destructive agents of whitebark pine, we must first know its distribution across the landscape.

Methods

Imagery and software.--Landsat 7 Enhanced Thematic Mapper Plus (ETM+) satellite imagery was used as the primary mapping data source for reasons of cost and computational efficiency. Each ETM+ image covers a 170 km by 185 km area and cost $600 per image, making mapping at regional scales highly cost effective. Seven ETM+ scenes covering the core of the GYE (Fig.1) were provided with geometric and radiometric corrections by the EROS Data Center, Sioux Falls, South Dakota. Although complete coverage was initially provided for July 2002 (summer scenes) and September 1999 (fall scenes), the summer scenes were eliminated from the analysis due to significant cloud coverage in the southern portion of the ecosystem. Also, since cambial growth, bud breakage and shoot growth of whitebark pine typically occur in late-May to mid-June (Weaver 2001), the July images lacked phenological differences in spectral signatures among the various conifer species in the GYE. ERDAS Imagine Professional (v.8.7), an image processing software, was used in conjunction with the statistical software package S-Plus and, at the recommendation of Collin Homer with Earth Resources Observation & Science (EROS) Data Center, a statistical software package called See5.
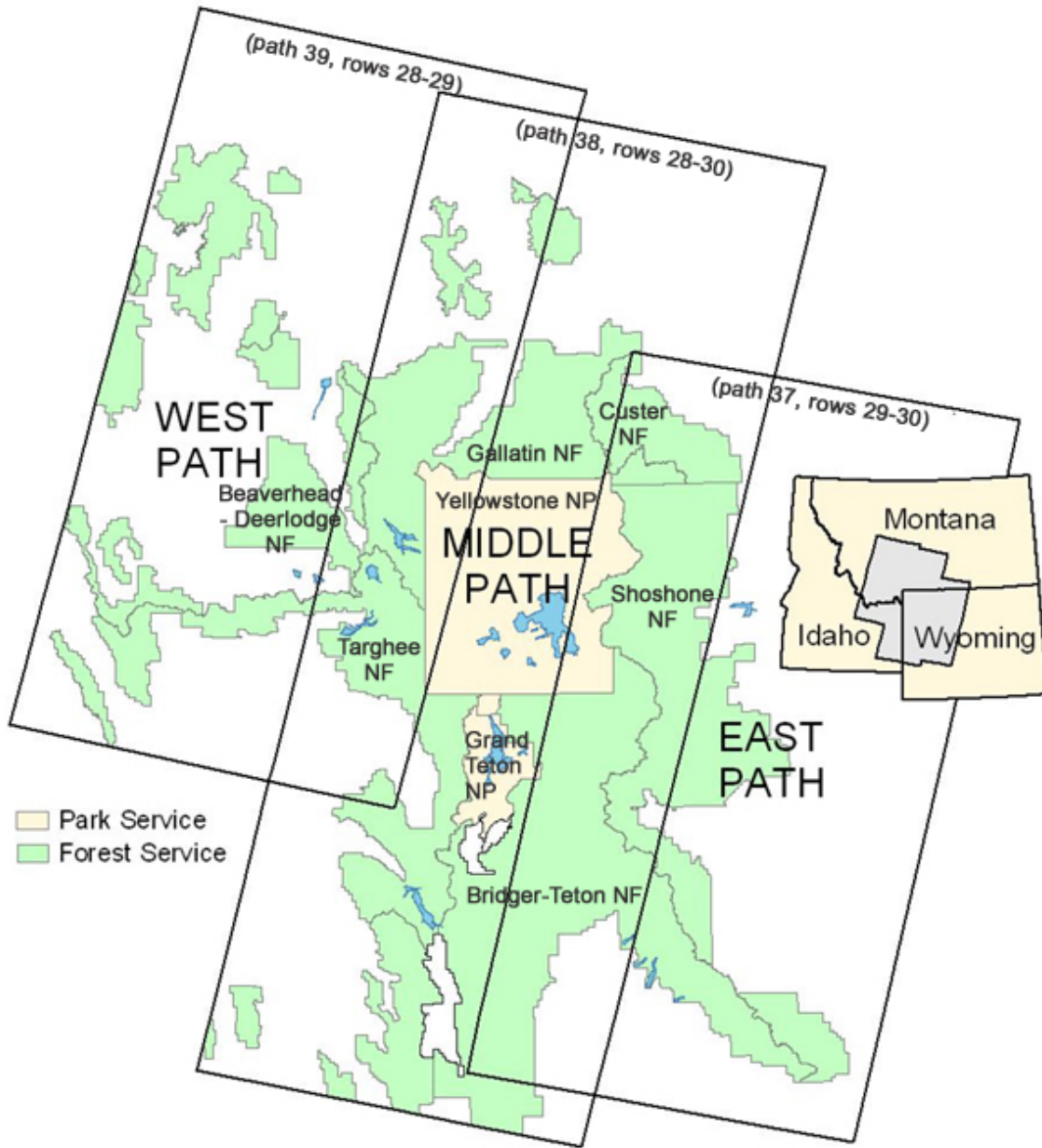
Fig. 1. Study area classification divisions based on east, west, and middle paths of Landsat ETM+ satellite imagery.

Training/validation data development.--Reference data consisting of known whitebark pine locations with accurate ground coordinates were necessary to "train" the software spectral pattern-recognition process to identify areas in the imagery that match the known or observed whitebark pine locations. Substantial time and effort was allocated to compile and generate reliable reference data that represented the complete variation in spectral response and terrain attributes associated with whitebark pine on the landscape.

Our initial expectations were to rely heavily on ground truth data collected by U.S. Forest Service and National Park Service personnel in conjunction with their standard timber-stand exams, vegetation plots, soil surveys, and other field activities where ground information was collected, since the scope of this project precluded adequate time and resources for extensive field work. The agencies responded well to our requests for data hence we were able to compile a large pool of vegetation data that collectively constituted a fairly sufficient representation of the spatial complexities of the ecosystem. These data were naturally populated with varying degrees of detail in descriptive content regarding vegetative coverage because they were initially collected to meet different purposes. We consequently had to employ the "least-common-denominator" approach and apply a binary response variable: either whitebark or non-whitebark. The presence of whitebark for our study was indicated by a dominant component of whitebark pine in forest canopy.

Aside from the variable descriptive content, the data also exhibited varying degrees of spatial accuracy. Since many of the data were collected before GPS units were readily available, various methods were used in estimating ground locations resulting in disparate degrees of spatial accuracy. A significant component of the data collected via GPS also had considerable error (up to +/- 300 m) due to selective availability and the lack of post-differential correction. Due to the 30-m resolution of Landsat imagery, a comparable locational accuracy of the reference data was a necessary requirement. In order to verify the spatial reliability of the data, random checks were performed by overlaying each datum on top of geo-referenced digital orthographic quad (DOQ) photos. From this, we determined that a substantial number of data points, both field-estimated and GPS-acquired, lacked necessary spatial accuracy. For example, data points allegedly representing a specific conifer class were often found either at the very edge of forest stands, in open meadows, or even within lakes! Preliminary analyses suggested that these inconsistencies in the data, if left uncorrected, resulted in poor predictive power of models (Appendix A1). Consequently, we examined each of 9000+ records against underlying DOQs. Points with questionable spatial accuracy were flagged. With the aid of aerial photos from government archives and/or expertise of government personnel with substantial field experience, points were shifted over distances ranging from ten to a few hundred meters to their most probable location. Points that could not be corrected with a high degree of certainty were eliminated from the analysis. Unfortunately, this accounted for 29% of the ground points.

While correcting training data locations with aerial photographs, other nearby whitebark pine stands often could be identified in photographs. Such stands were then located on DOQs within the GIS database, and new geo-referenced points were generated within whitebark pine stands. To maximize the information extracted from air photos, one of the authors (Landenburger) was trained in the skills of air photo interpretation by Don Despain (Scientist; USGS, NRMSC), who has over 30 years of experience in photo interpretation. Under Despain's tutelage Landenburger learned to differentiate whitebark pine from other conifer species by examining the structure, shape, color and texture of canopy, the shape of shadows, and ancillary layers of information such as elevation and aspect. Once proficient, Landenburger then traveled to Forest Service offices across the ecosystem to access their photo archives and to consult with their field experts. Despain provided occasional random checks on some of the photo interpretations to ensure an acceptable degree of confidence in Landenburger's interpretations. This method, although very time consuming, allowed for verification of existing data, and generation of new data that added tremendously to the overall integrity and quantity (8000+ points) of our reference data collection. From this final compilation of data points, 85% were

selected randomly for use in the classification "training" process, while the remaining 15% was reserved for accuracy assessment.

  Predictor variables.--Spectral and spectrally-derived predictor variables (also referred to as explanatory or independent variables) used in this analysis included (1) at-satellite reflectances scaled to 8-bit values by EROS Data Center for the six ETM+ reflective bands (the thermal band was not provided), (2) re-scaled at-satellite tasseled cap brightness, greenness, and wetness values (Huang et al. 2002), (3) principal component data values calculated in ERDAS Imagine for all six bands, and (4) normalized difference vegetation index (NDVI), where NDVI = (near infrared – red)/(near infrared + red). In addition, ancillary data considered to have strong predictive powers for whitebark pine occurrence included latitude and three data layers derived from the USGS 30-m National Elevation Dataset Digital Elevation Models including, elevation, slope, and aspect. Aspect was made continuous by taking the cosine of aspect in radians, and stretching it to an 8-bit value by adding 1 and multiplying the sum by 200. For computational reasons, latitude was generated from a 1-km regular grid and then re-sampled to 30 m (Δlatitude $\cong$ 0.00011 degrees per km).

  Statistical models:--In this study logistic regression and a rule-based method, classification tree analysis (CTA), were used for comparison to generate classified images depicting the distribution of whitebark pine. Logistic regression has been shown to be an appropriate statistical tool when the response variable is binary in nature (Bricklemyer et al., 2002). The goal of logistic regression is to determine the best fitting model to describe the relationship between a dichotomous characteristic (i.e., presence/absence of whitebark pine) and a set of independent predictor variables. A forward/backwards stepwise logistic regression algorithm in S-Plus was employed to generate the coefficients of a linear equation predicting the *logit transformation* of the probability of whitebark pine presence:

$$\text{logit (p)} = b_o + b_1 X_1 + b_2 X_2 \ldots + b_n X_n$$

where p is the probability of presence of whitebark pine, the logit of p is expressed as a linear combination of the *n* explanatory variables X_n, and the regression coefficients (b_n) are a measure of the predictive capability of the independent variables (Dallal 2001). Probability can be calculated from the logit since the logit is defined as the natural log of the odds:

$$\text{logit(p)} \equiv \ln \left[\frac{p}{1-p} \right] \equiv \ln \left[\frac{probability\ of\ presence}{probability\ of\ absence} \right]$$

Solving for the probability yields:

$$P = \left[\frac{1}{1 + e^{-\log it(p)}} \right] = \left[\frac{1}{1 + e^{-(b_0 + b_1 X_1 + b_2 X_2 \cdots + b_n X_n)}} \right]$$

  The model was determined to be fitted when the Akaike's Information Criterion, *AIC* ≤ 2 (Burnham and Anderson 1998). The resulting logistic equation was then applied to the input predictor variables using the *Model Maker* module in ERDAS Imagine. The probability of whitebark pine presence was thereby calculated for each pixel in the image. The resulting map is

an image with pixel values ranging from 0 to 1 where a value of 1 indicates a predicted probability of 100% that whitebark pine is present. A threshold criteria of $p > 40$, based on maximizing overall class accuracy, was used to segregate whitebark pine from non-whitebark pine.

CTA is also well suited to modeling response variables by producing predicted target classifications based on a series of if-then conditions (tree nodes). CTA has been shown to be an effective tool for classification of remotely sensed data in conjunction with ancillary data (Lawrence and Wright 2001). CTA examines the input reference observations (populated with predictor variable values) and recursively partitions the data based on binary splits of individual predictor variables such that deviance in the response variable is minimized (Breiman et al. 1984). By following the paths of the resulting tree, one can determine a series of rules predicting classes. These rules were entered into the ERDAS Imagine *Knowledge Engineer* module and applied to the input spectral and ancillary predictor variables for the entire study area. The resulting image maps the response variable, in our case presence or absence of whitebark pine.

In this study two distinct CTA splitting algorithms were used. The first algorithm was a standard class probability splitting rule employed in the S-Plus decision-tree function. In this method the partitioning algorithm essentially splits at every possible value of every predictor and chooses the split that minimizes deviance while maximizing node homogeneity. If all observations were classified correctly at a terminal node the deviance at that node was zero. The second method used in this study was the entropy splitting algorithm employed in See5, a proprietary software package produced by Rulequest. In this software program, the decision tree grows by applying a gain ratio criteria to recursively parse the training observations into homogeneous subsets (Quinlan 1993, Huang et al. 2001). One distinct advantage to the See5 program was the option for boosting, a technique reported to significantly reduce the training error and to boost or enhance the classification accuracy (Freund and Schapire 1999, Schapire 1999). Boosting generated a user-specified number of classification trees such that each successive tree attempted to correct misclassification of the previous tree (Lawrence et al. 2004). At each iteration, the training samples were re-assigned weights with misclassified data given greater weight. The final predicted classification was based on a plurality vote from the complete set of classification trees. See5 provides a default of 10 boosts and a maximum of 99 boosts, both of which were evaluated in this study.

Generating a seamless map.--Classifications were conducted separately on three sets of images covering the study area (Fig. 1): the middle-path (path 38, rows 28-30), the east-path (path 37, rows 29-30), and the west-path (path39, rows 28-29). Classification was first performed on the middle path yielding high accuracy rates that justified using the classification results of the middle-path in areas of path-overlap to identify supplemental training samples for the classification of the east- and west-paths (Parmenter et al. 2003). This method was used to ensure a smooth and seamless transition across the final merged classified image. In each overlap area, 4000 random points were generated and populated with the corresponding classification codes from the middle-path results (1 for whitebark pine, 2 for non-whitebark pine). These points were then added to the training samples for the east- and west-paths respectively.

Results

A total of 15,110 training data points, excluding random points generated in the image overlap areas, were compiled for this analysis. Photo-interpreted points comprised 54% and government field data comprised the remaining 46%. Five different predictive models were run on the middle-path to determine which yielded the best results and hence was the most appropriate for the entire study area. User's and producer's class accuracies as well as overall class accuracies (Table 1) for the five statistical methods indicate that CTA with boosting preformed the best.

Table 1. Comparative accuracies for classification of middle path Landsat ETM+ imagery.

Statistical Method	Producer's		User's		% Overall
	% WB[a]	% NWB[a]	% WB	% NWB	
Logistic regression / S-Plus	89.6	92.6	85.6	94.8	91.6
CTA[b] / S-Plus	90.7	93.8	87.8	95.3	92.8
CTA / See5, boost = 0	90.2	95.2	90.8	94.9	93.5
CTA / See5, boost = 10	92.5	97.0	94.2	96.1	95.5
CTA / See5, boost = 99	93.6	97.0	94.1	96.6	95.8

[a] WB = whitebark pine, NWB = non-whitebark pine.
[b] CTA = classification tree analysis.

Classification tree analysis, with or without boosting, yielded consistently higher accuracies compared with logistic regression. See5 with maximum boosting yielded the highest estimated overall accuracies, improving overall accuracy by 2.3% with respect to See5 CTA with no boosting, and by 3% compared to the S-Plus single decision-tree results. The user's accuracy, referred to as errors of commission, compares the number of correctly classified points in a given class with the total number of points that were classified to that class. Maximum boosting increased the user's class accuracy for whitebark by 3.3% compared to See5 with no boosting, and produced an increase of 6.3% compared to the single decision-tree of S-Plus.

