

2022-01-27

The Effects of the Kenow Wildfire on the Bat Community in Waterton Lakes National Park, Alberta

Low, Erin Blair

Low, E. B. (2022). The Effects of the Kenow Wildfire on the Bat Community in Waterton Lakes National Park, Alberta (Master's thesis, University of Calgary, Calgary, Canada). Retrieved from <https://prism.ucalgary.ca/http://hdl.handle.net/1880/114359>

Downloaded from PRISM Repository, University of Calgary

UNIVERSITY OF CALGARY

The Effects of the Kenow Wildfire on the Bat Community in Waterton Lakes National Park,
Alberta

by

Erin Blair Low

A THESIS

SUBMITTED TO THE FACULTY OF GRADUATE STUDIES
IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE
DEGREE OF MASTER OF SCIENCE

GRADUATE PROGRAM IN BIOLOGICAL SCIENCES

CALGARY, ALBERTA

JANUARY, 2022

© Erin Blair Low 2022

Abstract

Fire is one of the most important natural disturbances shaping forest communities. Fire impacts bat communities by changing forest structure, foraging opportunities, and roost availability. I examined the effects of the Kenow wildfire on the bat community in Waterton Lakes National Park (WLNP), Alberta. The Kenow wildfire was a severe fire that burned 38% of the park in September 2017. I radio-tracked female and male *Myotis lucifugus* to examine roosting behaviour and roost-tree availability two- and three-years post-fire. Reproductive female maternity colonies were found exclusively in buildings in the Waterton townsite. Males and non-reproductive females were tracked to tree and rock roosts in both burned and unburned areas. Roost-tree availability does not appear to have changed immediately after the wildfire. WLNP bat activity was examined using acoustic detectors to record bat echolocation calls at thirteen sites for three years before and three years after the wildfire. I analyzed echolocation recordings to examine bat activity changes pre- and post-fire as well as between burned and unburned areas. The Kenow wildfire negatively affected *Eptesicus fuscus*/*Lasionycteris noctivagans*, *Lasiurus cinereus*, and *Myotis evotis*. 40 kHz *Myotis* activity, likely predominately *M. lucifugus*, did not change pre- to post-fire. However, activity increased in burned areas and decreased in unburned areas. *Myotis lucifugus* are opportunistic and were likely able to adapt quickly to the changes in foraging habitat, insect communities, and roosting opportunities after the wildfire. *Eptesicus fuscus*/*L. noctivagans*, *L. cinereus*, and *M. evotis* were likely less adaptable to the significant ecological changes caused by the wildfire. However, the results are for the first three years post-fire and it may take time for the positive effects of the Kenow wildfire to be observed.

Keywords: post-fire, insectivorous bat, acoustic analysis, roost selection, nightly activity

Acknowledgements

I would like to thank my supervisor Robert Barclay for his ongoing guidance, support and encouragement. You have made me a better researcher and writer, and our ongoing banter made for very enjoyable and amusing discussions. I would also like to acknowledge my examination committee, Mary Reid, Mindi Summers, and Darren Bender. Thank you for your support and insightful feedback.

Thank you to all the field assistants and volunteers who joined me in the field: Mary Blair, Meaghan Bouchard, Mike Kelly, Jorin Klaehn, Caitlin Low, Ivor Low, Kelsey Low, Kaitlin Machacek, Kelly Melrose, Sara Pearce Meijerink, Emma Micalizzi, Jessica Theoret, and Joel Van Riper. Meaghan, Jorin, Sara, and Jessica, thank you for dropping everything to come and help me numerous times in the field when I needed you. You put up with all sorts of challenging field conditions and were constantly encouraging and enthusiastic. Emma, you were the best possible field partner anyone could hope for, and I am forever grateful to you for answering my endless messages about all the R coding errors I kept running into.

Lisa Sims, thank you for being my guide into graduate school and for providing ongoing insight and support. Many thanks to all my other lab mates in the Barclay lab for a stimulating and rewarding working environment. Thank you to Samuel Robinson for all the time spent helping my lab mates and I with R, and teaching us about modeling biological data. Katie Florko, words cannot express how grateful I am for the infinite modeling and R wizardry magic you possess. I would still be in the depths of the statistics world without you. Thank you to my colleagues with the Wildlife Conservation Society Canada (WCSC), Alberta Community Bat Program, Cory Olson, Susan Holroyd and Mike Kelly. You put up with endless late-night messages while I was in the depths of acoustic analysis and searching for papers on a certain topic. I would like to thank Cori Lausen and Jason Rae with WCSC for providing the bat species filters for the acoustic analysis. Cori, thank you also for your help trouble shooting issues with the detectors, and sharing data from your 2011/2012 Waterton Lakes National Park (WLNP) bat inventory.

I would like to thank everyone I had the opportunity to work with at WLNP. Jen Carpenter, Barb Johnston, Helena Mahoney, and Adam Collingwood, thank you for the ongoing

logistical help over the last three years. I would also like to thank the ecological integrity monitoring technicians who joined me in the field: Anne Blondin, Dylan Brassard, Erica Hanna, Beth Lucier, and Brad Yee.

Finally, I would like to thank my family, Kelsey, Caitlin, Mom, and Dad. Thank you for being my mental and physical support system. I am forever grateful for all the times you joined me in the field, putting up with everything from climbing straight up mountains, navigating obstacle courses of downed trees and streams, conducting vegetation surveys in the pouring rain, scooping bat guano out of bags, and carrying packs full of bat trapping equipment and a week's worth of backcountry camping gear. You have helped me in so many other ways throughout this process too; acting as a sounding board, answering my phone calls at all hours of the day, editing thesis drafts...

Personal funding for this project was provided by the University of Calgary and the Natural Sciences and Engineering Research Council of Canada (NSERC Canada Graduate Scholarship – Master's). Research funding was provided by Parks Canada (Waterton Lakes National Park), Alberta Conservation Association (Grants in Biodiversity), and an NSERC Discovery Grant to Robert Barclay.

Dedication

My family. For their constant support and encouragement.

Table of Contents

Abstract.....	ii
Acknowledgements.....	iii
Dedication	v
Table of Contents.....	vi
List of Tables	ix
List of Figures	x
List of Appendices Tables.....	xi
List of Appendices Figures	xiii
 Chapter 1: General Introduction.....	 1
1.1 Fire and Bats.....	1
1.1.1 Fire	1
1.1.2 Bats	2
1.1.3 The effect of fire on bats	4
1.2 Study Area	7
1.2.1 Kenow wildfire.....	7
1.2.2 Waterton Lakes National Park.....	9
1.3 Studying Bats.....	12
1.4 Thesis Overview	13
 Chapter 2: Little Brown <i>Myotis</i> Roosting Behaviour after the Kenow Wildfire	 15
2.1 Introduction.....	15
2.1.1 Fire and bats	15
2.1.2 Hypothesis and predictions	16
2.2 Methods	16
2.2.1 Study species	16
2.2.2 Bat capture and radio-tracking.....	18
2.2.3 Roost identification	20
2.2.4 Roost tree availability surveys.....	21
2.2.5 Statistical analysis.....	22
2.3 Results	23
2.3.1 2019/2020 captures	23
2.3.2 Female little brown <i>Myotis</i>	24
2.3.3 Male little brown <i>Myotis</i>	27
2.3.4 Tree roost availability surveys.....	33
2.4 Discussion.....	34
2.4.1 2019/2020 captures	34
2.4.2 Female little brown <i>Myotis</i>	36

2.4.3 Male little brown <i>Myotis</i>	37
2.4.4 Tree roost availability surveys	39
2.4.5 Study limitations	40
2.4.6 Future research	41
2.4.7 Management recommendations.....	41
2.5 Conclusion	42
 Chapter 3: Changes in Bat Activity after the Kenow Wildfire.....	44
3.1 Introduction.....	44
3.1.1 Fire and bats	44
3.1.2 Acoustically monitoring bats	45
3.1.3 Hypothesis and predictions	48
3.2 Methods	49
3.2.1 Study species	49
3.2.2 Acoustic surveys	51
3.2.3 Detector deployment	54
3.2.4 Acoustic call processing	54
3.2.5 Statistical analysis	57
3.3 Results	61
3.3.1 Species analysis	61
3.3.2 40 kHz <i>Myotis</i> feeding buzz analysis	64
3.3.3 Generalized linear mixed model selection	64
3.3.4 Fire metrics' results	69
3.3.5 Site and environmental covariates' results	70
3.4 Discussion	81
3.4.1 40 kHz <i>Myotis</i> activity and feeding.....	81
3.4.2 Big brown and silver-haired bat activity.....	83
3.4.3 Hoary bat activity.....	86
3.4.4 Long-eared <i>Myotis</i> activity	89
3.4.5 Study assumptions and limitations	90
3.4.6 Future research	93
3.4.7 Management recommendations.....	94
3.5 Conclusion	95
 Chapter 4: General Conclusions.....	96
 References	99
 Appendix A: Fire History of Waterton Lakes National Park, Alberta	125
Appendix B: Historic Mountain Pine Beetle Distribution in Waterton Lakes National Park, Alberta.....	130

Appendix C: Metadata for the little brown <i>Myotis</i> radio-tracked in Waterton Lakes National Park, Alberta in 2019 and 2020	131
Appendix D: Metadata for the Acoustic Detector Sites, the Waterton Townsite, and the Weather Stations	136
Appendix E: Acoustic Detector Deployment and Call Processing Methodology.....	138
E.1 Acoustic Detector Deployment	138
E.2 Acoustic Call Processing	139
Appendix F: Bat Echolocation Call Characteristics.....	144
Appendix G: Kaleidoscope Noise Scrubber and Zero-Cross Analysis Limitations	150
G.1 Overview	150
G.2 Methods.....	150
G.3 Results.....	154
G.3.1 Missed additional species	154
G.3.2 Missed species from Kaleidoscope Pro noise files.....	155
G.3.3 Missed species from AnalookW noise files.....	155
G.4 Discussion	163
G.4.1 Missed additional species	163
G.4.2 Missed species from Kaleidoscope Pro noise files.....	163
G.4.3 Missed species from AnalookW noise files.....	164
G.5 Conclusion.....	164
Appendix H: Species/Species Groups' Mean Passes/Night	167
Appendix I: 40 kHz <i>Myotis</i> Feeding Buzz Analysis	171
Appendix J: 40 kHz <i>Myotis</i> Activity Analysis	175
Appendix K: Big Brown/Silver-Haired Bat Activity Analysis.....	177
Appendix L: Hoary Bat Activity Analysis	179
Appendix M: Long-Eared <i>Myotis</i> Activity Analysis	182
Appendix N: Significant Site and Environmental Variables for the Generalized Linear Mixed Models	184

List of Tables

Table 3.1. The possible explanatory variables originally included in the generalized linear mixed models (GLMMs) that examined the effect of the Kenow wildfire on bats in Waterton Lakes National Park, Alberta.	59
Table 3.2. The total number of bat echolocation passes, and the percentage of total bat activity for each species/species group detected from 2015 to 2020 in Waterton Lakes National Park, Alberta.	63
Table 3.3. The total number of echolocation passes, the total detector nights monitored, and the mean passes/night \pm SE for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta.	64
Table 3.4. Model selection factors influencing the echolocation counts for 40 kHz <i>Myotis</i> in Waterton Lakes National Park, Alberta from 2015 to 2020	66
Table 3.5. Model selection factors influencing the echolocation counts for big brown/silver-haired bats in Waterton Lakes National Park, Alberta from 2015 to 2020	67
Table 3.6. Model selection factors influencing the echolocation counts for hoary bats in Waterton Lakes National Park, Alberta from 2015 to 2020	68

List of Figures

Fig. 1.1. The location of the study in Waterton Lakes National Park, Alberta.	8
Fig. 1.2. The different natural subregions found throughout Waterton Lakes National Park, Alberta.....	10
Fig. 2.1. A little brown <i>Myotis</i> with a radio-transmitter attached, and radio-tracking the bat in a burned area in Waterton Lakes National Park, Alberta.	20
Fig. 2.2. The locations of the capture sites and roosting locations of the female little brown <i>Myotis</i> tracked in Waterton Lakes National Park, Alberta in 2019 and 2020.	25
Fig. 2.3. The locations of the unburned capture sites and roosting locations of the male little brown <i>Myotis</i> tracked in Waterton Lakes National Park, Alberta in 2020.....	28
Fig. 2.4. The locations of the burned capture sites and roosting locations of the male little brown <i>Myotis</i> tracked in Waterton Lakes National Park, Alberta in 2020.....	29
Fig. 2.5. Natural roosts selected by male little brown <i>Myotis</i> in Waterton Lakes National Park, Alberta in 2020.....	31
Fig. 2.6. The mean (\pm SE) Euclidean distance male little brown <i>Myotis</i> traveled between capture sites and roosts (km) for trees, rock crevices and boulders in the unburned and burned areas in Waterton Lakes National Park, Alberta in 2020.....	32
Fig. 2.7. The locations of the 2019 and 2020 vegetation surveys completed in Waterton Lakes National Park, Alberta.....	34
Fig. 3.1. Examples of bat echolocation passes consisting of the three different call types and social calls.	47
Fig. 3.2. Examples of six of the thirteen acoustic detector sites monitored from 2015 to 2020 in Waterton Lakes National Park, Alberta.	52
Fig. 3.3. The site variation among the 13 acoustic sites monitored from 2015 to 2020 in Waterton Lakes National Park, Alberta.	53
Fig. 3.4. The acoustic detector deployment set-up at Cameron Lake, Waterton Lakes National Park, Alberta	55
Fig. 3.5. The survey date range for each acoustic detector site over the six-year study in Waterton Lakes National Park, Alberta.	63
Fig. 3.6. The 40 kHz <i>Myotis</i> nightly activity plotted for each fire metric.	72
Fig. 3.7. The big brown/silver-haired bat nightly activity plotted for each fire metric.....	74
Fig. 3.8. The hoary bat nightly activity plotted for each fire metric.....	76
Fig. 3.9. The long-eared <i>Myotis</i> nightly activity plotted for each fire metric.....	78
Fig. 3.10. The nightly proportion of feeding buzzes for 40kHz <i>Myotis</i> plotted for each fire metric.	80
Fig. 3.11. Hoary bat nightly activity plotted for each year of the acoustic study in Waterton Lakes National Park, Alberta.....	88

List of Appendices Tables

Table A.1. The prescribed fires and wildfires that occurred during the years of the acoustic surveys (2015 to 2020) in Waterton Lakes National Park, Alberta.	129
Table C.1. Metadata for the female little brown <i>Myotis</i> radio-tracked in Waterton Lakes National Park, Alberta in 2019 and 2020.	131
Table C.2. Metadata for the male little brown <i>Myotis</i> radio-tracked in Waterton Lakes National Park, Alberta in 2020.	133
Table D.1. Site metadata for the Waterton townsite and each acoustic detector site monitored from 2015 to 2020 in Waterton Lakes National Park, Alberta.	136
Table D.2. Site metadata for each weather station used to compile environmental data for the thirteen different acoustic detector sites in Waterton Lakes National Park, Alberta....	137
Table F.1. Echolocation call characteristics for the different bat species/species groups found in Waterton Lakes National Park, Alberta.	144
Table G.1. The percent of subsampled full spectrum bat files for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020 that contained additional species that were missed in the zero-cross files.	157
Table G.2. The number of subsampled full spectrum bat files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the additional species that were missed in the zero-cross files. ...	158
Table G.3. The percent of subsampled Kaleidoscope Pro noise files for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020 that contained bats.	159
Table G.4. The number of subsampled Kaleidoscope Pro noise files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the percent that contained bats.....	160
Table G.5. The percent of subsampled AnalookW noise files for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, that contained bats.	161
Table G.6. The number of subsampled AnalookW noise files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the percent that contained bats.....	162
Table H.1. The mean passes/night (\pm SE) for 40 kHz <i>Myotis</i> for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta.	167
Table H.2. The mean passes/night (\pm SE) for big brown/silver-haired bats for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta.....	168
Table H.3. The mean passes/night (\pm SE) for hoary bats for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta.	169
Table H.4. The mean passes/night (\pm SE) for long-eared <i>Myotis</i> for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta.	170

Table I.1. The percent of subsampled 40 kHz <i>Myotis</i> passes that contained feeding buzzes, for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020.	171
Table I.2. The number of subsampled files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the percent of passes that contained 40 kHz <i>Myotis</i> feeding buzzes.	172
Table I.3. Kruskal-Wallis rank sum test results for the proportion of echolocation calls that contained feeding buzzes for 40kHz <i>Myotis</i> (40kFeedBuzz) in Waterton Lakes National Park, Alberta from 2015 to 2020.	173
Table I.4. Dunn’s test of multiple comparisons for the three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) tested in the 40k <i>Myotis</i> feeding buzz (40kFeedBuzz) Kruskal-Wallis rank sum test.	174
Table J.1. Variable names, effect direction, parameter estimates, standard errors, and <i>P</i> -values for each 40 kHz <i>Myotis</i> (40kMyotis) fire metric model.	175
Table J.2. Post hoc results for the 40 kHz <i>Myotis</i> (40kMyotis) fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn).	176
Table K.1. Variable names, effect direction, parameter estimates, standard errors, and <i>P</i> -values for each big brown/silver-haired bat (EPFULANO) fire metric model.	177
Table K.2. Post hoc results for the big brown/silver-haired bat (EPFULANO) fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn).	178
Table L.1. Variable names, effect direction, parameter estimates, standard errors, and <i>P</i> -values for each hoary bat (LACI) fire metric model.	179
Table L.2. Post hoc results for the hoary bat (LACI) fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn).	180
Table L.3. Kruskal-Wallis rank sum test results for hoary bats (LACI) in Waterton Lakes National Park, Alberta from 2015 to 2020 for the fire metric PrePostUnburnBurn.	181
Table L.4. Dunn’s test of multiple comparisons for the fire metric PrePostUnburnBurn tested in the hoary bat (LACI) Kruskal-Wallis rank sum test.	181
Table M.1. Kruskal-Wallis rank sum test results for long-eared <i>Myotis</i> (MYEV) in Waterton Lakes National Park, Alberta from 2015 to 2020.	182
Table M.2. Dunn’s test of multiple comparisons for the three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) tested in the long-eared <i>Myotis</i> (MYEV) Kruskal-Wallis rank sum test.	183