For further comparison, Landsat images and ancillary data from the east and west paths were classified using See5 CTA without boosting and with maximum boosting (99 trials). With boosting, the producer's estimated class accuracy for whitebark showed an improvement ranging from 0.6 to 6.0% and the overall accuracy showed an increase between 1.2 and 2.9%. Although, the user's class accuracy for whitebark actually decreased with boosting for the west path, it was increased by 2.3–2.9% for the middle and east paths, respectively (Table 2).

Table 2. Comparative accuracies for classification of Landsat ETM+ imagery with no boosting and boosting set to 99 trials.

		Producer's		User's		
	Image path	% WB[a]	% NWB[a]	% WB	% NWB	% Overall
NO BOOSTING	Middle	90.2	95.2	90.7	94.9	93.5
	East	94.0	91.6	86.2	96.5	92.5
	West	83.0	98.4	95.1	94.0	94.2
BOOSTING = 99	Middle	93.6	97.0	94.1	96.6	95.8
	East	94.6	95.9	92.7	97.0	95.4
	West	89.0	97.8	93.7	96.0	95.4

[a] WB = whitebark pine, NWB = non-whitebark pine.

Another important result of boosting was its superior performance in minimizing the training errors as shown for the middle path (Table 3). Maximum boosting reduced See5's overall classification error by 3%, correctly classifying 99.7% of the total training observations.

Table 3. Comparative percentage of training observations correctly fitted for classification of middle path Landsat ETM+ imagery.

Statistical Method	% WB[a]	% NWB[a]	% Overall
Logistic regression / S-Plus	88.8	91.9	90.9
CTA[b] / S-Plus	93.1	93.7	93.5
CTA / See5, boost = 0	95.3	97.5	96.7
CTA / See5, boost = 10	98.7	99.1	99.0
CTA / See5, boost = 99	99.6	99.7	99.7

[a] WB = whitebark pine, NWB = non-whitebark pine.
[b] CTA = classification tree analysis.

All five statistical methods used in this analysis indicated terrain parameters as significant predictors of whitebark pine. The final logistic regression equation for predicting whitebark pine was:

Logit (p) = -81.51013 + 0.00348(elevation) − 0.01919(aspect) + 0.00110(latitude) − 0.35120(tasseled cap brightness) + 0.01950(slope) + 0.33386 (TM band 7) − 0.19694(TM band 5) + 0.28711(TM band 4)

CART analysis via S-Plus created 23 decision-rules for classifying whitebark pine (Table 4) with a misclassification error rate of 6.5%. Predictor variables actually used in the decision-tree construction included all of the variables in the logistic regression equation in addition to NDVI and principal component 2.

Table 4. Classification Tree results from S-Plus CART analysis

```
1. Elevation  < 8932.5 ft
     1.1. Elevation < 8127.5 ft, THEN Non-WBP *(rule 1)
     1.2. Elevation > 8127.5 ft
      1.1.1. Latitude < 44.5425°
        1.1.1.1. Aspect < 89
          1.1.1.1.1. Latitude < 42.8455°, THEN WBP *(rule 2)
          1.1.1.1.2. Latitude > 42.8455°, THEN Non-WBP *(rule 3)
        1.1.1.2. Aspect > 89, THEN Non-WBP *(rule 4)
       1.1.2. Latitude > 44.5425°
        1.1.2.1. Elevation < 8606.5 ft, THEN Non-WBP *(rule 5)
        1.1.2.2. Elevation > 8606.5 ft
          1.1.2.1.1. TM band 5 < 99, THEN WBP *(rule 6)
          1.1.2.1.2. TM band 5 > 99, THEN Non-WBP *(rule 7)
2. Elevation > 8932.5 ft
     2.1. Aspect < 147.5
      2.1.1. TM band 5 < 90.5
        2.1.1.1. Elevation < 9316.5 ft
          2.1.1.1.1. Latitude < 44.5165°
            2.1.1.1.1.1. TM band 4 < 52.5, THEN Non-WBP *(rule 8)
            2.1.1.1.1.2. TM band 4 > 52.5, THEN WBP *(rule 9)
          2.1.1.1.2. Latitude > 44.5165°
            2.1.1.1.1. NDVI < 93.5, THEN Non-WBP *(rule 10)
            2.1.1.1.2. NDVI > 93.5, THEN WBP *(rule 11)
        2.1.1.2 Elevation > 9316.5 ft
          2.1.1.2.1. NDVI < 99.5
            2.1.1.2.1.1. NDVI < 87.5, THEN Non-WBP *(rule 12)
            2.1.1.2.1.2. NDVI > 87.5, THEN WBP *(rule 13)
          2.1.1.2.2. NDVI > 99.5
            2.1.1.2.2.1. PC Band 5 < 110.5, THEN WBP *(rule 14)
            2.1.1.2.2.2. PC Band 5 > 110.5, THEN WBP *(rule 15)
       2.1.2. TM band 5 > 90.5
         2.1.2.1 Latitude < 43.048°, THEN WBP *(rule 16)
         2.1.2.2. Latitude > 43.048°, THEN Non-WBP *(rule 17)
     2.2. Aspect > 147.5 1909
      2.2.1. TM band 4 < 45.5
        2.2.1.1. Elevation < 9634.5 ft
          2.2.1.1.1. Latitude < 45.025°, THEN Non-WBP *(rule 18)
          2.2.1.1.2. Latitude > 45.025°, THEN WBP *(rule 19)
        2.2.1.2. Elevation > 9634.5, THEN Non-WBP *(rule 20)
       2.2.2. TM band 4 > 45.5
         2.2.2.1. Principal Component 2 < 55.5
           2.2.2.1.1. Latitude < 42.6205°, THEN Non-WBP *(rule 21)
           2.2.2.1.2. Latitude > 42.6205°, THEN WBP *(rule 22)
         2.2.2.2. Principal Component 2 > 55.5, THEN Non-WB *(rule 23)
```

CTA using See5 with no boosting produced a very large tree with 105 decision rules (terminal nodes) and 15 levels of branching. All 20 predictor variables except for TM band 7 and tassled cap wetness were used in the See5 classification. Examining the upper branches (first 5 levels) of the tree reveals elevation, aspect, latitude, tasseled cap greenness, TM bands 1, 4 and 5, and principal component 5 as the significant predictor variables for whitebark pine. Although these are similar to the variables used in the S-Plus and logistic regression classifications, there

are some differences. Notably, slope which was part of the logistic regression equation, was not a factor in the S-Plus decision rules and did not appear in the See5 classification tree until branching level 6 and lower. Also, TM band 7, a significant predictor in the logistic regression results, was not called upon in either of the CTA procedures. The overall misclassification error rate using See5 with no boosting was 3.3%.

Although results from all 5 methods shared high accuracy rates, close visual inspection of the non-boosting classification images revealed substantial non-forested areas in the higher elevations (> 8700 feet) that were misclassified as whitebark pine. Boosting reduced this over-estimation. Since boosting focused on reducing classification errors in those areas that were inherently ambiguous (Freund and Schapire 1999), it was likely that the boosting algorithm more readily segregated extremely sparse high-elevation whitebark pine stands from non-forest better than the other algorithms. All of the methods produced high accuracies in spite of these inherent errors and, hence, might indicate a bias in the training data, specifically an insufficient number of non-forest and non-whitebark conifer training and validation points in the higher elevations.

For the final map of the entire study area the individual classification results, generated using See5 CTA with maximum boosting, of all three paths were merged. The final classified image (Fig. 2) yielded an overall accuracy of 95.7% and a user's class accuracy for whitebark pine of 92.9%. The KHAT statistic calculated at 0.90 for the resulting study area indicated that a given observed classification was 90% better than a classification resulting by chance (Lillesand and Kiefer 2000).

Discussion

When we began this project several remote sensing experts were skeptical of success. Previous attempts to create species specific coverages, especially in mixed conifer forests, have met with mixed results or resulted in poor classification accuracy (Redmond et al. 2001). We believe our success and the high degree of predictability from models resulted from: (1) taking time to verify spatial and vegetation classification accuracy for each input datum, and (2) concentrating on one class at a time and maximizing the accuracy using alternative methods.

Although the final test of any predictive vegetation model is field verification, we lacked funds and time to do this. We plan to compare the accuracy of our remotely sensed map to an existing map of whitebark pine distribution for part of the GYE (USFWS Grizzly bear Recovery Zone). This map was produced by photo interpretation of 1:1584 nominal scale aerial photographs and is the available coverage of whitebark that currently exists. A supplement to this map is currently under production (D. Despain, U.S. Geological Survey, personal communication) for the remainder of the GYE using timber stand inventories from surrounding agencies. We will compare our results to that supplement when available, and attempt to rectify discrepancies between the coverages to determine which coverage is in error. This information will then be useful in providing supplemental guidance on areas where additional training data may be required. Our model is premised on the assumption that the corrections we applied to each datum were correct and the assumption that Landenburger correctly classified whitebark pine for air photo interpretation. Although we are highly confident in the correctness of these assumptions, they must be tested.
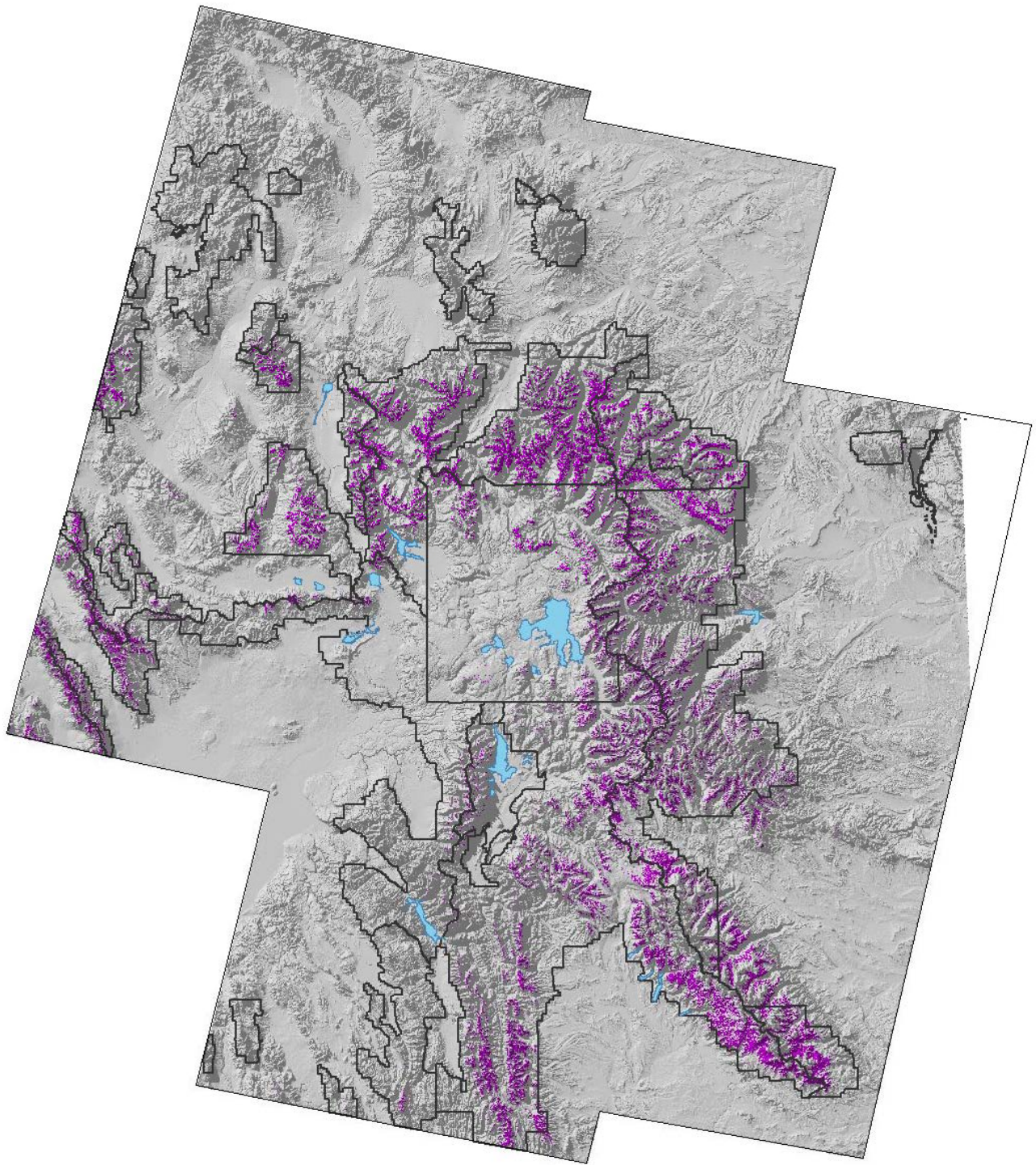
Fig. 2. Final classified image of whitebark pine distribution within the Greater Yellowstone
Ecosystem resulting from classification tree analysis with maximum boosting. (Map available in
color at www.nrmsc.usgs.gov/products/IGBST/2004report.pdf).

We consider this the first iteration of a longer program. The potential benefits of improving this map by conducting further research and field testing are many. Ongoing research projects that might be strengthened by the availability of these data include: (1) GYE Interagency Whitebark Pine Monitoring Program from which probabilistic samples will be derived from the whitebark pine map resulting from this study; (2) expansion of efforts to conduct a habitat-based grizzly bear Population Viability Analysis (USFWS 1993, Boyce et al. 2001), which is currently restricted to areas inside the Recovery Zone; (3) updates to data layers for the Yellowstone Grizzly Bear Cumulative Effects Model (Weaver et al. 1986, Dixon 1997); (4) modeling the potential effects of declines in major food sources or global climate change; (5) use in habitat selection models evaluating the effects of motorized recreation on denning and active grizzly bears; and (6) use in two studies examining GYE carnivore population dynamics that are sponsored by the USGS, National Park Service, and the Wildlife Conservation Society. Other efforts that would benefit include: (1) monitoring the distribution of white pine blister rust in the GYE, as part of key foods monitoring required by the recovery plan (USFWS 1993) and conservation strategy (USFWS 2003), (2) use by state wildlife and federal land agencies for planning and evaluation of management efforts, and (3) distribution through NBII, this data layer would be made available to the public.

Recommendations for Future Research

- Conduct independent ground validation of final classified map to verify the air photo based accuracy assessment. The accuracy assessments generated in this report, although statistically valid, are estimated accuracies based on the randomly selected 15% subset of the reference observations. Any biases inherent in the reference data will likely be represented in the validation sample. It is necessary, therefore, to conduct ground validation data independent of the original reference data.
- Target specific areas in the ecosystem to collect additional data: Newly collected ground validation data (previous bullet) might shed insight into where the model fails and might suggest where new data should be collected. For example, discrepancies (refer to results) in statistical models at higher elevations might be resolved by collecting additional non-whitebark field and photo-interpreted data above 8700 feet.
- Need to visit sites in the GYE where training observations are lacking, (i.e., Wyoming and Salt River ranges in the southern portion of the ecosystem).
- Incorporate ground validation data into the data classification and verification process to improve overall accuracy of whitebark pine map.
- Generate Federal Geographic Data Committee-compliant metadata for reference data and final map.
- Experiment with higher spatial resolution imagery such as ASTER (*Advanced Spaceborne Thermal Emission and Reflection Radiometer*), or Quickbird, and/or digital airborne imagery. The initial phase of the project, due to limitations in ground data and spectral resolution of Landsat imagery, was focused on identifying dominant to pure whitebark stands. In the next phase of this study, however, we would like to specifically target the mixed whitebark conifer stands since these are more crucial to grizzly bear survival (Mattson and Reinhart, 1997). The finer spatial and spectral resolution of ASTER might allow us to more readily resolve the spectral signals of these mixed conifer stands.

- Experiment with higher spectral resolution imagery, such as AVIRIS. Even without higher spatial resolution, it might be possible to "spectrally un-mix" mixed whitebark conifer stands to obtain an estimate of the percentage of whitebark present in these stands.

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Appendix A1:   Preliminary Evaluation of Training & Validation Data

Introduction

At the beginning of this study (see *Training/validation data development* section), we compiled field observation data from government agencies throughout the ecosystem. Based on preliminary analysis we found it necessary to verify the spatial location of these data. Consequently we invested several months of time and effort to verify and correct, when necessary and possible, spatial locations prior to model construction. Upon completing this we compared model results using uncorrected data with model results using the corrected data (supplemented with photo interpreted observations). Here we briefly summarize those results.

Sample Study Area

We chose the middle scene of our study (p38,r29) for this evaluation since (1) approximately 71.5% of the study scene falls within the Grizzly Bear Recovery Zone (delineated by the dark line), and (2) it offered a good representation of the reference data since it encompasses two National Parks and intersects five National Forests from which data was acquired.

Results

Our raw dataset, pre-modification and pre-verification, consisted of 4,569 field observations with whitebark observations comprising 18% of the total. The cleaned data set consisted of 5,743 reference observations with 2,603 of these points collected via air photo interpretation and whitebark comprising 35% of the total observations. Approximately 31% of the initial raw data had to be eliminated due to poor spatial accuracy and insufficient information to modify coordinate locations. Classification tree analysis (CTA) using See5 with 10 boosting trials was applied to the raw dataset and the modified dataset.

Results for the modified dataset indicate an overall accuracy increase of only 4% from the results of the raw data (Table 1), however, producer's class accuracy for whitebark increased by 23.5% and user's accuracy for whitebark improved by 15.4%. The KHAT statistic for the raw dataset was calculated as 0.683 versus 0.897 for the corrected data set, indicating an increased probability of 21.4% that an observed classification was better than one derived by chance.

Table 1. Classification results of raw versus modified training observations.

Classification Tree Analysis (See5 with 10 Boosting Trials)	Producer's		Users		
	% WB[a]	% NWB[a]	% WB	% NWB	% Overall
Raw dataset	68.0	96.3	79.8	93.3	91.3
Modified dataset	91.5	97.5	95.2	95.4	95.3

[a] WB = whitebark pine, NWB = non-whitebark pine.

APPENDIX B

The Feasibility of Detecting Trends in Whitebark Pine Cone Counts

Steve Cherry

Whitebark pine (*Pinus albicaulus*) seeds are an important nutritional resource for grizzly bears (*Ursus arctos*) in the Greater Yellowstone Ecosystem (GYE). Annual fluctuations in seed availability are associated with annual changes in bear mortality. Mortality in years with good seed production tends to be lower than mortality in years with poor seed production. Concerns have been raised about the future of this food resource due to possible adverse effects of whitebark pine blister rust, mountain pine beetle, and climate change.

Currently whitebark pine seed production is monitored by conducting cones on a number of transects located in the GYE. In 1999, for example, a total of 19 transects was run. Cones are counted on approximately 10 trees in each transect and the average number of cones per tree is computed.

The detection of trends via monitoring of ecological processes has received increasing attention in recent years. The results presented here are based on the work of VanLeeuwen et al. (1996) and Piepho and Ogutu (2002), principally the latter.

There were 19 transects, considered a representative sample of transects from some finite population, and a 14 year time period (1989-2002). The time period was chosen because all 19 of the transects have data collected for at least this long. There were a couple of missing values which were ignored. The response variable was the log of transect mean cone count. A small amount was added to each mean to avoid taking the log of 0. The model was

$$y_{ij} = (\mu + a_i) + w_j(\beta + t_i) + b_j + e_{ij}$$

where y_{ij} was the mean cone count on transect i in year j, μ and β were fixed intercept and slope parameters, w_j was a known constant indicating years, a_i was a random intercept effect for transect i, b_j was a random year effect, t_i was a random slope effect, and e_{ij} was noise (Piepho and Ogutu 2002). Each random effect was assumed to be normally distributed with mean 0 and variance components σ_a^2, σ_b^2, σ_t^2, and σ_e^2, respectively. The random intercept and slope components were assumed to be correlated with an unstructured covariance matrix. This was the only dependency structure allowed. The transect specific series of cone counts was given by $(\mu + a_i) + w_j(\beta + t_i)$. The mean series was $\mu + w_j\beta$ and the relevant variance components are σ_a^2 and σ_t^2. This model was a modification of one originally presented in VanLeeuwen et al. (1996). They assumed that a_i and t_i were independent which resulted in the intercept term being affected by the origin of the year effect, i.e., the otherwise arbitrary coding of year could affect the results. The assumed dependency structure between the random intercept and slope components led to the results being invariant to how year was coded (1 to 14 versus 1989 to 2002). Details can be found in Piepho and Ogutu (2002).

The above model says that each transect has its own trend line with its own intercept and slope. We can estimate the mean intercept and trend and the variability about these means. We can account for and estimate year to year variability and transect to transect variability.

The model was fit in SAS using PROC MIXED. Parameters were estimated using Restricted Maximum Likelihood (REML). No attempt was made to rigorously evaluate the model, although the requisite normality assumptions seemed acceptable. The goal was not to conduct a detailed analysis of temporal variation in cone counts but to get reasonable estimates of relevant parameters so that an evaluation of the feasibility of long-term monitoring of cone counts could be made.

Results

The parameters of most interest are the trend β, the interannual variability σ_b^2, and the residual variance σ_e^2. Estimates of these quantities and associated 95% confidence intervals are shown below (Table 1). The standard error of the estimate of the trend is 0.0916.

Table 1. Point estimates and approximate 95% confidence intervals for trend, interannual variance, and residual variance.

Parameter	Estimate	95% Confidence Interval
β	-0.034	(-0.234, 0.165)
σ_b^2	1.827	(0.321, 3.334)
σ_e^2	1.036	(0.864, 1.266)

The confidence interval for trend was computed using a t-distribution critical value with 12 degrees of freedom. The test of the null hypothesis $H_0 : \beta = 0$ (no trend) yielded a p-value of about 0.71. Thus, we estimate a negative trend in mean cone counts but there is a great deal of uncertainty associated with the estimate. In English, the data provide no guidance as to whether mean cone counts are increasing or decreasing over time.

The estimates of the variance components for year and residual are 1.83 (95% confidence interval of 0.32 to 3.33) and 1.04 (95% confidence interval of 0.87 to 1.27), respectively. There does appear to be fairly strong evidence that these quantities are significantly greater than 0.

Dividing the estimate of the trend $\hat{\beta} = -0.034$ by the residual variance of 1.04 yields a standardized trend component that can be interpreted as annual rate of change. Thus, $\hat{\beta}_S = -0.034 / 1.04 = 0.033$ leads to an estimate of a 3.3% annual decline in cone counts. Of course the wide standardized confidence interval (-0.225, 0.158) means that although we estimate an approximate 3% annual decline but these data are consistent with anywhere between an approximate 22% decline to a 16% increase.

The determination of power requires simulation but we can evaluate power indirectly. If the trend at all transects mirrors the regional trend, i.e., there is no random slope component then

$$\text{var}(\hat{\beta}) = \frac{\sigma_b^2 + \dfrac{\sigma_e^2}{n}}{\sum (j - \bar{j})^2}$$

where σ_b^2 is the interannual variability, σ_e^2 is the residual variance, n is the number of transects, and j is the year. The denominator is invariant to how year is coded, e.g., 1 to 14 versus 1989 to 2002. The model,

$$y_{ij} = \mu + w_j \beta + a_i + b_j + e_{ij}$$

with a fixed slope was refit to the data. The resulting estimates of all parameters were similar to those seen above (Table 2). We can use these results to indirectly investigate power.

Table 2. Point estimates and approximate 95% confidence intervals for trend, interannual variance, and residual variance.

Parameter	Estimate	95% Confidence Interval
β	-0.034	(-0.231, 0.163)
σ_b^2	1.801	(0.307, 3.295)
σ_e^2	1.232	(1.041, 1.482)

The standard error of the trend estimate is 0.0906.

Power can be evaluated indirectly by determining the margin of error associated with 95% confidence intervals for β_1. The margin of error is the half-width of the interval and is approximately given by

$$2\sqrt{\text{var}(\hat{\beta}_1)}.$$

This approach provides a lower bound on the trend that can be determined via a test of

$$H_0 : \beta_1 = 0$$
$$H_a : \beta_1 \neq 0$$

at $\alpha = 0.05$. For example, if the margin of error is 0.02 then an approximate 95% confidence interval for β_1 would have the form $\hat{\beta}_1 \pm 0.02$ and clearly β_1 would have to exceed 0.02 in magnitude before we would have a good chance of detecting a trend of that magnitude.

The table below shows the margins of error associated with approximate 95% confidence intervals for β_1. The number of sites was set equal to $n = 19$. Entries were computed using the

estimates for interannual variability ($\hat{\sigma}_b^2 = 1.8$) and residual variance ($\hat{\sigma}_e^2 = 1.2$) for time periods of different lengths.

Table 3. Margin of error associated with approximate 95% confidence intervals for trend with $n = 19$.

Time in years	Margin of error = $2\sqrt{\text{var}(\hat{\beta}_1)}$
5	0.863
10	0.301
25	0.076
50	0.027
75	0.015
100	0.009

The margin of error after 50 years is 0.027, which implies that at least 50 years of monitoring would be needed to have a reasonable chance of detecting a trend of that magnitude.

It is not realistic to consider the residual variance as being fixed as the time period increases but the effect of residual variance decreases as more data are collected over time so that the effect of violating that assumption is minimal. For example, even if there was no residual variance the margin of error at 25 years would be 0.074. The assumption has also been made of course that the interannual variability also remains constant. This may be more tenable. Even more tenable is the assumption that year to year variability in cone counts will be high relative to the residual variance. Cone counts are, after all, known to be highly variable from year to year. Thus, the behavior of the margin of error in the above calculations is driven primarily by the interannual variability and the length of the time.

Increasing the number of transects also does not help much. This is because the term σ_e^2 / n goes to 0 fairly quickly as the number of transects increases, decreasing the effect of the residual variance. Table 4 shows the margins of error associated with 95% confidence intervals for β_1 with $n = 200$. There is little if any difference in the margins of error seen in Table 3.

Urquhart and Kincaid (1999) pointed out that even small non-zero values for σ_b^2 lead to substantial reductions in power. They never considered values of σ_b^2 in excess of 0.30 in their simulations and this led to power to detect an annual trend of 2% of less than 0.20 over a 20 year time span.

Table 4. Margin of error associated with approximate 95% confidence intervals for trend with $n = 200$.

Time in years	Margin of error = $2\sqrt{\text{var}(\hat{\beta}_1)}$
5	0.850
10	0.296
25	0.075
50	0.026
75	0.014
100	0.009

Of course if one were willing to live with less confidence, then one could compute the margin of error for a 90% confidence interval (corresponding to an $\alpha = 0.10$ test of the two-sided hypothesis described above). The margin of error would be less in this instance, but not appreciably so. For example, the margin of error at 25 years would be 0.063 and at 50 years it would be 0.022.

Another caveat to keep in mind is to recall that the results in Tables 3 and 4 assumed constant slopes for all transects. If in fact this is not true, and it probably is not, then accounting for this would increase the margins of error in the above tables.

The general conclusion is that regardless of the number of transects, long monitoring periods will be required to provide statistical evidence of a decline in cone production. Further the large interannual variability means that there is little to be gained by considering other sampling plans (Urquhart and Kincaid 1999). By the time there was convincing statistical evidence of a problem, the problem would have become obvious nonstatistically.

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APPENDIX C

Interagency Whitebark Pine Health Monitoring Program
for the Greater Yellowstone Ecosystem

2004 Annual Report

Greater Yellowstone Whitebark Pine Monitoring Working Group

Current Working Group Participants[a]

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[a] This project represented a collaboration in the truest sense of the word, such that distinguishing order of participants with respect to relative contribution was virtually impossible. Consequently, order of participants is alphabetical.

Recommended citation for GYWPMWG (2005):

Greater Yellowstone Whitebark Pine Monitoring Working Group. 2005. Interagency Whitebark Pine Health Monitoring Program for the Greater Yellowstone Ecosystem, 2004 Annual Report. Pages 92-125 *in* C.C. Schwartz, M.A. Haroldson, and K. West, editors. Yellowstone grizzly bear investigations: annual report of the Interagency Grizzly Bear Study Team, 2004. U.S. Geological Survey, Bozeman, Montana, USA.

ABSTRACT

Whitebark pine (*Pinus albicaulis*) is a "keystone" species throughout the Greater Yellowstone Ecosystem (GYE), the cones of which serve as a major food source for grizzly bears (*Ursus arctos*) and other wildlife species. Whitebark pine stands have been decimated in areas of the northern Rocky Mountains due to the introduction of an exotic fungus—white pine blister rust (*Cronartium ribicola*)—as well as mountain pine beetles (*Dendroctonus ponderosae*). Resource managers from eight federal land management units have worked together as the Greater Yellowstone Whitebark Pine Subcommittee of the Greater Yellowstone Coordinating Committee (GYCC) to ensure the viability and function of whitebark pine in this ecosystem. In 2003 an additional working group was established to focus on an integrated monitoring effort throughout the GYE. The objectives of our monitoring were aimed at assessing the current status of white pine blister rust, whether or not blister rust is increasing within the GYE, and whether the resulting mortality of whitebark pine is sufficient to warrant consideration of management intervention (e.g., active restoration). Our study area is the entire GYE, comprised of 6 National Forests and 2 National Parks, although during 2004 our sample was restricted to within the Grizzly Bear Primary Conservation Area (PCA) because of limitations in the mapped distribution of whitebark pine (WbP) for the entire study area. We will extend beyond the PCA beginning in 2005. In 2004, transects were established and permanently marked for long-term trend monitoring in 45 stands of whitebark pine. In six of these stands, one additional transect was surveyed to assess within stand variation in blister rust. In total, 51 transects were sampled and 1,012 live trees surveyed. The number of whitebark pine trees sampled within these transects ranged from 1 to 141. Our data indicated that blister rust was relatively widespread throughout the PCA. Thirty six of the 51 (71%) transects had some indication of blister rust. Although blister rust was widespread, the infection severity was relatively low. We estimated that the proportion of trees infected with blister rust within the PCA to be 0.189 ± 0.05 SE, and most infected trees had ≤ 2 cankers. Further, most of the cankers also occurred on branches, which pose considerably less risk to the tree than cankers located on the trunks. Our data also indicated a high degree of observer variability in detecting indicators of blister rust and characteristics of trees used to indicate health. The concern over observer variability from our results extend well beyond the GYE effort and suggests a general need for better training and possible refinement of the methods used to assess blister rust.

INTRODUCTION

Whitebark pine occurs in the subalpine zone of the Pacific Northwest and northern Rocky Mountains of North America, where it is adapted to a harsh environment of poor soils, steep slopes, high winds, and extreme cold temperatures. Whitebark pine is a valuable species ecologically, and is considered a "keystone" species of the subalpine zone (Tomback et al. 2001). Whitebark pine's best known role in the Yellowstone ecosystem is as a food source for a variety of wildlife, most notably, the threatened grizzly bear (Mattson et al. 2001). In fact, in the GYE, annual whitebark pine cone production is one of the major predictors of annual survival and reproduction of the bears (Mattson et al. 1992, Mattson 2000). Whitebark pine seeds are high in fat and calories and are an important pre-denning food source for the Yellowstone grizzly bear. Grizzly bears gain access to large quantities of seeds stockpiled in red squirrel

(*Tamiasciurus hudsonicus*) middens (Mattson et al. 2001). As a pioneering species, whitebark often acts as a "nurse" plant for other trees, forbs, and shrubs that otherwise are not able to establish on their own in harsh environments (Tomback et al. 1993).