List of Appendices Figures

Fig. A.1. Fires (both wild and prescribed) that occurred in Waterton Lakes National Park, Alberta from the 1700s to 2020.	125
Fig. A.2. Prescribed fires and wildfires that occurred in Waterton Lakes National Park, Alberta from 1990 to 2020.	126
Fig. A.3. Prescribed fires and wildfires that occurred in Waterton Lakes National Park, Alberta from 2015 to 2020.	127
Fig. A.4. The mean number of fires in Waterton Lakes National Park, Alberta, per year for each group size (ha) and time period (1750 to 2020).	128
Fig. B.1. The mountain pine beetle (<i>Dendroctonus ponderosae</i>) distribution in Waterton Lakes National Park, Alberta from 1977 to 2016.	130
Fig. F.1. Examples of hoary bat passes.	145
Fig. F.2. Examples of big brown/silver-haired bat passes.	145
Fig. F.3. Examples of bat passes labeled as “LowF”	146
Fig. F.4. Examples of bat passes labeled as “30k”.	146
Fig. F.5. Examples of long-eared <i>Myotis</i> passes.	147
Fig. F.6. Examples of eastern red bat passes.	147
Fig. F.7. Examples of 40 kHz <i>Myotis</i> passes.	148
Fig. F.8. Examples of bat passes labeled as “HighF”	148
Fig. F.9. Examples of bat passes labeled as “NOTBAT”.	149
Fig. F.10. Examples of bat passes that contain multiple species.	149
Fig. G.1. Examples of the same bat pass displayed in full spectrum (Wildlife Acoustics’ Kaleidoscope Lite bat analysis software, version 5.4.0) and zero-cross (Titley Scientific’s AnalookW bat analysis software, version 4.5s).	151
Fig. G.2. Examples of bat echolocation passes (Wildlife Acoustics’ Kaleidoscope Lite bat analysis software, version 5.4.0) that were identified as “noise” by Kaleidoscope Pro and were automatically moved to a separate directory called “NOISE” during the conversion process.	152
Fig. G.3. Examples of the same bat pass displayed in full spectrum (Wildlife Acoustics’ Kaleidoscope Lite bat analysis software, version 5.4.0) and zero-cross (Titley Scientific’s AnalookW bat analysis software, version 4.5s).	153

Chapter 1: General Introduction

1.1 Fire and Bats

1.1.1 Fire

Although there is natural climate variability, anthropogenic influences (e.g. greenhouse gas emissions) are causing unprecedented climatic change (IPCC 2014). The increasing global average surface temperature is influencing climate-related extreme events including heatwaves, droughts, floods, and wildfires (IPCC 2014). Globally, the number, duration, and severity of wildfires are predicted to continue to rise due to increasing temperatures and changes in the precipitation cycle (Gillett et al. 2004, Westerling et al. 2006, Rocca et al. 2014, Stephens et al. 2014). In addition, current activities (i.e. land use and management changes) as well as historical practices (i.e. fire suppression) have intensified wildfire size and severity (Stephens et al. 2014, Singleton et al. 2019). The composition, structure, and function of a forest is heavily influenced by the fire regime (i.e. fire frequency, size, seasonality, and severity) (Heinselman 1973, Wright and Bailey 1982, Flannigan et al. 2000), with the fire regime being intricately linked to the climate of that area (Swetnam 1993, Flannigan and Harrington 1988). Fire regimes respond rapidly to changes in climate (Flannigan et al. 2000). Therefore, if current climate projections hold true, there could be immediate and significant impacts to forest communities (Flannigan et al. 2000).

Fire is an important natural disturbance shaping forest communities (Waldrop et al. 1992, Brose et al. 2001, Krawchuk et al. 2020), creating a mosaic of habitat patches and altering species composition and forest structure (Dale et al. 2001, Johnstone et al. 2016). Wildfires create ecosystem diversity, encouraging seed germination and new growth, recycling nutrients, while also killing trees and soil seed banks (Wright and Bailey 1982, Whelan 1995). The resulting impact on faunal species from fire-altered habitats include new food resources and roosting and nesting opportunities, reduced habitat complexity (increased visibility and movement), and changes to parasite, prey and predator communities (Gibbons and Lindenmayer 2002, Jaffe and Isbell 2009, Hancock et al. 2011, Romme et al. 2011, Pausas and Parr 2018). Although fire promotes landscape heterogeneity (Armitage and Ober 2012) and maintains suitable habitat

for wildlife (Pausas and Parr 2018), it is unknown what the effects of rapidly changing fire regimes will be on forest biodiversity (Fontaine and Kennedy 2012, Blakey et al. 2019).

1.1.2 Bats

Bats are a diverse order of mammals (Voigt and Kingston 2016), playing essential roles in ecosystem services (Kunz et al. 2011, Russo et al. 2018b). Insectivorous bats are the main predators of nocturnal crop and forest pests (Cleveland et al. 2006, Dodd et al. 2012), with an annual agricultural value in North America estimated to be in the billions of dollars (Boyles et al. 2011). Globally, bat populations appear to be declining (Jones et al. 2009) and population recovery is slow when there are high mortality rates (Voigt and Kingston 2016). Due to their unique life history, bats are particularly susceptible to anthropogenic pressures (Loeb et al. 2015, Voigt and Kingston 2016). North American bats have a low reproductive rate (typically one litter per year of one to three pups) (Barclay and Harder 2003, Barclay et al. 2004), are long-lived (Brunet-Rossinni and Austad 2004, Munshi-South and Wilkinson 2010), have high metabolic rates and thus high food requirements (Voigt and Kingston 2016), and use energetically expensive sustained flight (Voigt and Lewanzik 2011).

In Alberta, Canada, bats are active from approximately April to September (Lausen 2007, Olson and Flach 2016). Bat species in Alberta migrate to either more southern locations (e.g. California and Mexico) (Cryan 2003) or undergo regional movements to their hibernation sites (Schowalter 1980). Migratory bats begin to leave the province and return to their overwintering areas starting by the middle of July (Baerwald and Barclay 2011), with the hibernating species returning to their hibernacula starting in mid-August (Schowalter 1980). Bats in Alberta mate in the fall and winter (Wimsatt 1945, Cryan et al. 2012, Olson and Flach 2016), with females becoming pregnant in the spring, and typically giving birth from mid June to late July (Koehler and Barclay 2000, Solick and Barclay 2007). Pups are flying approximately three to five weeks after birth and are foraging independently after approximately five to seven weeks (Kunz 1971, Buchler 1980, Koehler and Barclay 2000).

During the summer, bats in Alberta use a variety of roosting structures including trees (e.g. in cavities, under bark, in foliage), rock crevices, and anthropogenic structures (e.g. buildings, bridges, bat houses) (Nagorsen and Brigham 1993). The roosting patterns of bats are

influenced by factors such as roost abundance and availability, as well as the distribution and abundance of food sources (Kunz 1982). Although both roosts and foraging habitat are critical resources to bats, it is difficult to determine if either (or both) are limiting factors affecting the populations of bats (Humphrey 1975, Fenton 1997). As flight enables bats to use a variety of habitat types across a large area, bats can commute between different environments for roosting versus foraging quickly and with relatively low energetic costs (Norberg 1990, Fenton 1997). To understand if roosts or food are limited for bats in a particular area, knowledge of the quality and distribution of these resources, as well as the species-specific requirements is needed (Fenton 1997, Salinas-Ramos et al. 2020). To assess roost availability, it is also important to consider that many species of bats show strong preferences for specific roost types (e.g. cavities, bark, buildings) (Kunz and Lumsden 2003).

Bats are currently facing unprecedented threats, including land use changes (e.g. urbanization and industrial development) (Russo and Ancillotto 2015, Voigt and Kingston 2016), habitat loss and fragmentation (Lacki et al. 2010, Fuentes-Montemayor et al. 2013), active persecution (Mickleburgh et al. 2002), pesticides and other pollutants (Oliveira et al. 2021), as well as climatic change (Jones et al. 2009, Sherwin et al. 2013). Recently, some North American bat species have been further threatened by white-nose syndrome (WNS) and wind energy development (Loeb et al. 2015). WNS syndrome is a disease caused by the fungus *Pseudogymnoascus destructans*, and has killed millions of hibernating bats since it was first introduced in North America in 2006 (Frick et al. 2016). The fungus causes bats to arouse from torpor frequently throughout hibernation, depleting their fat reserves before insects emerge in the spring (Czenze et al. 2017, Mayberry et al. 2018). As of fall 2021, WNS has been found in Canada as far west as Manitoba, however, it has also been detected in Montana and Washington state (U.S. Fish and Wildlife Service 2021). Wind energy development primarily affects migratory species of bats (Arnett and Baerwald 2013) and causes mortalities due to barotrauma (i.e. lung damage due to the air-pressure change that occurs near to the turbine blades) and being struck by the blades (Baerwald et al. 2008, Loeb et al. 2015). Population projection modeling has suggested that wind turbines may significantly reduce population sizes in the coming years (Frick et al. 2017, Friedenberg and Frick 2021).

More frequent and higher-severity wildfires are another possible threat to bats (Bosso et al. 2018). The distribution of bats across different habitats is influenced by microclimatic conditions, vegetation structure, and resource availability, including distances to water, and foraging and roosting sites (Rainho et al. 2010, Rainho and Palmeirim 2011, Dias-Silva et al. 2018). Many of these factors are heavily influenced by the fire regime of the area (Bosso et al. 2018). Thus, the resulting response by bats to wildfire is complex due to a variety of interactions between the fire regime (e.g. severity, frequency, season), regional climate, habitat type, and the individual species' biology (Perry 2012, Griffiths et al. 2015, Kelly et al. 2017, Law et al. 2018, Steel et al. 2019). Overall, however, the effect of wildfire on bats is understudied and poorly understood (Perry 2012, Buchalski et al. 2013, Law and Blakey 2021).