Background

In the early 1900s, white pine blister rust, a pathogen lethal to many 5-needled conifers, was introduced to the west coast of North America on imported, European nursery stock (McDonald and Hoff 2001). Since its arrival, it has decimated stands of whitebark pine in areas of the Cascades and northern Rocky Mountains (Kendall and Keane 2001). Although a whitebark pine tree infected with blister rust may survive for decades, its ability to produce cones, which grow in the upper canopy, is often compromised. When active, the site of infection, or canker, is a sweet attractant for rodents. The infected area is often consumed thus cutting off the flow of vital nutrients to the section of the tree above the gnawed portion. If this area is located on the trunk of the tree and it has been girdled, "top kill" will eventually occur and cone production will cease.

A more immediate threat to whitebark pine populations in the GYE is the mountain pine beetle. The mountain pine beetle is a native insect that has coevolved with pine forests in the western United States (Logan and Powell 2001). Variations in climate are largely responsible for the success of mountain pine beetle outbreaks. Mild summers and winters favor outbreaks, while cold winters and hot summers tend to decrease beetle activity and increase brood mortality (Kipfmueller et al. 2002). Evidence has shown that mountain pine beetles prefer to attack—and are more successful when attacking—trees that are already weakened by some other process, such as moisture stress, pathogens, or mistletoe (Kipfmueller et al. 2002). Some evidence indicates that older trees, weakened by other pathogens are more susceptible to mountain pine beetle infestations (Perkins and Roberts 2003, Tomback et al. 2001). It has also been suggested (e.g., Arno 1986) that fire suppression may lead to an increase of successional replacement of whitebark by more shade tolerant species such as Engelmann spruce (*Picea engelmannii*) or subalpine fir (*Abies bifolia*) (Keane 2001, Tomback et al. 2001).

Fire is an integral part of the ecology of whitebark pine communities. Yet larger, stand-replacing fires can kill mature, seed-producing whitebark pine trees, and may increase in frequency with a warmer and drier climate (Koteen 2002). Climate change may confound all of these threats and is hypothesized to affect whitebark pine communities through three mechanisms: 1) causing a shift in pathogens, which may lead to new regions of hospitable climate for white pine blister rust and increase the potential for pine beetle infestation; 2) increasing temperatures, which can lead to decreases in range availability for whitebark pine, due to exclusion by more heat-tolerant species, such as lodgepole pine (Mattson et al. 2001, Campbell and Antos 2003); and 3) changes in the frequency of severe fires, which lead to overall decreases in whitebark pine numbers (while they are adapted to small fires, large, stand-replacing fires may be detrimental to their overall distribution and abundance [Koteen 2002]).

Rationale for the Current Efforts

The "Final Conservation Strategy for the Grizzly Bear in the Yellowstone Ecosystem" (USFWS 2003) directs the National Park Service (NPS), U.S. Forest Service (USFS), and U.S. Geological

Survey (USGS) to monitor food sources of the grizzly bear, including ungulate carcasses, cutthroat trout, army cutworm moths, and whitebark pine. Specifically mentioned in the conservation strategy is monitoring of select transects throughout the GYE for cone production and white pine blister rust occurrence. Cone transect monitoring has been led by the Interagency Grizzly Bear Study Team and consists of cone counts and some blister rust monitoring (Haroldson et al. 2004). Blister rust is an important factor in the survival and reproduction of whitebark pine stands throughout the Northwest, and it has been determined that current cone counts within the GYE is not sufficient to understand the impacts of this introduced pathogen on whitebark pine stands and cone production (see Appendix B).

Given the Grizzly Bear Conservation Strategy directives, the ecological importance of WbP in the ecosystem, and that 98% of WbP occurs on public lands, the conservation of this species depends heavily on the collaboration of all public land management units in the GYE. Established in 1998, the Greater Yellowstone Whitebark Pine Subcommittee of the Greater Yellowstone Coordinating Committee is comprised of resource managers from eight federal land management units. This committee has been working together to ensure the viability and function of WbP throughout the region. As a result of this mutual conservation interest by these agencies, an additional working group (The Greater Yellowstone Whitebark Pine Monitoring Working Group) was formed in 2003-2004 with representatives from the U.S. Forest Service, National Park Service, U.S. Geological Survey, and Montana State University (MSU) for the purpose of integrating their interest, goals and resources into one unified monitoring program for the GYE. The group's intent is to estimate current status of whitebark pine relative to infection with white pine blister rust as well as to assess the vital rates that would enable us to determine the probability of whitebark pines persisting in the GYE. This project represents the initial results of that effort.

Objectives

General Questions Being Asked

Our specific monitoring objectives are intended to answer the following question(s): Is white pine blister rust increasing within the GYE, and is the resulting mortality of whitebark pine sufficient to warrant consideration of management intervention (e.g., active restoration)?

Specific Monitoring Objectives

OBJECTIVE 1 – To estimate the proportion of individual whitebark pine trees (>1.4 m high) infected with white pine blister rust, and to estimate the rate at which infection of trees is changing over time.

Justification/Rationale for this Objective: White pine blister rust has devastated whitebark pine in other parts of the northern Rocky Mountains (Kendall and Keane 2001, Koteen 2002), and anecdotal evidence suggests that infection rates may be escalating in the GYE (Koteen 2002, D. Tomback, personal communication). Given whitebark pine's importance in the upper subalpine ecosystem, and its being a key food source for a variety of wildlife, including grizzly bears, the

loss of seed-producing trees can affect not only a multitude of species, but also the persistence of this community type within the GYE.

OBJECTIVE 2 – Within infected transects, to determine the relative severity of infection (i.e., stage and magnitude of infection, and proportion of canopy kill) of white pine blister rust in whitebark pine trees >1.4 m high.

Justification/Rationale for this Objective: Determining the proportion of trees infected with white pine blister rust can be misleading without a further understanding of the magnitude of the infection. Given that within-tree spread of blister rust occurs primarily from new infections from the source, rather than spread from existing infections, trees that are infected at low levels may persist for considerable time in the absence of new infections (Koteen 2002). If the tree is infected near the crown, then the infection is most likely to cause cessation of cone production; it has been hypothesized that these types of infections occur more often than other types of infections in the GYE (Koteen 2002). The influence of the infection on tree mortality is highly dependent on the location of the infection, the age of the tree and other factors (such as mountain pine beetle infestations, root diseases, etc.); for instance, young trees that become infected almost always die relatively quickly, as do trees weakened by other causes (Koteen 2002).

OBJECTIVE 3 – To estimate survival of individual whitebark pine trees >1.4 m high, explicitly taking into account the effect of infection with and severity of white pine blister rust, and infestation by mountain pine beetle and dwarf mistletoe, and fire.

Justification/Rationale for this Objective: There has been some debate as to whether whitebark pine in the GYE is as vulnerable to the effects of white pine blister rust as it is in other regions (Carlson 1978, Arno 1986). Basidiospores of white pine blister rust are thought to be transported primarily during high moisture events (e.g., during periods of rain and fog [Hirt 1942, Van Arsdel 1956]), and the GYE is generally drier than other regions where white pine blister rust has been devastating to whitebark pine. Estimating survival will enable us to distinguish the occurrence (and severity) of white pine blister rust from the ecological effect of infestation (i.e., loss of mature whitebark pine); thus enabling us to determine the vulnerability of whitebark pine in the GYE directly, rather than relying on potentially controversial extrapolation from other regions.

Future Objectives under Consideration

At the present time our monitoring objectives focus on the status and trends of infection by white pine blister rust, to a lesser degree on occurrence of mountain pine beetle, and the resulting effect of these on survival of whitebark pine trees. Two additional topics that are currently being considered for future objectives are recruitment of whitebark pine trees into the reproductive population and the effects of forest succession on existing whitebark pine within suitable whitebark pine habitat types. Persistence of whitebark pine within the GYE depends on not only the survival of seed-producing trees, but also the recruitment of immature trees to the seed producing segment of the population. Monitoring changes in survival could result in misleading conclusions without some knowledge of the extent to which increased mortality is offset by recruitment. From a management perspective, there is also considerable concern about potential

replacement of whitebark pine by more shade-tolerant conifers (primarily subalpine fir). Understanding if, and to what extent, this is occurring could have important implications for potential restoration management of whitebark pine.

METHODS

Study Area

Our study area is the Greater Yellowstone Ecosystem and is comprised of 6 National Forests and 2 National Parks (Fig. 1). During 2004, our sample of WbP stands were restricted to within the Grizzly Bear PCA because of limitations in the mapped distribution of WbP for the entire study area. This region is approximately 2.4 million ha (5.9 million acres) (USFWS 1993) and includes approximately 50% of the known distribution of WbP within the GYE. An ongoing mapping effort will enable expansion of our study area beyond the PCA during 2005.
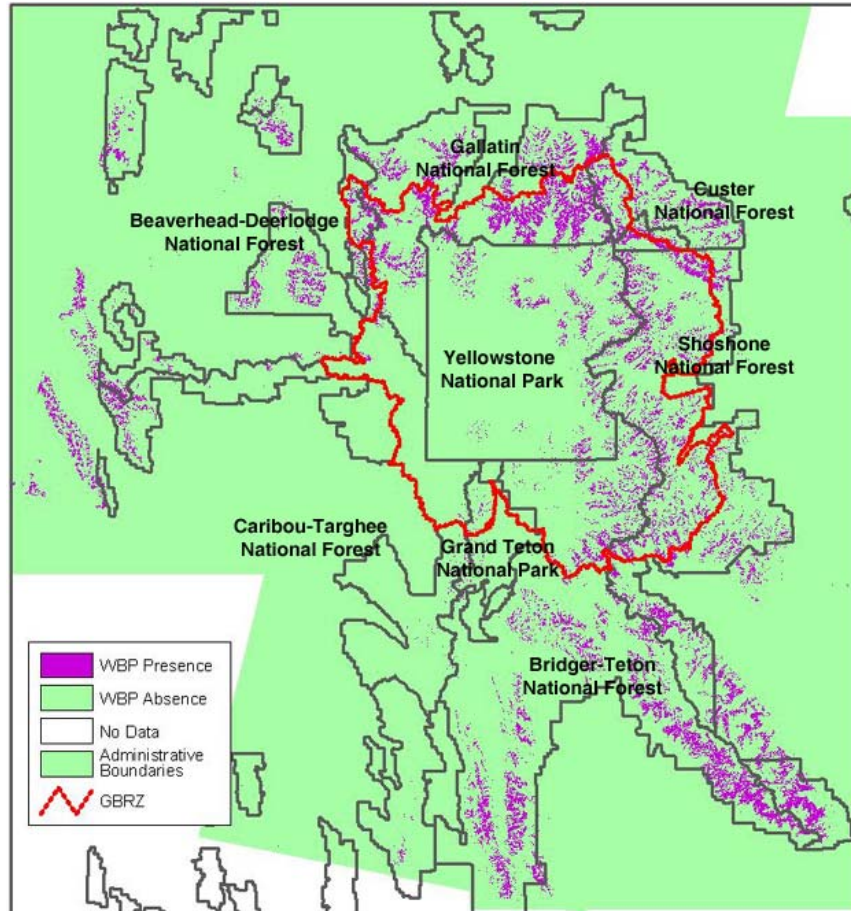
Fig. 1. Predicted occurrence of whitebark pine in the Greater Yellowstone Ecosystem. Sampling during 2004 was restricted to the Grizzly Bear Primary Conservation Area shown in red as the Grizzly Bear Recover Zone (GBRZ).

Sampling Design

Some Preliminary Assessments leading to our Design

A common problem encountered when designing studies is the balance between the number of samples (in this case transects) and the elements measured within each sample (in this case the number of trees within each transect). The Whitebark Pine Ecosystem Foundation (WPEF) Protocol (Tomback et al. 2004) initially approached this problem by using a variable transect length to ensure a minimum number of trees within each transect. We were concerned that this would produce biased estimates. To explore this concern and to evaluate the tradeoffs between the number of transects verses the number of tree per transect, we used two preliminary simulations which are described below.

Effect of Variable Transect Length for Obtaining Minimum Sample of Trees

The WPEF protocol (Tomback et al. 2004) initially suggested using variable length transects. The length of the transect is extended from 50 m if there are not at least 50 trees within the transect. Variable length transects can be a part of a valid sampling plan but varying the length of the transect to attain a minimum sample size will lead to biased estimators.

A simple simulation was used to show the effect of requiring a minimum sample size on the estimation of a proportion. We assumed a population with an infection rate of 0.20, very close to that observed in the GYE, and a random number of trees for each transect. A check was made for each transect, and if the number of trees was less than 50, an additional sample was taken to bring the minimum sample size up to 50. This was repeated 1000 times.

The result of this simulation was a mean of the empirical sampling distribution of 0.175, approximately 9% less than the actual rate of 0.200. Thus, the estimator (the sample proportion) was biased low. The statistical reason for this is may not seem intuitively obvious, but it centers on the fact that we no longer have an assumed binomial distribution; rather a new distribution which is a mixture of the binomial and another distribution with a lower mean. The practical consequences of this in the GYE whitebark pine monitoring program are not clear, but it raises concern that the earlier WPEF protocol may produce estimates that are biased low. The best way to avoid this concern is to use transects whose width and length are not determined by the number of trees in them. It should be noted that the WPEF protocol has since adapted their methods to a 10 x 50 m fixed transect length (Tomback et al. 2005).

Balancing the number of transects with the number of trees within each transect

Recognition of the potential bias resulting from variable length transects still does not resolve our concern about how to balance the number of transects with the number of trees within each transect. Thus a second simulation was used to explore these tradeoffs.

For this simulation, the mean number of trees within transects varied from 10 to 50 in increments of 10. The number of transects varied from 50 to 150 in increments of 50. Thus, there were 15 combinations of transect/tree numbers. The number of trees in each transect was determined by

drawing a random sample from a negative binomial distribution with the specified mean. The negative binomial was used because the number of trees was more variable than required for sampling from a Poisson distribution. Once the number of trees on a plot was determined, the number infected was determined by assuming each tree had a probability of 0.10 of being infected (based on infection rates observed during previous studies). The mean and standard error was computed for each trial and 1000 trials were run for each of the 15 transect/tree size combinations. The results indicated that the standard errors were fairly low in each case (Fig. 2).
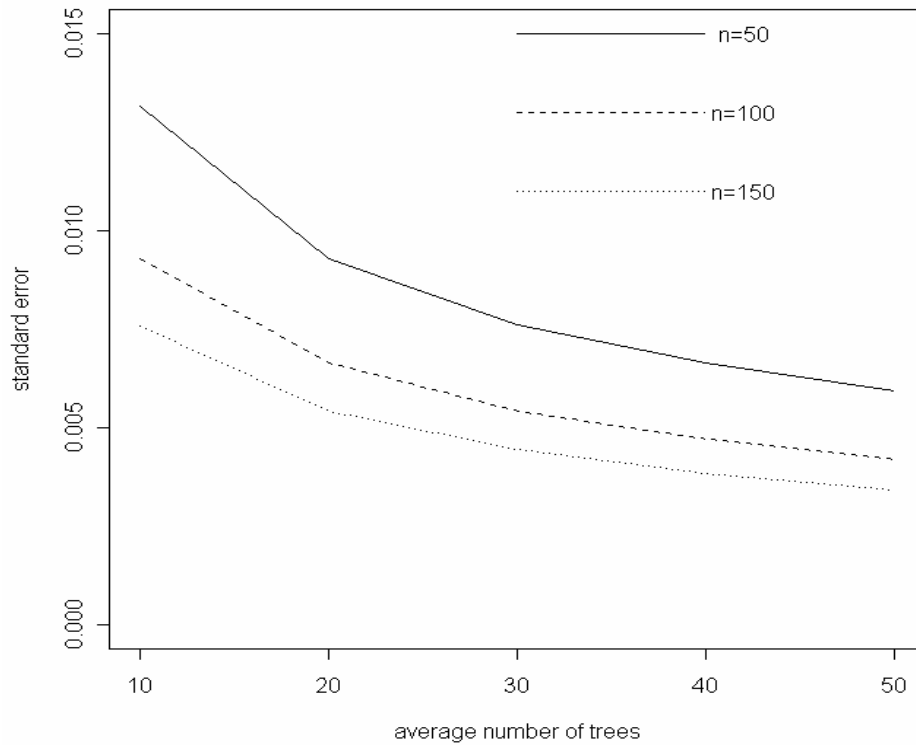
Fig. 2. The mean standard errors resulting from simulation increasing numbers of transects and increasing numbers of trees within each transect.

Another view of the tradeoffs between the number of transects and the number of trees within each transect is to examine the resulting confidence intervals (Fig. 3).
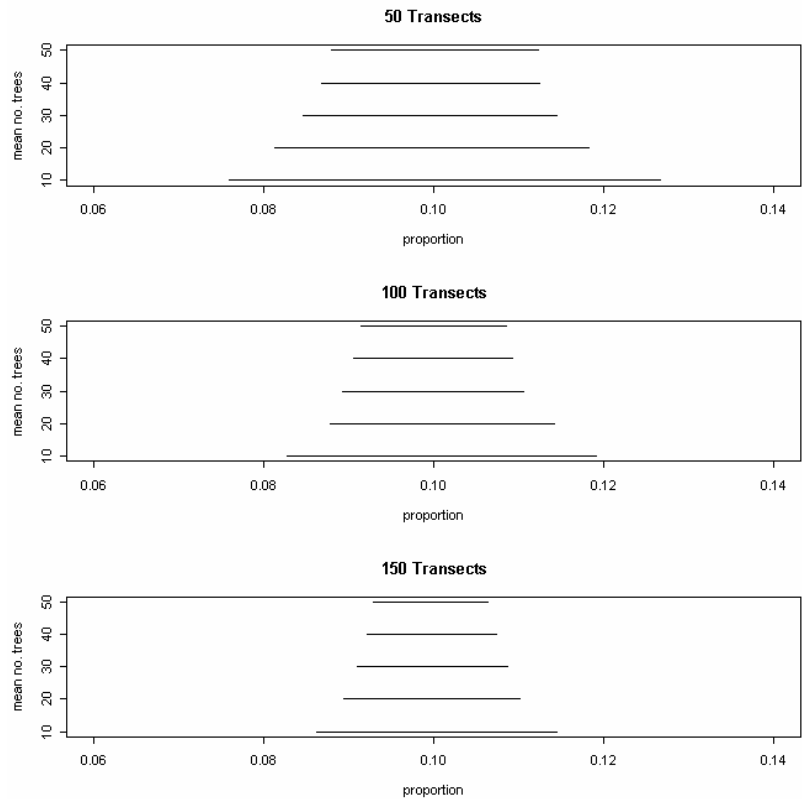
Fig. 3. The empirical 95% confidence intervals for the true mean resulting from simulations.

Obviously the more transects and the more trees within transects the better. But realistically, it appears that 100 transects with somewhere around 15 to 20 larger trees per transect on average, will be sufficient to produce reasonable estimates of status. It also appears that we gain relatively more efficiency by increasing the number of transects in our sample, as opposed to increasing the number of trees per transect. However, these results are based on an assumed simple random sampling plan with clusters as the primary sampling unit. This simulation does not take into account spatial considerations nor does it take into account stratification. We know that we will not be using this approach, relying instead on some kind of stratified sampling plan. However, the results do provide an initial indication of the precision we may expect.

Overall Design

The goal of the 2004 sampling effort was to characterize the current status of blister rust in the GYE. The sampling effort started early July and continued through late October.

The basic design was a 2-stage cluster design. Primary sampling units were stands of high-elevation whitebark pine dominated stands of approximately 2.5 hectares or larger. Secondary sampling units were 10 by 50 meter transects located within each stand. A simple random sample of primary units was selected followed by random selection of secondary units within each primary unit.

Target Population, Sampling Frame, and Sampling Units

The target population of ultimate interest is all whitebark pine trees in the GYE. It was clearly not possible to identify and map all whitebark pine trees. The sampling frame was actually defined in terms of mapped whitebark stands. The primary sampling unit was a whitebark pine polygon (stand) from the vegetation layer of the Cumulative Effects Model for grizzly bears derived from photo-interpretation (Dixon 1997). As indicated above, these were high-elevation whitebark pine dominated sites of approximately 2.5 ha or greater. A further restriction was that these stands were from areas that matched a forest layer derived from satellite imagery. Areas that were allegedly burned were omitted. The result was a sampling frame of 3,382 primary sampling units. These sampling units varied in size from approximately 2 ha to 594 ha. Each primary sampling unit was comprised of 10 by 50 m transects (secondary sampling units).

Selection of Primary and Secondary Sampling Units

We selected a simple random sample of 100 polygons from our sampled population of 3,382 such polygons. We wanted to ensure that the sample of polygons was adequately spatially distributed over the ecosystem and although we considered a stratified sampling approach we settled on the simple random sample because it did achieve the desired spatial distribution. The sampling frame is subject to inaccuracies due to mapping errors. There is the potential for the field crew to spend a great deal of time walking into an area only to find that a mapped polygon does not contain whitebark pines. Accordingly we also selected additional polygons near the initial selection. If the initial polygon is not suitable, then the crew is to choose the next nearest polygon. This seems a minor constraint on the randomization procedure that is justified by the limited time during which the crew has to collect data.

Secondary sampling units were 10 by 50 m transects located within each polygon. The number of secondary sampling units varied with polygon size. We selected a simple random sample of starting points for 6 secondary sampling units within each polygon. The first unit to visit was also chosen randomly. Details of how the transect is laid out and data are collected are described below.

Initially we planned on sampling one secondary unit per polygon. This is not an ideal sampling design because it will not be possible to estimate within polygon variability and it will not be possible to estimate standard errors of estimates of population totals and means without further assumptions. We have at least three options available for standard error estimation. First, we can take two 10 by 10 m subplots located at the end of each of the larger plots and use those to provide some information on within plot variability. This will not be ideal because the two plots will be paired, but it could still provide useful information for designing a long-term plan. Second, field crews have been instructed to attempt to read two transects per polygon whenever possible. Initial indications are that getting to a selected polygon and reading one transect takes the better part of a day but we anticipate that as the summer progresses the crew should become more proficient. Third, we can consider the resulting sample of transects to be a random sample of transects and get estimates of standard errors based on that assumption.

Field Methods

Many of our field methods were based on the protocol developed by the Whitebark Pine Ecosystem Foundation (Tomback et al. 2004). We deviated from these methods when appropriate to meet our specific objectives, or when we felt that an alternative was more biologically or statistically reliable.

Starting points for Transects

To identify potential monitoring transect locations throughout the study area, 100 primary stands of varying whitebark pine density were randomly selected from the primary units. Of these 100 stands, 45 were sampled by completing transects in as many stands as possible within a region within a 10-day sampling period and then shifting to a new region of the study area after each sampling period. Stands were grouped by geographic proximity into clusters of 3 stands, with the designations of one "Primary" stand and two associated "Alternate" stands per cluster. Within each stand, 5 random points were selected to serve as potential center points for each transect. A corresponding number between 0 and 359 was randomly chosen to define the vector for the transect. The random points were listed in rank order of selection, such that the first point in the list is the intended starting location. Should however, that location be unsuitable (i.e., misclassified as having whitebark pine when it does not), the next, closest point on the list became the starting point, and so on.

A handheld GPS was used to navigate to these coordinates with an error factor that varied from a few to several meters. If a site was suitable for sampling (i.e., had at least one whitebark pine tree), a 10 x 50 m transect was established. The start and end points of the transect were monumented by a 12" steel nail with a large washer and numbered tag driven in at ground level. Universal Transverse Mercator (UTMs) were recorded and photos were taken into the plot at both monuments. Inside the plot, 3 bearing trees were noted by species, diameter-at-breast-height (DBH) (1.4 m), and azimuth in regards to the monument. The overall habitat type (Steele et al. 1983) and forest cover type (Despain 1990) of the stand were recorded.

Inclusion of Individual Trees Within a Plot

All whitebark pine trees within the transect and >1.4 m high were marked and included as secondary sampling units. A given tree or cluster of trees (see below) was included within a plot if, and only if, the center of the trunk (or cluster of trunks) at ground level was within 5 m of the center of the transect line. This differs slightly from the criteria of the WPEF protocol, which includes trees if the center at breast height (rather than ground level) is within the plot boundary. The reason for this departure is that using a hypothetical plane 1.4 m above ground level is (1) more ambiguous than using ground level, (2) more difficult to determine than using ground level, and (3) results in a different sampling frame for trees (<1.4 m) and marked trees >1.4 m. For practical purposes the differences between these criteria should be negligible.

Tree Clusters

Tree clusters may form when multiple seeds are cached at the same location by Clark's Nutcrackers *(Nucifraga columbiana).* Although multiple branches of an individual tree are certainly possible, it is more often the case that multiple trees can sprout from the same location. Thus, to ignore that these are individual trees can be problematic and under sample the density of trees at a given location. Further, a given tree within a cluster may suffer damage and/or mortality from blister rust, while an adjacent trunk with less or no infection may remain undamaged and survive. However, we also recognize that because such trees sprouted from the same site, the canopy may be intertwined, etc., that these trees are not statistically independent. However, this may also be true for other trees within the same plot, depending on a variety of factors, including proximity. Thus, for the purposes of our sampling, we followed the guidelines of the WPEF protocol (Tomback et al. 2005) and record data for each individual tree within a cluster if the tree(s) stems are separate below ground (root level). In such cases we also identified it as being part of a cluster, allowing for analyses to take this into account when appropriate.

Dead trees

Dead whitebark pine trees >1.4 m height within the plot were not marked, but were recorded as being present. DBH (see below) was the only individual tree measurement taken. In contrast with the WPEF protocol (Tomback et al. 2004), no determination of cause of death was recorded due to the unreliability of retrospective assignment of cause of death. Evidence of insect or disease agents was noted in the comments.

Individual Tree Measurements

All live whitebark pine trees >1.4 m height within the transect were individually marked at 1.4 m breast height (BH) on the downside slope of the tree with a numbered metal tag (in federally-designated Wilderness Areas, tags were placed on the downside slope at the base of the tree). As per WPEF protocol, if stems are separate below BH (4.5 ft or 1.4 m), each bole stem is tagged and receives a letter designation to indicate clump membership. For live trees, we recorded DBH and crown ratio (i.e., the percentage of live canopy for the entire tree from top to bottom)(USFS 2002). The tree was surveyed for blister rust cankers and aecia (the active, fruiting body of the canker) in the upper, middle, and bottom thirds of the branches and on the upper, middle, and bottom third of the trunk (Fig. 4). Five auxiliary signs of blister rust infection: rodent chewing, branch flagging, swelling, roughened bark and oozing sap were also recorded. If three of the five auxiliary signs occurred in the same spot on a tree, that location was noted as having a canker. The number of branch and trunk cankers was recorded for each of these tree sections. A tally of the regenerating whitebark less than 1.4 m in height was taken and noted for the presence, absence, or uncertainty of blister rust.

Step 1. Divide tree into foliage and bole categories. Rate each category separately

Step 2. Divide foliage/bole into thirds

Step 3. Count the number of cankers observed in each third of each class (bole or foliage)

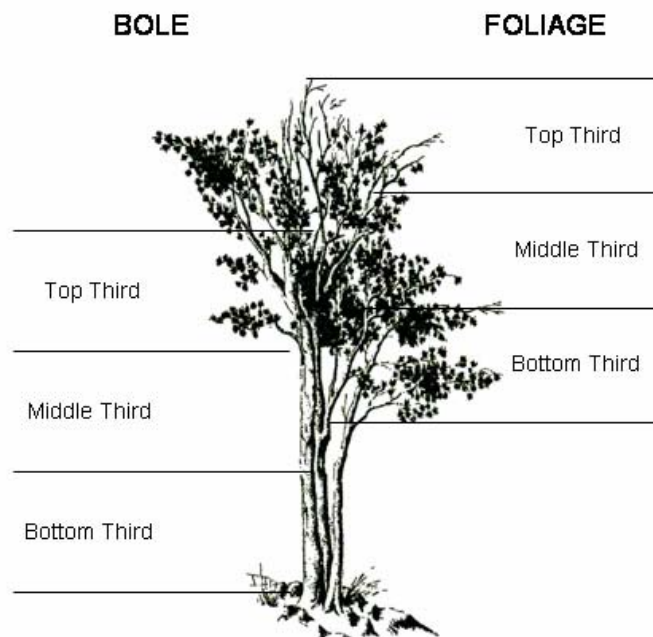Step 4. Estimate the percent of the foliage volume of each third that is live.

Fig. 4. Conceptual diagram of severity rating adapted from Six and Newcomb (in preparation).

In order to assess the extent of observer variability on this monitoring effort, 6 of the 45 transects were independently surveyed on the same day by three different observers. The variability amongst the observers was assessed out of the field (no discussion while at the plot between the crew). The focus of these additional six was on blister rust identification (cankers and aecia) and canopy percentage estimation.

Mountain pine beetle is also a major source of mortality for whitebark pine. Although this source of mortality is native to the GYE, there may also be an interaction between blister rust and pine beetle and there may be changes in the patterns and extent of beetle outbreaks resulting from climate change (Kipfmueller et al. 2002). Thus, as per the WPEF protocol, we recorded presence/absence of mountain pine beetle.

Statistical Analyses

Proportion of Infected Trees

Estimation of the proportion of infected trees requires estimation of the population total of infected trees and total trees. Most sampling texts present two alternative methods of doing this

for two-stage cluster designs. We assume below that we have a simple random sample of primary units followed by a simple random sample of secondary units within each primary unit.

Let M_i be the number of secondary units in the ith primary unit. Let m_i be the number of secondary units sampled from the ith primary unit. The total number of primary units will be denoted by N and the sample size of primary units by n. The total number of trees observed in the jth transect of the ith polygon is y_{ij}. Then

$$\hat{t}_i = \sum_{j=1}^{m_i} \frac{M_i y_{ij}}{m_i} = M_i \bar{y}_i$$

is the estimated total number of trees in the ith primary unit. This simply says that the estimated total number of trees in the ith polygon is the sample mean number per transect times the number of transects in that polygon. A similar equation holds for the total number of infected trees. Denoting the total number of trees observed in the jth transect of the ith polygon is x_{ij}. Then

$$\hat{u}_i = \sum_{j=1}^{m_i} \frac{M_i x_{ij}}{m_i} = M_i \bar{x}_i$$

is the estimated total number of infected trees in the ith primary unit. An unbiased estimator of the population total number of trees is

$$\hat{t}_{unb} = \frac{N}{n} \sum_{i=1}^{n} \hat{t}_i \ .$$

This just says that the estimated total number of trees in the population of polygons is equal to the average number per polygon times the total population size. Similarly an unbiased estimator of the population total number of infected trees is

$$\hat{u}_{unb} = \frac{N}{n} \sum_{i=1}^{n} \hat{u}_i \ .$$

These estimators tend to be highly variable.

A second estimator of the total number of trees is

$$\hat{t}_r = M_o \frac{\sum_{i=1}^{n} \hat{t}_i}{\sum_{i=1}^{n} M_i}$$

where M_o is the total number of transects. The corresponding estimator of the total number of infected trees is

$$\hat{u}_r = M_o \frac{\displaystyle\sum_{i=1}^{n} \hat{u}_i}{\displaystyle\sum_{i=1}^{n} M_i} .$$

This ratio estimator is biased but less variable and is generally preferred because the bias tends to be small. The number of polygons is $N = 3382$. The polygon sample size is $n = 3382$. There are approximately $M_o = 3717433$ transects in the population.