1.1.3 The effect of fire on bats

There have been several studies on the effects of prescribed fire on bat communities (Boyles and Aubrey 2006, Loeb and Waldrop 2008, Silvis et al. 2016), as well as fire-surrogate treatments such as overstory thinning (Humes et al. 1999, Menzel et al. 2002, Patriquin and Barclay 2003, Titchenell et al. 2011, Thomas et al. 2019). However, wildfires can differ from silvicultural treatments in terms of fire extent, intensity, and time of year (Buchalski et al. 2013, Lawes et al. 2015, Perry and McDaniel 2015). Studies on the effects of wildfires on bats have varied in terms of their location and the species of bats involved, time of year the fire occurred, size, frequency, and severity of the fire, length of time since the fire occurred, as well as the data available for comparisons (e.g., Perry 2012, Doty et al. 2016, Steel et al. 2019, Jung 2020, Starbuck et al. 2020, Ancillotto et al. 2021). As such, the apparent effects of wildfire on bat communities have also varied. Studies have found that fire (both prescribed and wildfire) generally have an overall neutral or positive effect on most bat species (Malison and Baxter 2010a, Armitage and Ober 2012, Buchalski et al. 2013, Inkster-Draper et al. 2013, Cox et al. 2016, Austin et al. 2018a, 2020, Braun de Torrez et al. 2018a, Steel et al. 2019, Ancillotto et al. 2021, Blakey et al. 2021, Taillie et al. 2021). However, species-specific negative responses have also been reported (Blakey et al. 2019, Steel et al. 2019, Jung 2020, Starbuck et al. 2020, Ancillotto et al. 2021). Overall, fire impacts insectivorous bat communities by changing the

foraging habitat, insect communities (i.e. prey), and roosting opportunities (Lacki et al. 2009, 2017, Armitage and Ober 2012, Buchalski et al. 2013).

Forest structure affects bat species' spatial distribution (Patriquin and Barclay 2003, Blakey et al. 2017) as different species have morphological and echolocation characteristics that make them suited for specific habitats (Schnitzler et al. 2003) and foraging styles (i.e. clutter, edge, or open specialists) (Blakey et al. 2019). Fires result in altered habitats with distinct edges, reduced vegetative complexity in the midstory and understory, lower tree densities and more-open canopies (Boyles and Aubrey 2006, Perry 2012, Jung 2020). Large bodied, lower echolocation-frequency, less-maneuverable bat species typically avoid cluttered fly-ways as dense understory and canopy vegetation can interfere with flight, echolocation, and foraging success (Buchalski et al. 2013, Denzinger and Schnitzler 2013, Cox 2016). Although densely vegetated environments reduce flight efficiency (Norberg 1981), via physical (Brigham et al. 1997a) and acoustical clutter (Miller and Degn 1981), open habitats are avoided by some bat species due to increased predation risks and winds that affect flight and prey capture (Verboom and Spoelstra 1999). Smaller bodied, higher echolocation frequency, more maneuverable bats, can exploit a larger range of habitats and are less affected by clutter and tree density (Patriquin and Barclay 2003, Sleep and Brigham 2003). Overall, however, bats are highly mobile (Henry et al. 2007), and are thus likely not confined to one area, exploiting a variety of habitat patches at a relatively large spatial scale (Patriquin and Barclay 2003).

In addition to habitat use based on the bat species' ecomorphology, the availability of foraging opportunities also influences their distribution (Erickson and West 2003, Lacki et al. 2009). Past research has suggested that the increase in abundance of pyrophilous insects lasts for the first one to three years after a fire (Buchalski et al. 2013) with changes to the diversity and abundance of insect communities up to several decades post-fire (Buddle et al. 2005, Moretti et al. 2006). Severe fires also create pulses of insect productivity, resulting in a large number of benthic, emerging aquatic, as well as terrestrial insects (Lacki et al. 2009, Malison and Baxter 2010a, Buchalski et al. 2013, Kral et al. 2017). Wildfires can promote algal growth due to higher water temperatures in burned areas (from reduced canopy cover) and increased amounts of nutrients such as nitrogen and phosphorus (Spencer and Hauer 1991, Minshall et al.

1997, Malison and Baxter 2010a). The increase in algae may then promote greater numbers of primary consumers, especially disturbance-adapted species which have short generation times, produce many offspring, and have high dispersal rates (e.g. Chironomidae) (Minshall 2003, Malison and Baxter 2010b). The increase in primary consumers is often associated with a corresponding increase in the insect Orders that prey on them (Malison and Baxter 2010b, Vieira et al. 2004). Fires also promote plant growth, providing habitat to insects that rely on the foliage, pollen and nectar of plants (Swengel 2001, Perry 2012). Bats capitalize on post-fire insect pulses (Lacki et al. 2009, Malison and Baxter 2010a) and adapt their foraging behaviours and habitat use to the timing of insect outbreaks (Brigham et al. 1992, McCracken et al. 2012, Kalda et al. 2015).

Many species of bats are generalist predators, consuming a wide variety of species of insects (Barclay and Brigham 1994, Carter et al. 2003, Kunz et al. 2011). Lepidoptera, Coleoptera and Diptera are some of the most important Orders of insects consumed by bat species in Alberta (Brigham and Saunders 1990, Reimer et al. 2010, Maucieri and Barclay 2021). Studies examining the effect of fire on insects have yielded differing results depending on the natural history of the insect species and the characteristics of the fire (e.g. size, severity, season) (Perry 2012). Although fire can increase insect productivity (Malison and Baxter 2010a, Buchalski et al. 2013), fires also consume insect habitat (i.e. ground vegetation, duff layer and coarse woody debris; Langor 2019), and cause insect mortality (Kral et al. 2017). Lepidopterans typically decline and coleopterans and dipterans generally increase after fires (Lacki et al. 2009, Armitage and Ober 2012, Kral et al. 2017, Tormanen and Garrie 2021). Even though certain Orders appear to be positively affected by fire, it is unclear to what extent bats prey upon and select specific insect species that become available after a fire (Perry 2012). Bats' foraging behaviours also change if the availability of roosts becomes limiting on a landscape (Verboom and Huitema 1997).

Fire has both positive and negative effects on roosting opportunities for bats. Cavity roosting bats roost in damaged or diseased trees that contain natural hollows, small cavities, and longitudinal splits, as well as under loose bark (Lacki and Schwierjohann 2001, Ford et al. 2006, Perry and Thill 2008). Foliage roosting bats roost within the foliage of tree branches

(Nagorsen and Brigham 1993). Fire can enhance roosting habitat by: creating snags, increasing the occurrence of loose bark, cavities, and hollows, reducing structural clutter, and increasing the amount of solar radiation reaching roosts by creating gaps in the forest canopy (Johnson et al. 2009, Perry 2012, Buchalski et al. 2013, O’Keefe and Loeb 2017). However, fire can also consume or weaken trees otherwise suitable for roosting (Randall-Parker and Miller 2002, Jung 2020) and result in expanses of defoliated habitat that are not conducive for foliage-roosting bats (Johnson et al. 2009, Perry 2012, Blakey et al. 2019).

1.2 Study Area

1.2.1 Kenow wildfire

The Kenow wildfire occurred in Waterton Lakes National Park (WLNP) (12U 287451 5437358) in southwestern Alberta in September 2017 (Fig. 1.1), burning 38% (19,303 ha) of the park before being brought under control (Parks Canada 2021a). The wildfire, started by lightning, was driven by extremely dry, hot, and windy conditions (Parks Canada 2021a), resulting in a predominantly extreme burn-severity throughout the park. WLNP defines burn severity as the “amount of vegetation change between pre- and post-fire condition” and uses it “as a way of describing the amount of ecological change” (Parks Canada 2021b). Although the fire consumed much of the canopy cover of the forested areas, the Waterton townsite was largely protected (Parks Canada 2021a).