If we were simply interested in these totals then we would probably choose the ratio estimators because they tend to be less variable. However, we are interested in the proportion of infected trees in the population. A ratio estimator of this quantity can be obtained using either of the estimation procedures above,

$$\hat{p} = \frac{\hat{u}_{unb}}{\hat{t}_{unb}} = \frac{\hat{u}_r}{\hat{t}_r} = \frac{\displaystyle\sum_{i=1}^{n} \hat{u}_i}{\displaystyle\sum_{i=1}^{n} \hat{t}_i} .$$

The variance estimator is somewhat complicated. It has two components; one measuring the between polygon variability and one measuring the within polygon component. Let

$$d_{ij} = x_{ij} - \hat{p} y_{ij} \text{ and } \bar{d}_i = (1/m_i) \sum_{j=1}^{m_i} d_{ij} .$$

The estimator is

$$\hat{v}(\hat{p}) = \left(\frac{1}{\displaystyle\sum_{i=1}^{n} (\hat{t}_i / n)} \right)^2 \left[\left(1 - \frac{n}{N} \right) \frac{s_r^2}{n} + \left(\frac{1}{nN} \right) \sum_{i=1}^{n} M_i^2 \left(1 - \frac{m_i}{M_i} \right) \frac{s_i^2}{m_i} \right],$$

where

$$s_r^2 = \frac{\displaystyle\sum_{i=1}^{n} \left(M_i^2 (\bar{x}_i - \hat{p} \bar{y}_i)^2 \right)}{n - 1}$$

and

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$$s_i^2 = \frac{\sum_{j=1}^{m_i}\left(d_{ij} - \bar{d}_i\right)^2}{m_i - 1}.$$

The term in the square brackets containing s_r^2 is the between polygon component and the term containing s_i^2 is the within polygon component. If there is only one sampling unit in the ith polygon (i.e. $m_i = 1$), then the within polygon component for that polygon cannot be determined and the total variance will be underestimated. Note however that the within polygon component will tend to be smaller than the between polygon component because of the $\frac{1}{nN}$ term. The standard error of $\hat{p}$ is of course the square root of the estimated variance.

RESULTS AND DISCUSSION

A total of 45 polygons (stands) were visited with 51 transects being sampled. A total of 1,012 live trees were examined. The number of live trees per transect ranged from 1 to 141 with an average of 19.84 trees per transect.

Proportion of Trees Infected

The estimators described above produced the following estimated numbers of trees and number of infected trees in the population (Table 1). Note that these are estimates of the overall population totals, not the totals from our samples.

Table 1. Intermediate results of the estimated population totals of the number of trees and number of infected trees using unbiased and ratio estimators.

Parameter	Parameter description	Estimator type	Estimate
$\hat{t}_{unb}$	Total number of trees in population	Unbiased	73,910,491
$\hat{u}_{unb}$	Total number of infected trees in population	Unbiased	14,031,505
$\hat{t}_r$	Total number of trees in population	Ratio	65,673,532
$\hat{u}_r$	Total number of infected trees in population	Ratio	12,467,763

From these intermediate results, the estimated proportion of infected trees in the population is:

$$\hat{p} = \frac{14031505}{73910491} = \frac{12467763}{65673532} = 0.189.$$

Ignoring the between polygon component we get an estimate of the standard error of 0.0616. This is the minimum the standard error can be. Adding within polygon variance will only increase this quantity. This standard error is much larger than the standard error computed assuming a simple random sample of transects, and larger than that seen in our simulations.

We can get an idea of the amount of within polygon variance relative to between polygon variance by considering the 6 polygons in which two transects were sampled. Ignoring the within polygon variance the standard error of the proportion of infected trees on these 6 transects is 0.0499. The standard error computed by considering the within polygon variance is 0.0502. This suggests that within polygon variance may be negligible.

There are some encouraging and discouraging parts of this analysis. The encouraging part is the minor role played by the within polygon variability. This is a preliminary assessment based on a sample of size 6 so more data are needed to confirm this. The discouraging part is how large the overall standard error is just considering between polygon variability. This suggests that a lot of polygons will have to be visited to have a reasonable chance to detect meaningful changes in infection rates in a reasonable length of time.

Of the 51 transects, 36 were infected with white pine blister rust and the proportion of infected trees on a given transect ranged from 0 to 1.0 (Fig. 5).
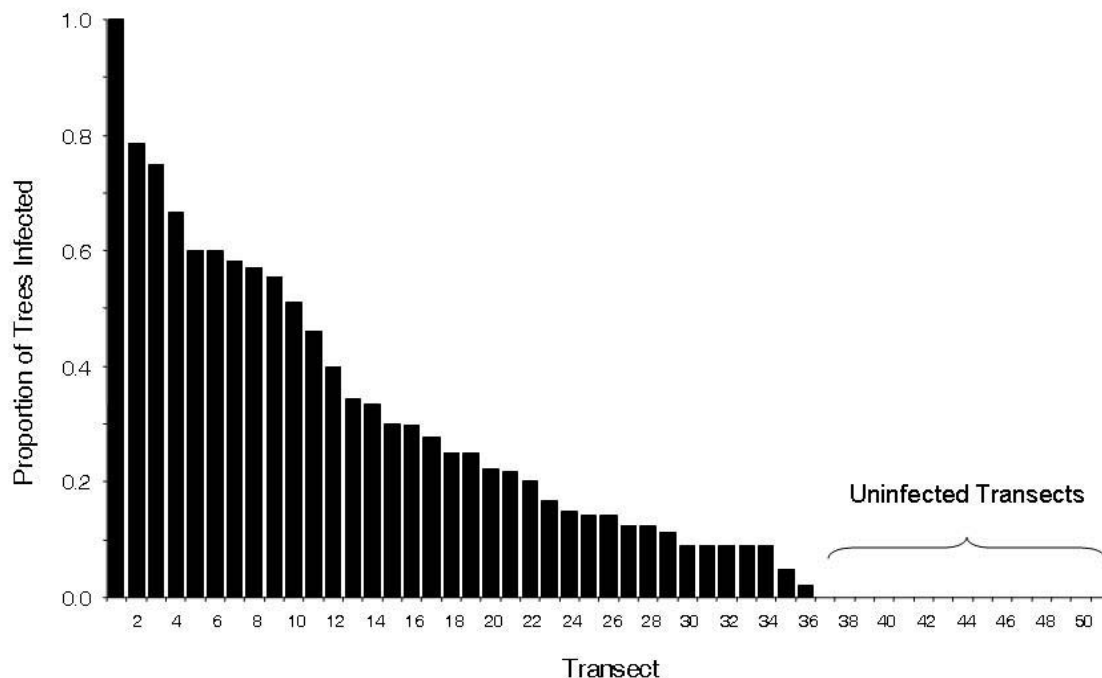
Fig. 5. The proportion of whitebark pine trees infected on each of the 51 transects sampled during 2004 arranged in rank order from most infected to least infected.

Although a formal spatial analysis has not yet been conducted, our preliminary data indicate that infection rates were highest in the northwest portion of our study area (Fig. 6). It should be noted, however, that our 2004 sampling effort was restricted to the Grizzly Bear Recovery Zone, and that the spatial distribution of infection may change substantially as we expand our effort beyond this zone starting in 2005.
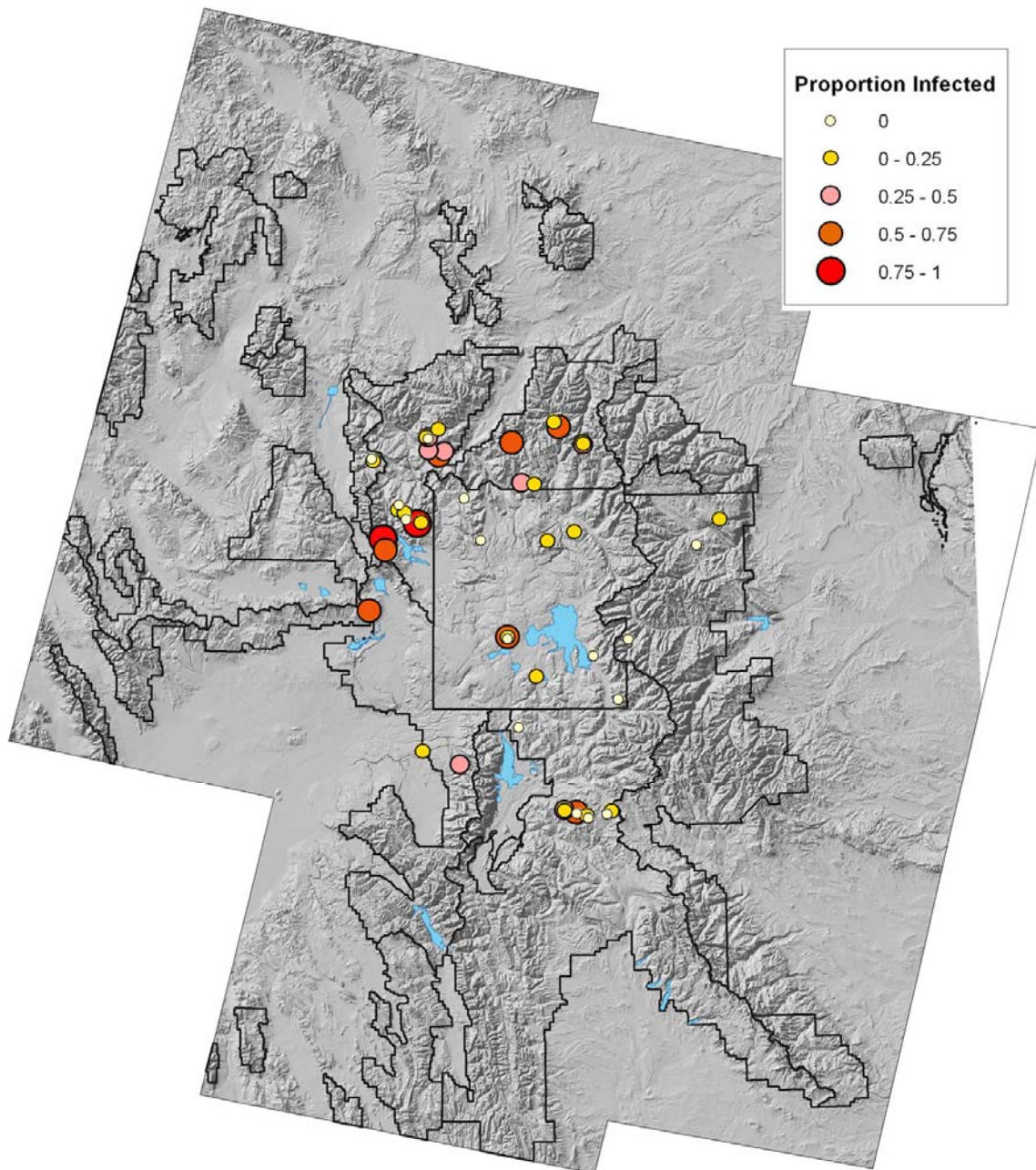
Fig. 6. Distribution of transects sampled during 2004, and class of proportion of trees infected.

Severity of Infection

Of 230 whitebark pine trees reported as infected, 128 (56%) were reported as having aecia. This is considered the gold standard for determining the presence of whitebark infection (Tomback et al. 2005).   If we limit our analysis to those 128 trees where identification of blister rust cankers is virtually certain, the proportion of trees having branch cankers is substantially greater than those with cankers on the trunk (Fig. 7).
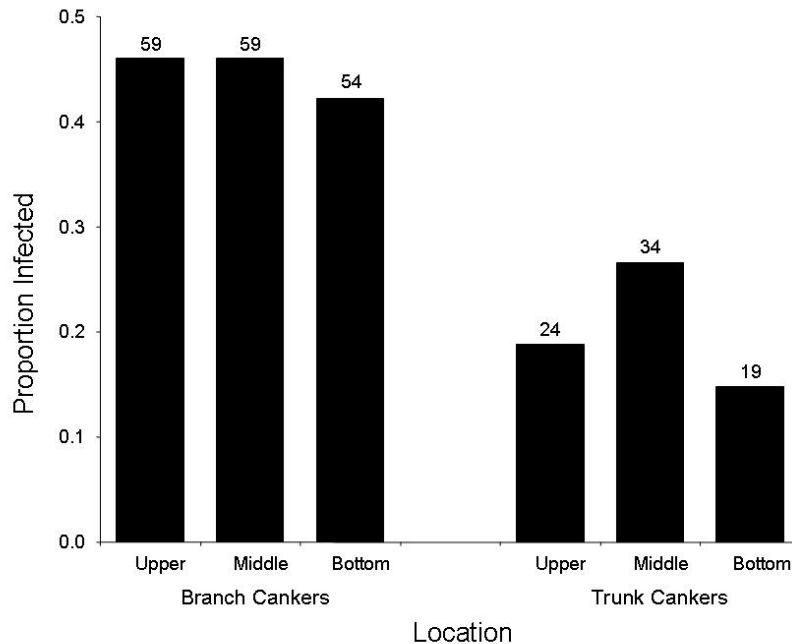
Fig. 7. The proportion of 128 trees infected with branch or trunk cankers, as determined by the presence of aecia, in each of three height classes.

This result is somewhat encouraging since cankers on the trunk are generally considered lethal to the tree, or may reduce the potential for cone production. In contrast, branch cankers, unless close to the trunk, may kill individual branches, but pose substantially less threat (Koteen 2002).

A similar conclusion is evidenced by the number of cankers within infected trees. This assessment was based on 230 trees that exhibited either aecia, and/or the presence of at least three of the other indicators; this was perhaps less conservative than the analysis above that was restricted to those trees with aecia present, but more conservative than if we had considered cankers based on the presence of any of the indicators.

This result indicated that most trees infected with blister rust had few cankers, particularly for trunk cankers. The number of branch cankers on infected trees ranged from 0 (i.e., when a tree had only trunk cankers) to 32 with approximately 90% of the trees having ≤5 branch cankers

(Fig. 8). The number of trunk cankers on infected trees ranged from 0 (i.e., when a tree had only branch cankers) to 4 with approximately 90% of the trees having ≤1 trunk canker.
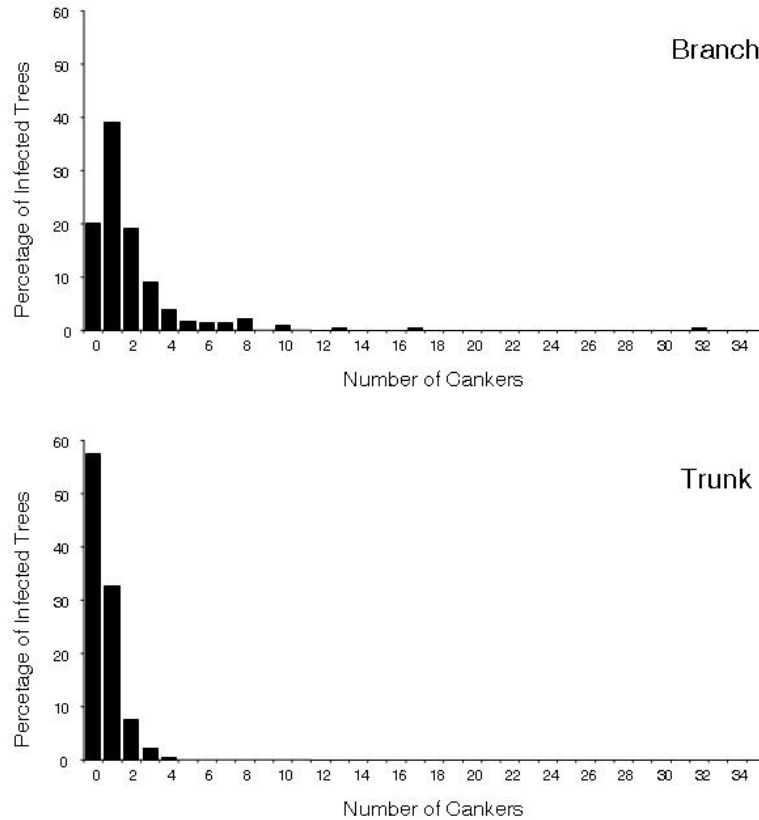
Fig. 8. The number of branch and trunk cankers on infected trees.

Mountain Pine Beetle and Mistletoe

The total number of live and dead trees for the 51 transects was 1,360. Of the live trees, 8 of 1,012 (<1%) had evidence of mountain pine beetle infestation, while 94 of 348 (27%) dead trees had evidence (galleries) of successful mountain pine beetle attack.

Dwarf mistletoe was found on one tree.

Indicators of Blister Rust Infection

A total of 128 trees were reported as having aecia, although information was also recorded on 5 other indicators of infection: rodent chewing, flagging, swelled bark, roughened bark, and oozing sap. The proportion of 128 trees with one or more cankers in each of the height and position classes along with an analogous assessment from 230 trees if a different standard is used (i.e., aecia and/or at least 3 other indicators) is given below (Table 2).

Table 2. The proportion of trees with one or more cankers in each of the height and position classes based on two criteria for identifying cankers. One assessment is based on 128 tress where aecia were present. The second assessment was based on 230 trees where aecia and/or at least 3 other indicators were present.

Height Class	Position in tree	Based on cankers with aecia present	Based on cankers with aecia and/or at least 3 other indicators
Upper	Branch	0.461 (59/128)	0.457 (105/230)
Middle	Branch	0.461 (59/128)	0.387 (89/230)
Lower	Branch	0.422 (54/128)	0.296 (68/230)
Upper	Trunk	0.188 (24/128)	0.217 (50/230)
Middle	Trunk	0.266 (34/128)	0.209 (48/230)
Lower	Trunk	0.148 (19/128)	0.104 (24/230)

These results initially appear to indicate that the more restrictive criteria of using aecia only to confirm the presence of infection did not produce substantially more conservative estimates of the proportion of trees infected in each height and position class as might have been expected. In fact only one height/position group (Upper/Trunk) had higher estimates from the more inclusive criteria. However, there are some confounding influences on this result. We included the additional indicators after the field season had started. Consequently, some transects included in the analysis based on aecia only were not included in the other analysis. Because the indicators were added after the season started, the observers also had gained additional experience in identifying cankers by the time these other indicators were included. These results should be interpreted with caution. We have no evidence that the criteria we established for using three of five indicators, in the absence of aecia, to confirm an infection does not produce substantially higher estimates.

Part of our initial effort is to assess the relationship among the various indicators and methods. By doing so, we have a better understanding of what these indicators and methods are telling us about blister rust infections. One aspect of this assessment is to understand the association, if any, of infections on different parts of the tree (i.e., height and position). The correlation matrix below shows the pairwise correlations between the numbers of cankers at the 6 locations on the trees (Table 3).

These results indicate a generally low correlation among infection on different parts of the tree. This implies that knowing something about the infection level in one part of the tree would tell us very little about infections in other parts. This reaffirms our existing protocol for recording

cankers in different parts of each tree. Where cankers occur on a given tree can have a dramatic effect on survival and reproduction for that tree, and trying to simplify our methods to ignore the height and position classes where cankers occur could result in a substantial loss of information.

Table 3. Correlation matrix showing correlations between locations of white pine blister rust cankers on trees. Data was presence/absence of cankers.

Location of cankers	Upper branch	Middle branch	Bottom branch	Upper trunk	Middle trunk	Bottom trunk
Upper branch	1	0.26	0.19	0.05	0.04	-0.05
Middle branch		1	0.74	0.19	0.40	0.02
Bottom branch			1	0.18	0.38	0.07
Upper trunk				1	0.20	0.14
Middle trunk					1	0.11
Bottom trunk						1

To further evaluate the relationship among different indicators, we also looked at what proportion of trees had indicators in addition to aecia (Fig. 9) and of those, whether they had 1, 2, 3, etc. of these other indicators (Fig. 10). For this assessment, we used 82 of the 128 trees with aecia (i.e., with confirmed blister rust), for which the other 5 indicators were also recorded.

These results indicate that the proportion of trees exhibiting at least one of the other indicators ranged from approximately 51-83%. Rodent chewing was the least frequent indicator and flagging was the most frequent. About 85% of the 82 trees with aecia (i.e., definitely infected with blister rust) had 3 or more of the indicators. We next examined the co-occurrence of individual indicators. Table 4 shows the proportion of trees with an indicator present that has another indicator. For example, 42 trees had evidence of rodent chewing and 33 (0.786) of those also had flagging.
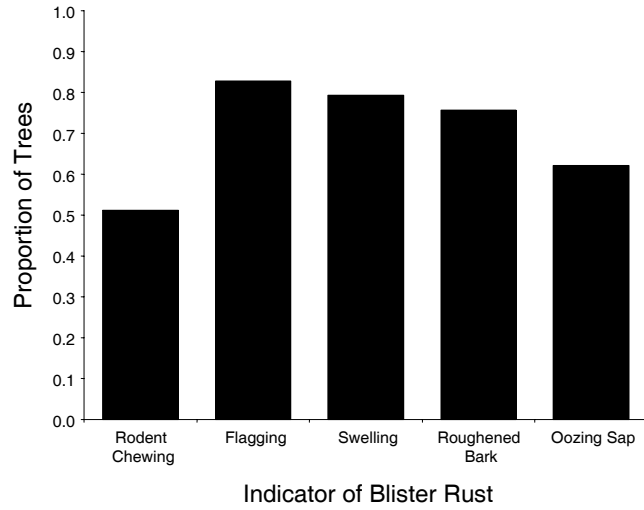
Fig. 9. Proportion of 82 trees with one or more incidences of each indicator of white pine blister rust.
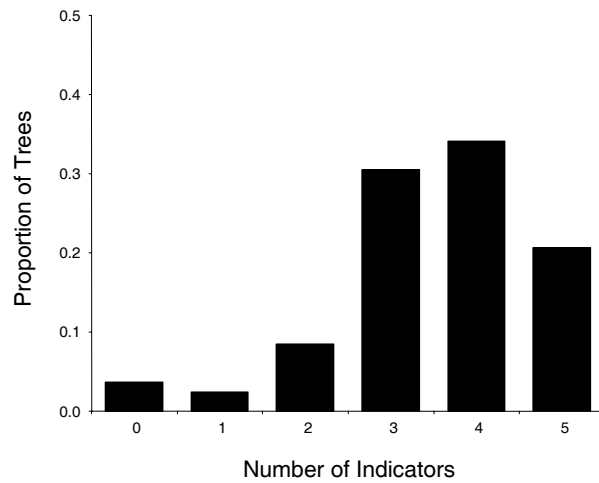
Fig. 10. The proportion of 82 trees exhibiting different numbers of indicators.

Table 4 provides information on the conditional relationships among the variables. This may provide an incomplete picture of the associations among the variables however. There are a number of ways to measure the similarity (dissimilarity) among such variables.

Table 4. The co-occurrence (proportion of trees [number of trees]) of indicators with each of the other indicators.

Indicator	Chewing	Flagging	Swelling	Roughness	Oozing sap
Chewing (42)	–	0.786 (33)	0.714 (30)	0.619 (26)	0.714 (30)
Flagging (68)	0.485 (33)	–	0.838 (57)	0.794 (54)	0.561 (46)
Swelling (65)	0.462 (30)	0.876 (57)	–	0.892 (58)	0.661 (43)
Roughness (62)	0.419 (26)	0.871 (54)	0.935 (58)	–	0.694 (43)
Oozing sap (51)	0.588 (30)	0.902 (46)	0.843 (43)	0.843 (43)	–

Table 5 is an extension of the previous analysis which contains chi-square test statistics and associated p-values of tests of independence of the occurrence of pairs of these indicators. The data were transformed from count data (number of incidences of rodent chewing for each tree) to presence/absence data (rodent chewing present or not). The 10 pairwise 2 by 2 contingency tables of matches and mismatches were constructed followed by the tests of independence. A low p-value indicates an association among the pairs. For example, the combination of swelling and roughened bark had a p-value of <0.001, indicating a strong association, in this case positive, among these two indicators (i.e., the occurrence of one indicated that the other was also likely to be present).

Table 5. Chi-square statistics and associated p-values (in parenthesis) for pairwise tests of independence among indicators of blister rust. The matrix is symmetric so only the upper diagonal is shown.

Indicator	Chewing	Flagging	Swelling	Roughness	Oozing sap
Rodent chewing	–	1.15 (0.28)	3.22 (0.073)	8.77 (0.0031)	3.12 (0.077)
Flagging		–	5.03 (0.025)	3.12 (0.077)	5.03 (0.025)
Swelling			–	31.54 (0.000)	2.09 (0.14)
Roughened bark				–	5.54 (0.018)
Oozing sap					–

These results need to be interpreted with caution due to the fact that we have evidence of considerable observer variability in the assessment of these indicators (see below). If we had data from another observer the results could conceivably change, perhaps quite dramatically. We have refined how we will record this information during our 2005 field season and should be able to provide a better assessment of the relationship among indicators and the implications of those relationships following our 2005 field season.

Observer Variability

Six transects were read independently by 3 different observers to evaluate the extent of observer variability. We plan to continue this evaluation program during the 2005 field season. Data were collected on: (1) crown ratio (%), (2) number cankers at each height/position class, (3) the presence/absence of aecia, rodent chewing, swelling, roughened bark, oozing sap, and flagging, (4) the percent of live canopy volume at each height/position class, (5) middle, and lower thirds of a tree, and the presence/absence of mistletoe and mountain pine beetle. Results for each variable are summarized below.

Crown Ratio

The crown ratio was visually estimated for 59 trees by the 3 observers. Pairwise correlations between observers was low (Table 6). A pairwise scatter plot of the estimates is also shown (Fig. 11).

Table 6. Pairwise correlations between crown ratio estimates (%) for 3 possible observer pairs.

Observer pairs	Correlation
Observer 1 and 2	0.675
Observer 1 and 3	0.533
Observer 2 and 3	0.766

These correlations are not as high as we would have liked. It is interesting to note that even though this is presumably a continuous variable, out of 177 possible ratings 176 ended in either a 0 or a 5.
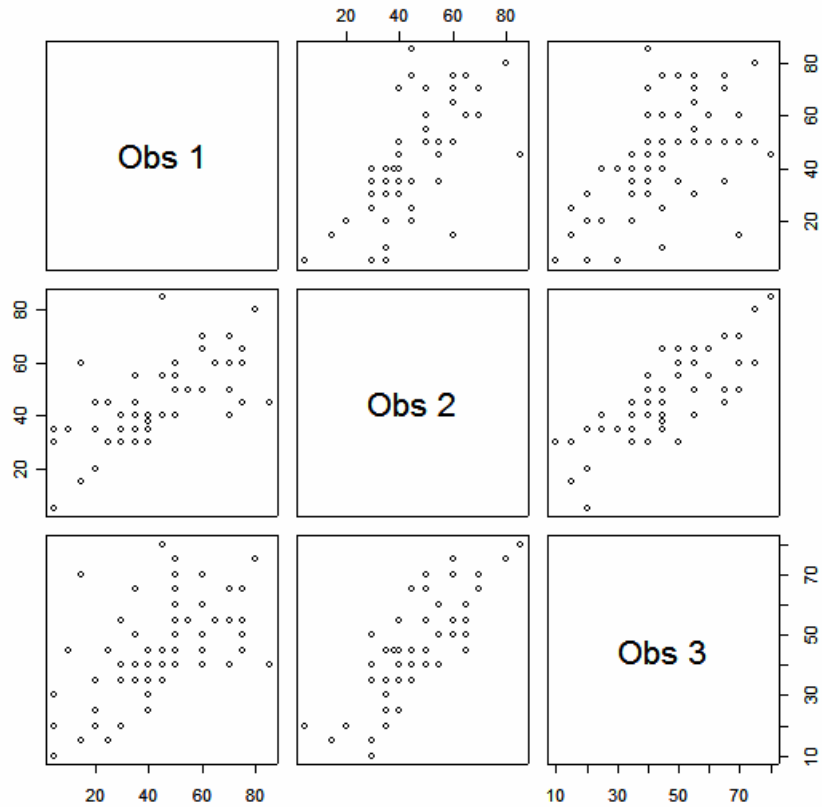
Fig. 11. Pairwise scatter plots of crown ratio estimates (%) for observer pairs.

Proportion of Trees Infected with Blister Rust

The proportion of infected trees was determined using canker counts and aecia presence/absence from each of the 3 observers (Table 7).

The final estimate, based on 60 trees, is identical for 2 of the observers. However, it is obvious that they did not identify the same 9 trees. The data suggest that there was no or little bias in how cankers were counted, i.e., one observer was not always counting more or less than another. However, even though the totals tend to be close to one another, estimates of variability are affected by the variability among transects. Further, these errors will tend to complicate any analysis requiring information from individual trees, e.g., building models examining the relationship of environmental covariates and infection status.

Aecia were noted on 9 of the 60 trees in the 6 transects but in only 2 of those cases did all 3 observers note aecia on the same tree.

Table 7. Proportion of infected trees as determined by canker counts for 3 different observers on 6 transects. Bold faced transects indicate transects where differences were recorded.

Transect ID	Observer 1	Observer 2	Observer 3
4280	0/5 (0)	0/5 (0)	0/5 (0)
2602	1/4 (0.25)	1/4 (0.25)	1/4 (0.25)
4119	**2/11 (0.182)**	**1/11 (0.091)**	**4/11 (0.364)**
1830	1/7 (0.143)	1/7 (0.143)	1/7 (0.143)
531A	**1/13 (0.077)**	**2/13 (0.154)**	**0/13 (0)**
4299A	**2/20 (0.10)**	**3/20 (0.15)**	**3/20 (0.15)**
Total	7/ 60 (0.117)	9/60 (0.15)	9/60 (0.15)

Canker Counts

All 3 observers recorded no cankers on trunks of the 63 trees. There was some variability among branch counts (Table 8).

Table 8. Counts of cankers in branches of 63 trees recorded by 3 different observers.

Observer	Upper branch cankers	Middle branch cankers	Bottom branch cankers	Total canker count
1	3	3	3	9
2	6	5	3	14
3	5	5	2	10

It was rare for observers to identify cankers on the same tree. Upper branch cankers were recorded on 8 different trees but in only 3 cases did 2 or more observers record upper branch cankers on the same tree. Middle branch cankers were recorded on 7 different trees with 5 cases of 2 or more observers recording cankers on the same tree. Lower branch cankers were recorded on 4 trees but in only 1 case did more than 1 observer note cankers on the same tree. There were 4 cases where all the 3 observers noted cankers on upper, middle, or lower branches of the same tree. In all other cases where cankers were recorded at least one observer did not see any.

Live Canopy Volume

Correlations between estimated live canopy volume (%) for 3 different areas of a tree are shown below (Table 9). Pairwise scatter plots are shown in Figures 12, 13, and 14.

Table 9. Pairwise correlations between estimated live canopy volume (%) for upper, middle, and lower sections of the canopy.