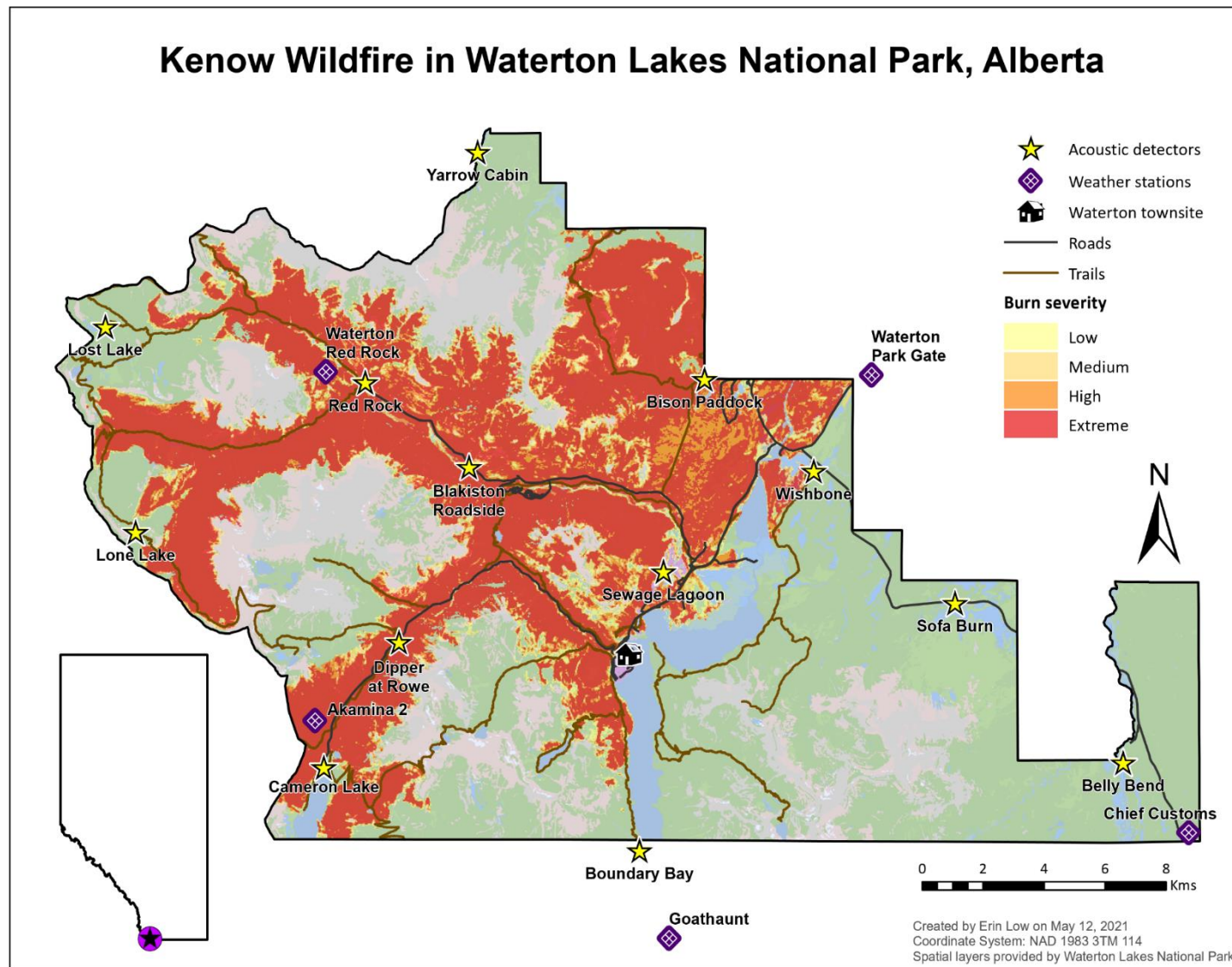


Fig. 1.1. The location of the study in Waterton Lakes National Park, Alberta. The area affected by the 2017 Kenow wildfire is displayed, as well as the corresponding burn severities (i.e. the amount of vegetation change between the pre- and post-fire condition). The locations of the thirteen acoustic detector sites monitored to assess bat activity changes pre- and post-wildfire are illustrated. The weather stations used to compile environmental data are also displayed. The map contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

1.2.2 Waterton Lakes National Park

WLNP is located in the southwest corner of Alberta and is characterized as the park “where the mountains meet the prairie” (Parks Canada 2010). WLNP is 505 km² in size and is located within four different subregions (Alpine, Subalpine, Montane, and Foothills Parkland) of the Rocky Mountain and Parkland Natural Regions (Parks Canada 2010, Alberta Parks 2015; Fig. 1.2). The elevation ranges from 1,280 m at the Waterton townsite to 2,910 m at the summit of Mount Blakiston (Parks Canada 2021c). The vegetation communities include grasslands (e.g. fescue species [*Festuca* spp.], bluebunch wheatgrass [*Pseudoroegneria spicata*]), shrublands (e.g. bearberry [*Arctostaphylos uva-ursi*], wolf willow [*Elaeagnus commutata*], common juniper [*Juniperus communis*]), wetlands (e.g. grey alder [*Alnus incana*], dwarf birch [*Betula nana*], Drummond’s willow [*Salix drummondiana*]) as well as coniferous (e.g. subalpine fir [*Abies lasiocarpa*], Engelmann spruce [*Picea engelmannii*], lodgepole pine [*Pinus contorta*], Douglas fir [*Pseudotsuga menziesii*]) and deciduous (e.g. paper birch [*Betula papyrifera*], poplar spp. [*Populus* spp.]) forests (Parks Canada 2010). A large diversity of vertebrates and invertebrates are found within these vegetation communities (Parks Canada 2010).

WLNP receives large amounts of both precipitation (mainly in the winter in the form of snow), and wind (Parks Canada 2018, ACIS 2020). The distribution of precipitation is variable across the park with areas on the west side (mountains) receiving a mean of 152 cm/year, the townsite (centrally located) receiving 107 cm/year and the east side of the park (parkland) receiving 76 cm/year (Parks Canada 2018). The mean wind speed in WLNP is 30 km/hr (Parks Canada 2018), however, it is variable with stronger winds in the winter months and in the parkland areas (ACIS 2020). Summers are short with occasional hot spells (> 30°C) (Parks Canada 2018, ACIS 2020). Winters are long with mild temperatures (due to chinooks) and occasional cold snaps (< -30°C) (Parks Canada 2018, ACIS 2020).

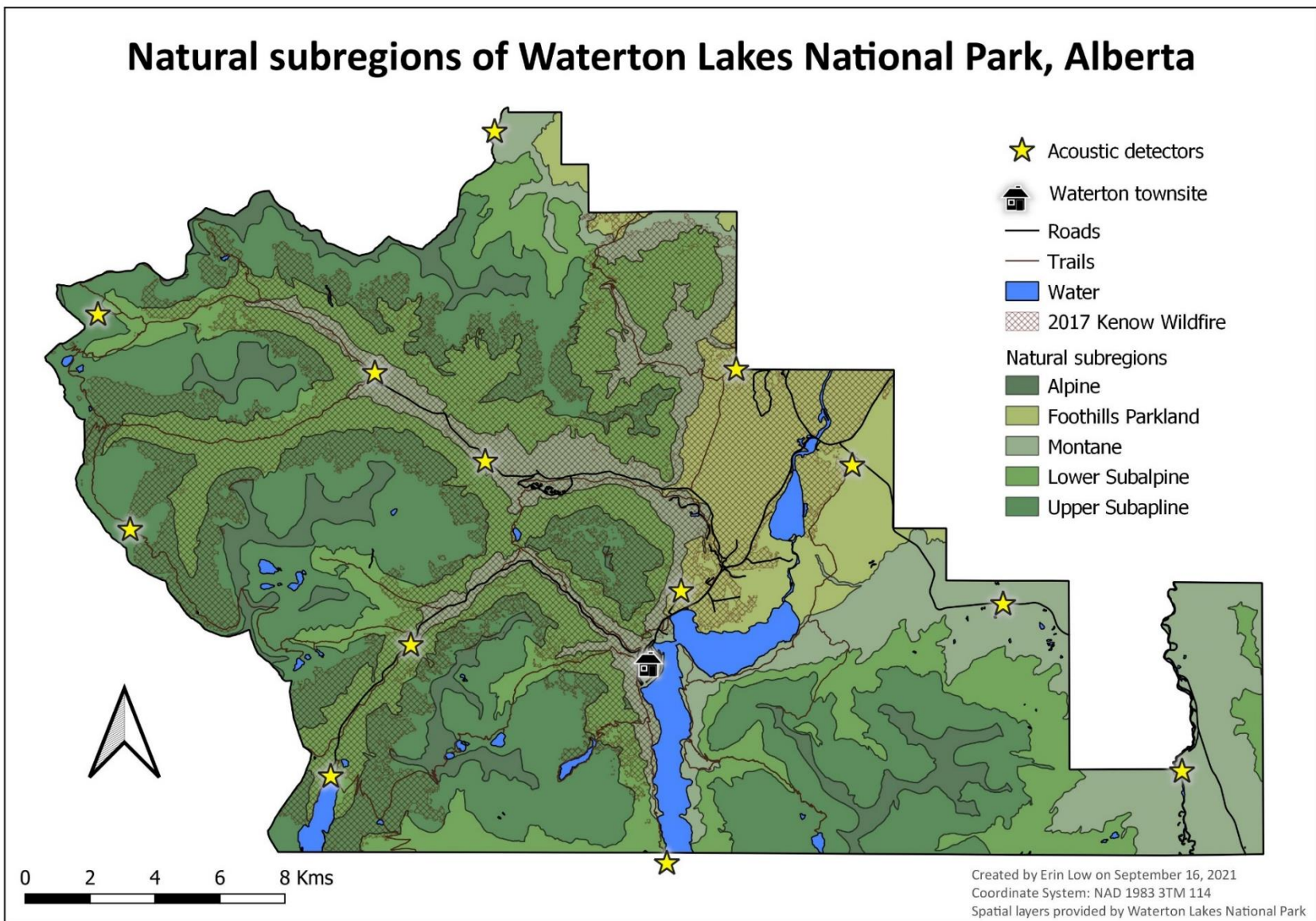


Fig. 1.2. The different natural subregions found throughout Waterton Lakes National Park, Alberta. The map contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

WLNP has a diverse fire regime due to its unique climate, terrain, and vegetation (Barrett 1996). The fire history of WLNP have been documented since the 1700s, primarily through sampling fire scars and fire-initiated age classes (Barrett 1996). Fire frequency increased significantly between the mid 1800s and early 1900s. However, active fire suppression efforts led to a decline after 1940 (Barrett 1996). Stand replacement regimes were most common prior to 1940, although mixed-severity fires in the grassland and valley-edge areas were also frequent (Barrett 1996).