Observer pairs	Upper live canopy volume	Middle live canopy volume	Bottom live canopy volume
1 and 2	0.587	0.404	0.368
1 and 3	0.771	0.512	0.028
2 and 3	0.747	0.459	0.292

With the exception of observers 1 and 3 on upper live canopy volume, these correlations are quite low, indicating a general lack of consistency among observers. This result raises serious concern regarding the validity of this measurement. Either additional training is required to ensure better consistency among observers or this measure may need to be discarded in the future for lack of reliability.
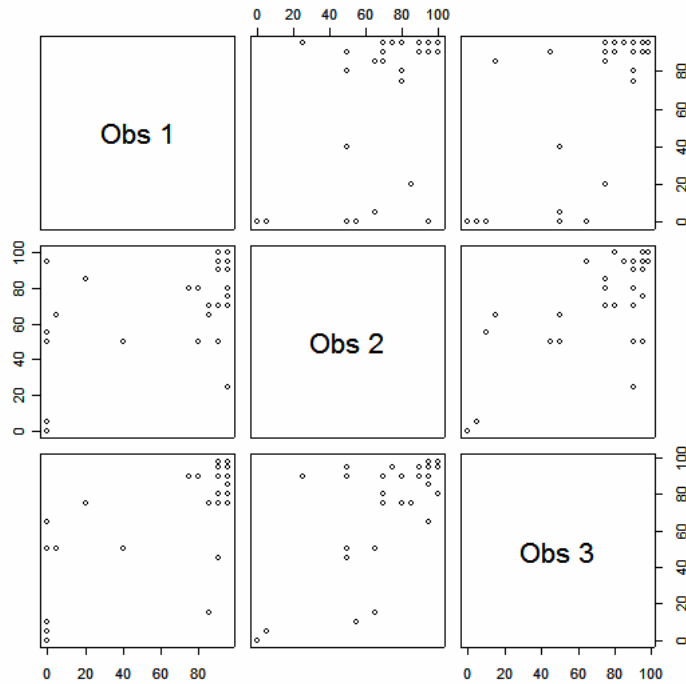
Fig. 12. Pairwise scatter plots of estimates of upper live canopy volume.
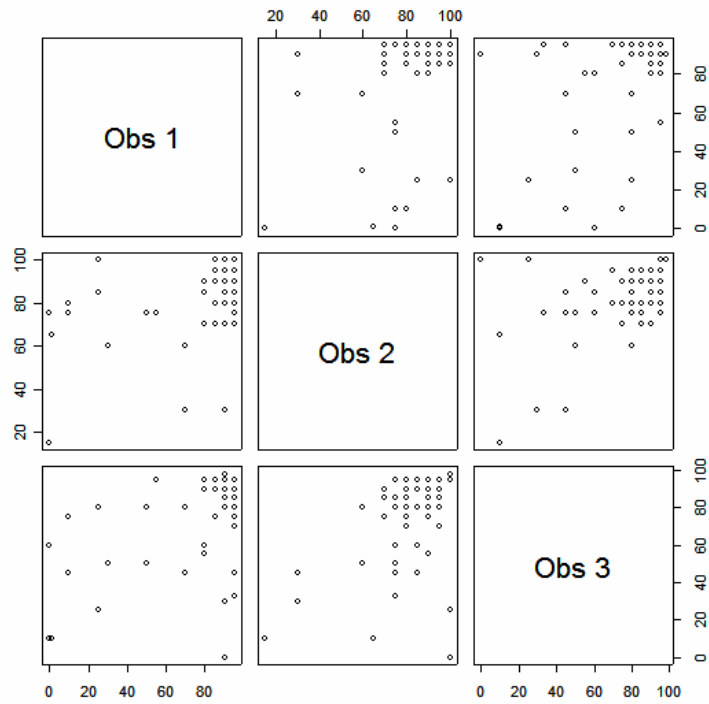
Fig. 13. Pairwise scatter plots of estimates of middle live canopy volume.
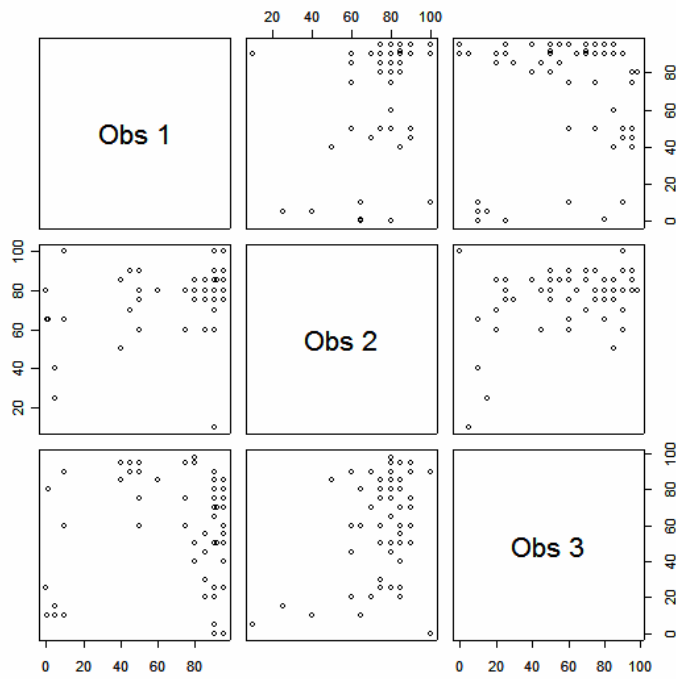
Fig. 14. Pairwise scatterplots of estimates of lower live canopy volume.

Mountain Pine Beetle and Mistletoe

There was no observer variability for the presence/absence of these 2 pests of whitebark pine.

Other Indicators

Five additional indicators of tree health were added after the start of the summer field season. These were instances of rodent chewing, flagging, bark swelling, roughened bark, and oozing sap. These were recorded for 43 trees on 5 transects. Results are summarized below (Table 10). Oozing sap was noted on 7 trees but in only 2 cases did more than one observer note an instance of this on a single tree. Roughened bark was noted on 15 different trees but in only 2 cases did more than one observer record rodent chewing on the same tree. Swelling was noted on 5 trees and in each case only one observer recorded swelling. Flagging was noted on 20 trees but in only 9 cases did 2 or more observers note flagging on the same tree. Rodent chewing was noted on 10 trees and on 7 by 2 or more observers.

Table 10. Instances of 5 potential diagnostics recorded on 43 trees by 3 different observers.

Observer	Oozing sap	Rodent chewing	Flagging	Swelling	Roughened bark
1	3	6	11	2	8
2	5	7	18	2	3
3	3	12	24	1	9

Future Considerations

Stratification

The sampling design for long-term monitoring will almost certainly involve stratified sampling. We have not decided on appropriate strata and indeed one of the goals of this summer's effort is to potentially identify relevant stratifying variables. Some sort of post-stratification may be necessary for analysis of the data collected this summer depending on how many transects are ultimately run. We have given some thought to potential strata, however. One potential pitfall to avoid is to not use strata that will change over time.

Preliminary data on blister rust infection rates is available from data collected over the past 10 years by field crews working for Gardiner District Wildlife Biologist Dan Tyers. Data have been collected on several hundred transects located primarily in the Absaroka-Beartooth Wilderness Area north of Yellowstone National Park. Data were also collected on more than 100 transects

located within the GYE by field crews supervised by USGS Research Biologist Katherine Kendall. Although these transects were not collected using a probability based sampling design they do indicate that infection rates tended to increase with elevation. However, the relationship is not particularly strong.

It may also be logical to stratify based on access with units far from roads, trails, and overnight facilities being assigned sampling weights smaller than more accessible units. Although field crews could certainly camp out during visits to inaccessible units, we need to minimize the need for camping due to safety concerns. Preliminary inquiries have indicated that elevation and access are likely strongly associated, with roads being located disproportionately at lower elevations. Thus, stratification based on access may also serve as a stratification of elevation. A final determination of strata will be made at the end of the 2005 effort.

Observer Variability

While other studies have recognized that field identification of blister rust can be difficult, we know of no other monitoring programs that have explicitly estimated the effects of variability among observers. Our results have indicated substantial variability among observers in several measurements. For certain measurements (e.g., crown ratio), this extensive variability may cause us to reconsider whether the measurement is reliable or contains any value for this program. For other measurements (e.g., primary blister rust severity measures), the variability may imply a need to refine our methods and/or to develop a more effective training program. If training is not sufficient to reduce observer variability to reasonable levels, then we may need to consider incorporating observer variation in our future analyses by incorporating that source of variation in our statistical models.

Temporal Design

Because it is infeasible to obtain a sufficient sample of transects in any one season, our monitoring effort will likely entail a repeating panel design, where the total sample is accumulated over several years and then each panel (i.e., the sample from a given year) is sampled again after several years (probably on the order of 5-10 years). This temporal design works well for whitebark pine because white pine blister rust (the primary focus of our monitoring objective) is a slow acting pathogen that has relatively little inter-annual variation. Thus, sampling a given panel every year would be extremely inefficient. Sampling panels at longer intervals allows us to develop a sufficient sample size over several years while maintaining a reasonable ability for potential changes to be detected. The final panel design will be determined and documented within our monitoring protocol.

Within Stand Variation

Preliminary analysis from our 2004 efforts have indicated that within stand variation may be a minor component of the overall population variance. However, this assessment was based on only six polygons (stands) that had replicate transects within the stand. During 2005 we will attempt to obtain a substantially great sample of within stand replicates and use these data to assess the need for within stand replication in our overall sampling design.

Additional Objectives

Since this project was initiated, additional objectives (discussed above) have been introduced that may be necessary to fully understand the viability and health of whitebark pine in the GYE. Over the next year, we will continue to evaluate and refine these objectives so as to make a determination as to their potential to be incorporated into our monitoring effort.

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APPENDIX D

Pesticides in army cutworm moths (*Euxoa auxiliaris*) from the Greater Yellowstone Ecosystem and their potential consequences to foraging grizzly bears (*Ursus arctos horribilis*)

H.L. Robison, C.C. Schwartz, J.D. Petty, and P.F. Brussard

Abstract: During the summer, grizzly bears (*Ursus arctos horribilis*) in the Greater Yellowstone Ecosystem can each excavate and consume millions army cutworm moths (*Euxoa auxiliaris*) (ACMs) that aggregate in talus. ACMs are agricultural pests and concern exists about whether they contain pesticides that could be toxic to bears. Consequently, in 1999 we collected and analyzed ACMs from six moth aggregation sites. ACMs were screened for 32 pesticides with gas chromatography with electron capture (GCE). Because gas chromatography with tandem mass spectrometry (GC-MS/MS) can be more sensitive than GCE, we revisited one site in 2001 and analyzed a second sample of ACMs with GC-MS/MS. This sample was screened for six pesticides previously screened with GCE and one pesticide not included in the GCE analysis, but approved to control ACMs. Results suggest ACMs contained trace or undetectable levels of pesticides in 1999 and 2001, respectively. Based on chemical levels in ACMs and the number of ACMs bears can consume, we calculated the potential of chemicals to reach physiological toxicity. These results allay concerns that bears are at risk from pesticides. If chemical control of ACMs changes in the future, screening new ACM samples taken from bear foraging sites may be warranted.

APPENDIX E

Army cutworm moth habitat and grizzly bear conservation in the Greater Yellowstone Ecosystem

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Abstract: Army cutworm moths (*Euxoa auxiliaris*) (ACM) are an important food for grizzly bears (*Ursus arctos horribilis*) in the Rocky Mountains, U.S.A. ACMs lay their eggs in soil in low elevations in the Great Plains and intermountain west (hereafter low elevations), and larvae emerge in spring to feed on emerging plants. They pupate underground, emerge as adults in late June, and migrate from low elevations to alpine areas in the Rocky Mountains. Here ACMs consume flower nectar at night and form large aggregations in talus during the day. Bears excavate ACMs from talus and consume them in the millions from July-September. Grizzly bears can eat close to half their yearly energy needs in 30 days feeding on moths.

The U.S. Fish and Wildlife Service's conservation strategy for the Yellowstone grizzly bear allows the population to expand into biologically suitable and socially acceptable areas beyond the Primary Conservation Area (PCA). The conservation strategy requires use of georeferenced habitat data to aid in monitoring the four major Yellowstone grizzly bear foods (ACMs, cutthroat trout, WBP seeds, and winter-killed ungulates) and to identify habitats into which bears may expand.

To this end, we developed models of high elevation ACM habitat in the Greater Yellowstone Ecosystem (GYE) with the purpose of creating a tool with which bear scientists and managers can identify potential ACM habitats into which grizzly bears may expand. We developed presence/random models using attribute data extracted from bear locations ($N = 490$) collected during aerial surveys from 1986-2002 and random points generated in a GIS. The variables we used in developing our models included elevation, heat load index, topographic roughness index, aspect, rate of change along contours, slope, rate of change in slope, and Landsat TM bands 1,2,3,4,5,7. Because there is likely error associated with bear locations from aerial surveys, we buffered bear locations in 30 m intervals up to 540 m, a distance approximate to that used to define moth sites. Weights at each interval decreased as a function of distance bear locations. We developed general linear models (GLMs) and generalized additive models (GAMs) in S-plus and GAMs in the program generalized regression analysis for spatial prediction (GRASP). We then used Bayes' equation to update probabilities generated from GRASP, GAMs and GLMs (e.g., by using geology layers). Our models indicate that of the significant variables, elevation, aspect, and rate of change in slope are most important. This is not surprising as moth sites are located in cirque basins and talus slides below rock headwalls. TM bands were significant and

important in the following order4>3>2>1. Our model outputs are displayed as maps showing probabilities of moth habitat in the GYE. Models were generated using a subset of bear locations and are being tested with locations not used in model development.

APPENDIX F

Army cutworm moth nectar plants

Hillary Robison

Project objective

      In this project, I am investigating which flowers ACM are visiting in the alpine. Observing cryptically-colored ACMs feed on nectar plants is difficult and is complicated by precipitous terrain and the fact ACMs forage at night. Hence, I am trying to determine on which nectar plants ACMs feed based on pollen retrieved from their heads and mouthparts.

Methods

      In 2001, we established four to five 2 x10 m transects at different elevations at four high elevation sites – one in the Absaroka range and one in the Teton range. One site in the Absaroka range was revisited four times to investigate temporal differences in flower use. Transects were visited during the day, and all inflorescences were counted and flowers were keyed to species. Sites were revisited a night to attempt to observe ACMs feed and to collect ACMs as they visited flowers.

Results to date

      To date, I have identified pollen on ACMs collected at different transects from the one site I revisited as well as from one of the three additional sites. Results to date indicate that ACMs carry pollen from local alpine flower species as well as from plants from lower elevations, which they may be visiting enroute to high elevations.

APPENDIX G

Management of habituated grizzly bears in North America: report from a workshop

Kerry A. Gunther, Kathy Tonnessen, Peter Dratch, and Chris Servheen

Abstract: Habituated wildlife have a unique legacy in the national parks: the likelihood of viewing predators has been an attraction for so many visitors, and yet proximity to visitors can put predators at risk. The National Park Service sponsored a workshop in October, 2003, that brought together bear management specialists from the Rocky Mountains, North Cascades, Canada, and Alaska to discuss the current science and management of habituated grizzly bears. State and federal agency biologists, university and agency scientists, and bear managers shared their experiences and strategies for dealing with bears that have become habituated to humans, but not conditioned to human food. Policy representatives were called on to the review issues associated with liability and legal requirements relating to bear-human interactions. Grizzly bear researchers shared the latest information on bear behavior and demographics related to habituation. This publication summarizes the major recommendations and knowledge gaps discussed during the workshop.

APPENDIX H

Interactions between wolves and female grizzly bears with cubs in Yellowstone National Park

Kerry A. Gunther and Douglas W. Smith

Gray wolves (*Canis lupus*) were extirpated from Yellowstone National Park (YNP) in the 1920s, then reintroduced back into the park from 1995 to 1996 to restore ecological integrity. Prior to reintroduction, the potential effects of gray wolves on the regions threatened grizzly bear (*Ursus arctos*) population were evaluated. It was predicted that wolves would reduce the availability of winter-killed ungulate carcasses available for bears to scavenge, and that wolves would occasionally kill grizzly bear cubs. It was also predicted that solitary adult grizzly bears would occasionally usurp wolf-killed ungulate carcasses from wolf packs, but that females with cubs would avoid wolf packs at ungulate carcasses due to the potential danger to cubs. From 1995-2002, we documented 96 grizzly bear-wolf interactions. Only 15 of the interactions involved females with young. Of these, 8 involved females with cubs-of-the-year. We documented 1 wolf-killed ungulate carcass that was successfully usurped from a wolf pack by an adult female grizzly bear accompanied by cubs-of-the-year, and 2 incidents where cubs were killed by wolves at large ungulate carcasses. These observations lend insight into interference competition between wolves and grizzly bears and factors that contribute to interspecific killing of grizzly bear cubs by wolves.