In the 1900s lightning-caused fires were infrequent, with human-caused fires (from European settlers and Indigenous communities) being common (Barrett 1996). A similar pattern is thought to be true prior the 1900s (Barrett 1996). Since 1990 there have been three significant wildfires (i.e. defined as greater than 100 ha). These include the Sofa Mountain Wildfire in 1998 (881 ha), Kenow Wildfire in 2017 (19,303 ha), and the Boundary Wildfire in 2018 (272 ha); they were all caused by lightning (GOC 2019). Although there has been a history of fire suppression in the park, Parks Canada has been setting prescribed fires since the 1990s (GOC 2019). Since 1990 there have been over 30 prescribed fires, ranging in size from less than one ha to close to 1000 ha (GOC 2019). Most of these fires occurred in the Foothills Parkland subregion, although some fires were set in the Montane and Subalpine subregions (GOC 2019). The historic fire distribution and frequency are illustrated in Appendix A.

In addition to the Kenow wildfire, other fires (both wild and prescribed) occurred in WLNP between 2015 and 2020. A prescribed fire occurred within 50 meters (i.e. the predicted maximum distance bats may be detected by an ultrasonic microphone) of the acoustic detector “Wishbone” site in the spring of 2015. Although the fire may have influenced the activity levels of bats in the area, all years of my study would have been affected. Prescribed fires occurred within 50 meters of the acoustic detector “Blakiston Roadside” and “Red Rock” acoustic sites in the spring of 2016, sites that also burned in the 2017 Kenow wildfire. Although the prescribed fires may have influenced the activity levels of bats in the area, any potential variation in activity levels would likely have been appropriately captured by including three years of monitoring prior to the Kenow wildfire. For the remaining prescribed and wildfires that occurred during the study period, all acoustic detector sites were located more than 50 meters

from the burn. The complete list of fires that occurred in WLNP from 2015 to 2020 is provided in Appendix A.

Although WLNP actively suppressed fires from the 1940s to the 1990s (Barrett 1996), a mountain pine beetle (MPB; *Dendroctonus ponderosae*) outbreak occurred in WLNP from 1977 to 1987 (Axelson et al. 2018). The MPB is a species of bark beetle (Coleoptera, Curculionidae) native to Alberta (Dempster and Meredith 2021). MPBs bore into the bark of trees, primarily lodgepole pine (Axelson et al. 2018), to reproduce and lay eggs, thereby introducing fungal pathogens and ultimately killing the tree (Koch et al. 2021). Similar to fire events, MPB outbreaks are a natural disturbance that creates both spatial and temporal forest heterogeneity and complexity (Axelson et al. 2018). The MPB outbreak in WLNP caused a shift in the forest community structure from a lodgepole-pine dominated overstory prior to the outbreak, to a mixed-conifer type with increases in non-pine conifer and broadleaf species (Axelson et al. 2018). Although the MPB is still found in WLNP, the species currently exists in low population levels (Parks Canada 2017b). The distribution of the MPB in WLNP from 1977 to 2016 is illustrated in Appendix B.

1.3 Studying Bats

Bats are difficult to study due to their nocturnality, elusiveness and other behavioural characteristics (Russo and Voigt 2016). However, monitoring bat populations has become increasingly important to understand how they are changing due to anthropogenic pressures (Jones et al. 2009). Researchers have studied bats using a variety of techniques including capture methods (e.g. mist nets, harp traps and hand nets) and acoustic surveys (Zamora-Gutierrez et al. 2021). Bat capture surveys provide information that can only be determined through physically handling bats (Vonhof 2006). This includes positive species identifications, sex, age, reproductive condition, mass and other morphometric data (Anthony 1988, Racey 1988). However, capture surveys are biased towards bat groups that are easier to catch, involve a significant disturbance to the animal, and are only feasible in certain habitat types (e.g. over small water bodies, forest trails) (Vonhof 2006, Zamora-Gutierrez et al. 2021).

Acoustic surveys use detectors with high frequency microphones capable of recording bat echolocation passes (Zamora-Gutierrez et al. 2021). A bat echolocation pass is a series of

calls (individual echolocation 'clicks'), recorded as the bat flies near the detector's microphone (Fenton 1970). Acoustic surveys are not as invasive and typically have less biases compared to capture surveys (Zamora-Gutierrez et al. 2021). Ultrasonic detectors have been becoming increasingly affordable, with reasonable battery life and data storage capacities, making it possible to deploy multiple detectors over a large area for extended periods of time (Hill et al. 2019). However, definitive species identification is often impossible for many recorded passes as there is much overlap in echolocation frequency and pattern amongst different bat species (Barclay 1999, Walters et al. 2013). Bat echolocation passes vary based on whether the bat is searching, approaching, or attacking prey (Griffin et al. 1960), whether the bat is flying in an open or cluttered habitat (Findlay and Barclay 2020), and whether other bats are nearby (Obrist 1995). There is also regional variation among different populations, suggesting that call libraries and the resulting auto-identification software should be specific to a certain geographic area and habitat (Barclay 1999). Overall, despite some of the limitations, acoustic surveys have provided a cost-effective and consistent method to collect long-term data on bat populations (Zamora-Gutierrez et al. 2021).

1.4 Thesis Overview

In this thesis, I examine the effects of the 2017 Kenow wildfire on the bat community in WLNP, Alberta. As the wildfire occurred at the end of summer (August 30 to October 3, 2017), there was likely no direct bat mortality. Previous research has shown that bats can detect and respond to smoke even when in deep torpor (Doty et al. 2018, Geiser et al. 2018). This suggests that any roosting bats in the path of the fire would have been able to leave in time, including the bats born that summer. Pups are typically volant and independent by September in Alberta (Schowalter 1980). Bats in Alberta disperse from the summering grounds beginning in mid July, migrating to the areas they will spend the winter (i.e. hibernation sites or southern locations) (Cryan 2003, Baerwald and Barclay 2011, Olson and Flach 2016). As the fire occurred at the end of the active season, it is likely that many of the summer residents had left and any migrants in the area would be passing through quickly. Therefore, any changes to bat diversity and abundance can likely be attributed to the wildfire's impact on the environment (i.e. changes in

the forest structure, foraging opportunities, and roosting availability) and not direct bat mortality.

In 2019 and 2020, I radio-tracked female and male little brown *Myotis* (*Myotis lucifugus*) to their daytime roost sites. My field season in 2020 was negatively impacted by COVID-19. Due to the COVID-19 restrictions I had a very short field season (i.e. approximately three weeks rather than the three months I had planned) which limited the sample sizes of radio-tracked bats and the associated data collected. In addition, there was concern from the North American bat community about reverse zoonosis (i.e. SARS-CoV-2 could be transmitted from an infected person to a bat) (Runge et al. 2020), therefore minimal handling was recommended (CWHC 2021). In Chapter 2, I discuss my results regarding roosting behaviour and roost availability for little brown *Myotis* in a post-fire landscape. From 2015 to 2020 acoustic detectors were deployed at thirteen different sites throughout WLNP. Bat activity was recorded in areas that burned or did not burn, as well as in pre- and post-fire years. In Chapter 3, I examine the effects of the wildfire on the different bat species/species groups found in WLNP by comparing the activity levels (i.e. echolocation passes/night) between the burned and unburned areas, as well as from pre- and post-fire years.

Chapter 2: Little Brown *Myotis* Roosting Behaviour after the Kenow Wildfire

2.1 Introduction

2.1.1 Fire and bats

Fire is an important natural disturbance shaping forest communities (Waldrop et al. 1992, Brose et al. 2001, Krawchuk et al. 2020), creating a mosaic of habitat patches and altering species composition and forest structure (Dale et al. 2001, Johnstone et al. 2016). Although fire promotes landscape heterogeneity (Armitage and Ober 2012) and maintains suitable habitat for wildlife (Pausas and Parr 2018), it is unknown what the effects of rapidly changing fire regimes will be on forest biodiversity (Fontaine and Kennedy 2012, Blakey et al. 2019), including the bat community. The response by bats to wildfire is complex due to a variety of interactions between the fire regime (e.g. severity, frequency, season), regional climate, habitat type, and the individual species' biology (Perry 2012, Law et al. 2018, Steel et al. 2019). Overall, however, the effect of wildfire on bats is understudied and poorly understood (Perry 2012, Buchalski et al. 2013, Law and Blakey 2021).

Fire (both prescribed and wildfire) generally has an overall neutral or positive effect on most bat species (e.g. Austin et al. 2020, Ancillotto et al. 2021, Blakey et al. 2021, Taillie et al. 2021). However, species-specific negative responses have also been reported (e.g. Jung 2020, Starbuck et al. 2020, Ancillotto et al. 2021). Fire impacts insectivorous bat communities by changing the foraging habitat, insect communities (i.e. prey), and roosting opportunities (e.g. Buchalski et al. 2013, Lacki et al. 2017). Fires promote reduced vegetative complexity in the midstory and understory, and lower tree densities and more-open canopies, conditions that are conducive for less maneuverable bat species (Boyles and Aubrey 2006, Perry 2012, Jung 2020). Past studies have suggested that severe fires create pulses of insect productivity (e.g. Malison and Baxter 2010a, Buchalski et al. 2013), with bats adapting their foraging behaviours and habitats to the timing of insect outbreaks (Brigham et al. 1992, McCracken et al. 2012, Kalda et al. 2015). Fire has both positive and negative impacts on roost availability. Fire enhances roosting habitat (e.g. creates snags, increases occurrence of loose bark, reduces structural clutter), while also reducing roosting availability (e.g. consumes potential roost trees) (Perry 2012, Buchalski et al. 2013, O'Keefe and Loeb 2017, Jung 2020).

2.1.2 Hypothesis and predictions

Prior to my field seasons, limited bat research had been conducted in WLNP. The roosting behaviour of little brown *Myotis* and roost availability had never been examined in WLNP prior to the wildfire. As such, there were large knowledge gaps about the general roosting ecology of little brown *Myotis* in WLNP, even before a high severity wildfire occurred. Based on information from Chapter 1, I hypothesized that the Kenow wildfire affected the species' distribution and abundance of bats in Waterton Lakes National Park (WLNP) due to changes in the forest structure, foraging opportunities, and roost availability. Although fire can consume or weaken roost trees (Randall-Parker and Miller 2002, Jung 2020), fire also enhances roosting habitat by creating snags, increasing the number of potential roost structures (e.g. loose bark, hollows), and improving roost site characteristics (e.g. reduced structural clutter, increased solar radiation) (Johnson et al. 2009, Perry 2012, Buchalski et al. 2013, O'Keefe and Loeb 2017). Specifically examining roosting availability for little brown *Myotis* (*Myotis lucifugus*), I hypothesized that although roost trees were destroyed by the fire, there would have been an overall increase in the number of potential roost trees post-fire. I predicted burned areas would have more confirmed and potential roost trees compared to unburned areas.

2.2 Methods

2.2.1 Study species

The little brown *Myotis* is a medium-sized *Myotis* (mass 6.2 – 10.2 g, forearm length 33.0 – 40.3 mm; Nagorsen and Brigham 1993) and is arguably the most common species in Alberta. Currently, the little brown *Myotis* is provincially listed as “May Be at Risk” (AEP 2020). However, it has been recommended by Alberta's Endangered Species Conservation Committee to be listed as “Endangered” under the Alberta Wildlife Act (L. Wilkinson, pers. comm.). It is federally listed under the Species at Risk Act as “Endangered” (GOC 2021) due to mortality associated with white-nose syndrome. Little brown *Myotis* hibernate during the winter, with reproductive females forming maternity colonies in spring to raise their offspring in (Fenton and Barclay 1980). Males and non-reproductive females roost alone or in small groups in spring and summer (Fenton and Barclay 1980). Little brown *Myotis* roost in buildings, bat houses,

trees (e.g. under bark, in crevices and cavities) and rock crevices (Fenton and Barclay 1980, Crampton and Barclay 1998, Psyllakis and Brigham 2006, Randall et al. 2014). Maternity colonies are regularly found in anthropogenic features such as in the attics of buildings, under roofs, and in bat houses (Fenton and Barclay 1980, Johnson et al. 2019). Pregnant and lactating females experience a high energy demand (Kunz et al. 1995), and therefore choose warm roosting locations that they share with other reproductive females to minimize thermoregulation costs (Barclay 1991). Males typically use natural roosts (Randall et al. 2014, Johnson et al. 2019).

Both males and females switch roost sites throughout the summer (e.g. Lacki et al. 2009, Johnson et al. 2010, Randall et al. 2014). Roost switching is thought to be used by bats as a way to meet their thermoregulatory needs, find optimal roosting microenvironments, as well as reduce predation risks, parasite loads, and distances to foraging areas (Kunz and Lumsden 2003, Johnson et al. 2009). Roost fidelity varies depending on roost type and energetic demands, with roost switching ranging from nightly to every few weeks (Vonhof and Barclay 1996, Brigham et al. 1997b). Roost tree availability is often assessed by examining roost switching frequency and distance between successive roost trees (Sedgeley and O'Donnell 1999, Kunz and Lumsden 2003, Chaverri et al. 2007, Johnson et al. 2009). High roost switching frequency and short distances traveled between successive roost trees is typically attributed to greater roost availability (Sedgeley and O'Donnell 1999, Chaverri et al. 2007, Johnson et al. 2009).

Little brown *Myotis* fly variable distances between roosting and foraging sites. They typically forage within one to three kms of their roost sites (Anthony and Kunz 1977, Crampton and Barclay 1998, Henry et al. 2002), although distances greater than five kms have also been reported (Randall et al. 2014). The distance will vary depending on the bat's sex, age, and reproductive condition (Lacki et al. 2007). Little brown *Myotis* forage over water, along forest edges, and in openings, feeding on a variety of insect orders including Diptera, Ephemeroptera, Trichoptera and Lepidoptera (Nagorsen and Brigham 1993, Clare et al. 2011, 2014a, Nelson and Gillam 2017). They are opportunistic predators, consuming large amounts of aquatic insects (Belwood and Fenton 1976, Saunders and Barclay 1992), with their diet reflective of both

seasonal and geographical insect availability (Clare et al. 2011, 2014a). They primarily consume their prey mid-flight (i.e. aerial hawking) (Burles et al. 2008).

2.2.2 Bat capture and radio-tracking

I captured bats in WLNP from June to August 2019, and July and August 2020. I set up mist nets in natural fly-ways, such as along trails, openings in the forest, and over slow-moving water bodies. The areas I chose for netting locations were based on selecting sites in a variety of habitats with different characteristics (e.g. burn/unburn locations, different subregions and forest types, varying elevations and distances to the Waterton townsite). The primary purpose of conducting the capture surveys was to catch bats to radio-track to roosting sites. The netting effort (e.g. number/size/height of nets set up, length of time nets were open) among the different captures sites was not identical, and as such, comparing capture rates between the different habitats is not appropriate. As I was targeting little brown *Myotis*, a species regularly found foraging around water (Mackey and Barclay 1988), most capture sites were located at water features (e.g. lakes, wetlands, creeks). I primarily netted at sites with roads nearby (i.e. within 150 m) so as to the limit how far the capture equipment needed to be carried. I also focused on netting at the previous bat capture sites used in 2011/2012 (i.e. Lausen 2012), as well as at the acoustic detector deployment sites. The sites were selected randomly in relation to date.

For each bat that I caught, I determined species, age, sex, and reproductive condition, and measured forearm and mass (Vonhof 2006). Juvenile bats can be distinguished from adult bats by examining the cartilaginous epiphyseal growth-plates of the phalanges (Brunet-Rossinni and Wilkinson 2009). Reproductive condition in females was categorized as: non-reproductive (small furred nipples and no fetus detected by palpation of the abdomen), pregnant (fetus detected by palpation of the abdomen), lactating (nipples enlarged and milk can be expressed), and post-lactating (nipples enlarged and unfurred but no milk can be expressed) (Racey 2009). Reproductive females included females that were pregnant, lactating, or post-lactating. All females were considered not obviously pregnant if they did not show signs of reproduction (i.e. pregnant or lactating) until July 31st, after which time they were considered non-reproductive. Male reproductive condition was categorized as: non-scrotal (testes not descended and/or not

swollen), early scrotal (testes partially descended and/or swollen), or scrotal (testes have descended and/or are swollen) (Haarsma 2008, Racey 2009). I held bats for one hour to collect a guano sample and ensure an accurate mass (Vonhof 2006).

I attached radio-transmitters (LB-2X, LB-2N, and BD-2N, 0.31 g, 0.35 g and 0.44 g; Holohil Systems, Carp, Ontario) to adult female (both reproductive and non-reproductive) and male little brown *Myotis* and tracked them to their roosts during the day (Fig. 2.1). I trimmed fur from between the scapulae and applied a small amount of surgical adhesive (Osto-Bond, Montreal Ostomy, Quebec, Canada) to attach the transmitter. Once the transmitter was attached, I held the bat for ten minutes to allow the glue to dry, and then released it. All transmitters were between four and six % of the bats' body mass to ensure that the transmitter did not interfere with their foraging behaviours (Aldridge and Brigham 1988).

I radio-tracked bats to their day roosts to examine the characteristics of the selected roosts and assess roost switching behaviour (Fig. 2.1). I used hand-held receivers (R-1000 Telemetry Receiver, Communication Specialists, Inc., Orange, California, USA and R410 Scanning Receiver, Advanced Telemetry Systems, Isanti, Minnesota, USA) with a roof-mount omni-directional antenna, as well as three- and five-element Yagi antennae to track the radio-tagged bats. My research was conducted under the following approvals and permits: #AC17-0094 from the Life and Environmental Sciences Animal Care Committee of the University of Calgary, and #WL-2019-32138 from the Parks Canada Agency (Research and Collection Permit, and SARA-Compliant Authorization).



Fig. 2.1. A little brown *Myotis* with a radio-transmitter attached (left; M. Blair), and radio-tracking the bat in a burned area in Waterton Lakes National Park, Alberta (right; J. Theoret).

2.2.3 Roost identification

In 2019, I radio-tracked primarily reproductive females to their maternity colonies, and in 2020 I tracked predominantly males to their roosts. Non-reproductive (i.e. not pregnant, lactating or post-lactating) and/or not obviously pregnant (i.e. caught early in the season and could not determine if pregnant) females were also tracked each year. For bats tracked to building roosts, few measurements were collected due to the buildings being privately owned. For building roosts, I confirmed the building and recorded the coordinates with a GPS (Garmin GPSMAP 64ST, Olathe, Kansas) determined the exit point(s), and counted the total number of bats that exited at night (i.e. conducted emergence surveys). I also checked the building roosts daily to examine any roost switching behaviour.

For bats tracked to either trees or rock roosts, I confirmed the roost by looking for defects in the suspected roost structure to determine if the tree/rock was a potential roosting structure (e.g. trees that had sloughing bark, cracks, cavities, or broken tops, and rocks with cracks of an appropriate size). I then visually inspected the potential roost to determine if the bat and/or antenna was visible, and conducted emergence surveys (i.e. watched the bat leave

at night) to confirm the exact tree and exit location when possible. Bats were tracked for multiple days to examine roost switching behaviour. I conducted vegetation surveys at tree roost sites, recording roost characteristics including the tree species, tree height and diameter at breast height (dbh), decomposition stage (Cline et al. 1980, Kaupas 2016), percent bark remaining on the tree, and the roost entrance direction. For rock roosts, I recorded the roost entrance direction. I measured the dbh of trees using a diameter tape (Forestry Suppliers, Jackson, Mississippi), and tree height with a clinometer (Suunto PM-5, Vantaa, Finland).

2.2.4 Roost tree availability surveys

To determine if there was a difference in roost-tree availability between burned and unburned areas, I conducted vegetation surveys. I defined a potential roost tree using criteria from other studies examining potential bat roost trees (e.g. Vonhof and Barclay 1996, Brigham et al. 1997b, Crampton and Barclay 1998, Kaupas 2016). Potential roost trees include trees that are dying or dead, have a defect that a bat could access (e.g. sloughing bark, crack, cavity, broken top), and have a dbh > 10.0 cm. However, due to field measurements being taken incorrectly, a dbh > 6.3 cm threshold was used in 2019. Although this is smaller than previous studies have used, the same diameter threshold was used between burned and unburned areas. As such, comparing the availability of potential roost trees was still appropriate. In addition, in 2020, only sloughing bark was considered to be a defect that a bat could access (i.e. cracks, cavities, and broken tops were not included). This was due to my findings from radio-tracking bats showing that sloughing bark was the only feature type used by the bats roosting in trees in WLNP.

Different methods were used to conduct the availability surveys in 2019 and 2020. In 2019 I surveyed trees using a variation of the point-quarter method (Krebs 1989) where for each sampling point, I divided the area into four 90° quadrants and measured the nearest tree (dbh > 6.3 cm) in each quadrant. I surveyed a linear transect of five points that were random distances between 20 and 50 m apart, sampling a total of 20 trees per transect. I recorded the number of trees surveyed that could be potential roost trees. The starting point of the survey was selected to be representative of the targeted habitat (i.e. burned/unburned deciduous, coniferous or mixed forests). As WLNP is divided into distinct forest types, I went to the general

area of the targeted habitat (e.g. burned deciduous, unburned mixed) and from the road looked for a section of forest that appeared to be large enough to conduct the linear transect. The survey bearing direction was generated randomly. In 2020 potential roost-tree availability was assessed at the locations I tracked the tree-roosting bats to. I examined the 0.1 hectare area (17.8 m radius) surrounding the confirmed roost tree and counted the number of trees (dbh > 10.0 cm) and the number of potential roost trees (i.e. dying/dead tree with sloughing bark and a dbh > 10.0 cm).

2.2.5 Statistical analysis

To determine if little brown *Myotis* were associated with different site characteristics depending on their sex or reproductive state, I conducted Kruskal-Wallis rank sum tests (functions `shapiro.test()` and `kruskal.test()` in package *stats*, R Core Team 2020) in RStudio (version 1.3.1093, R Core Team 2020). I used the capture site's elevation and distance to the Waterton townsite to examine differences in site selection among reproductive females, non-reproductive females and males. Juveniles and females that were not obviously pregnant or were post-lactating, were excluded from the analyses. Not obviously pregnant females were excluded due to the ambiguity of their reproductive state. Post-lactating females were excluded due to uncertainties of whether the bats were residents of WLNP, or individuals migrating through the park after rearing young elsewhere. It was also unknown if they had successfully raised a pup, or had lost/aborted their young earlier in the summer. As such, the behaviours of the post-lactating females were likely affected depending on the above listed circumstances. The Euclidean distance from the capture site to the Waterton townsite was calculated in QGIS (version 3.16.8-Hannover) using the vector analysis Distance Matrix tool. The elevation of capture sites was determined using Google Earth Pro, version 7.3.3.7786.

Due to the small sample sizes, I primarily used descriptive statistics (e.g. mean, standard error, range) to assess roost selection for the radio-tracked bats. The Euclidean distances from capture sites to roost sites were calculated in QGIS (version 3.16.8-Hannover) using the vector analysis Distance Matrix tool. To determine if there was a difference in the number of potential roost trees (i.e. number of potential bark-roosts/number of trees surveyed) between burned and unburned areas, I conducted Wilcoxon rank sum tests with continuity corrections (function

hist() in package *graphics*, and function wilcox.test() in package *stats*, R Core Team 2020) as there were small sample sizes and non-normal data distributions. I also conducted a Wilcoxon rank sum test with continuity corrections to determine if there was a difference in the percent bark remaining on roost trees between burned and unburned areas.

2.3 Results

2.3.1 2019/2020 captures

During my field seasons in 2019 and 2020, I caught a total of 364 bats over 41 nights of netting, at 16 different capture sites. Most sites were located within 150 m of a road/the vehicle (13/16 sites). However, three sites were further distances away (two sites within 1.5 km of a road), with the furthest site being nine kms from the nearest road. There was a much larger netting effort in 2019 (325 bats over 37 nights at 15 different sites) compared to 2020 (39 bats over four nights at four different sites). Little brown *Myotis* made up the majority of captures (90.7%, n=330), followed by silver-haired bats (*Lasionycteris noctivagans*, 2.7%, n=10) and hoary bats (*Lasiurus cinereus*, 2.7%, n=10) as the next most commonly captured species. Long-eared *Myotis* (*Myotis evotis*, 1.4%, n=5), long-legged *Myotis* (*Myotis volans*, 0.5%, n =2), and big brown bats (*Eptesicus fuscus*, 0.3%, n=1) were captured infrequently. There was also a small number of *Myotis* (1.7%, n=6) that escaped before a species identification could be determined.

My capture results yielded similar results to past capture surveys conducted in WLNP (i.e. Lausen 2012). In July and August 2011 and 2012, 81.2% of total captures were little brown *Myotis* (Lausen 2012), compared to 90.7% in my 2019/2020 data. Hoary bats were the next most common capture in 2011/2012 (8.1%; versus 2.7% in 2019/2020), followed by eastern red bats (*Lasiurus borealis*, 3.7%; versus 0% in 2019/2020), with the remaining species (long-legged *Myotis*, long-eared *Myotis*, silver-haired bat, big brown bat) making up 7.0% (versus 4.9% in 2019/2020) of the remaining captures (Lausen 2012). Interestingly, I did not catch any eastern red bats, despite netting at a number of the same sites and at similar times of year as Lausen (2012). Although Lausen's (2012) species' capture rates were different than mine, it does not necessarily reflect changes in the bat community from 2011/2012 to 2019/2020. I was specifically targeting little brown *Myotis* for my study, while Lausen (2012) was conducting a bat species inventory throughout the park.

Little brown *Myotis* displayed geographical sexual segregation between reproductive (i.e. pregnant or lactating) females (n=29), non-reproductive females (n=32) and males (n=185). The remaining little brown *Myotis* excluded from the analyses were juveniles (n=17), not obviously pregnant females (n=36), post-lactating females (n=28), or individuals that escaped before their age/sex/reproductive condition could be assessed (n=3). There was a significant difference in the elevation of the capture site (Kruskal-Wallis chi-squared= 62.686, df= 2, $P < 0.001$), with males foraging at higher elevations (1503 ± 13.8 SE m) compared to reproductive females (1323 ± 17.9 SE m) and non-reproductive females (1330 ± 19.7 SE m). There was no significant difference in the Euclidean distance of the capture site to the Waterton townsite (Kruskal-Wallis chi-squared= 1.164, df= 2, $P = 0.559$) for the different groups. The mean Euclidean distance for reproductive females was 7.6 km (SE ± 0.2), non-reproductive females was 7.4 km (SE ± 0.3), and males was 8.4 km (SE ± 0.3).

2.3.2 Female little brown *Myotis*

Between 2019 and 2020, 12 female little brown *Myotis* were radio-tagged. Four were not obviously pregnant, one was pregnant, six were lactating and one was non-reproductive. Of the 12, 11 were successfully tracked to either a confirmed roost or general roosting and/or foraging area. One female (not obviously pregnant but likely non-reproductive) was only ever detected late at night, foraging over Cameron Lake. The locations of the capture sites and roosting locations of the females tracked in WLNP in 2019 and 2020 are illustrated in Fig. 2.2.

Seven females were tracked to maternity colonies in seven different buildings in the Waterton townsite. These colonies ranged in size from 52 to 200 individuals (mean size of 99 ± 20.1 SE), with colony sizes likely indicative of either pre-volancy numbers or the start of volancy (i.e. pups were just beginning to fledge). Building roosts typically had one main exit, with bats emerging from gaps under the roof. Although no roost switching behaviour was observed for any females tracked to buildings, I strongly suspect most of the females roosting in the buildings were able to get their tags off very quickly. I checked the building roosts regularly at various times of the night to confirm if the tag was still attached (i.e. if the radio-tag was consistently in the building roost at night, the bat had likely managed to get it off). I suspect most tags were dropped within three days of tagging for bats roosting in buildings.

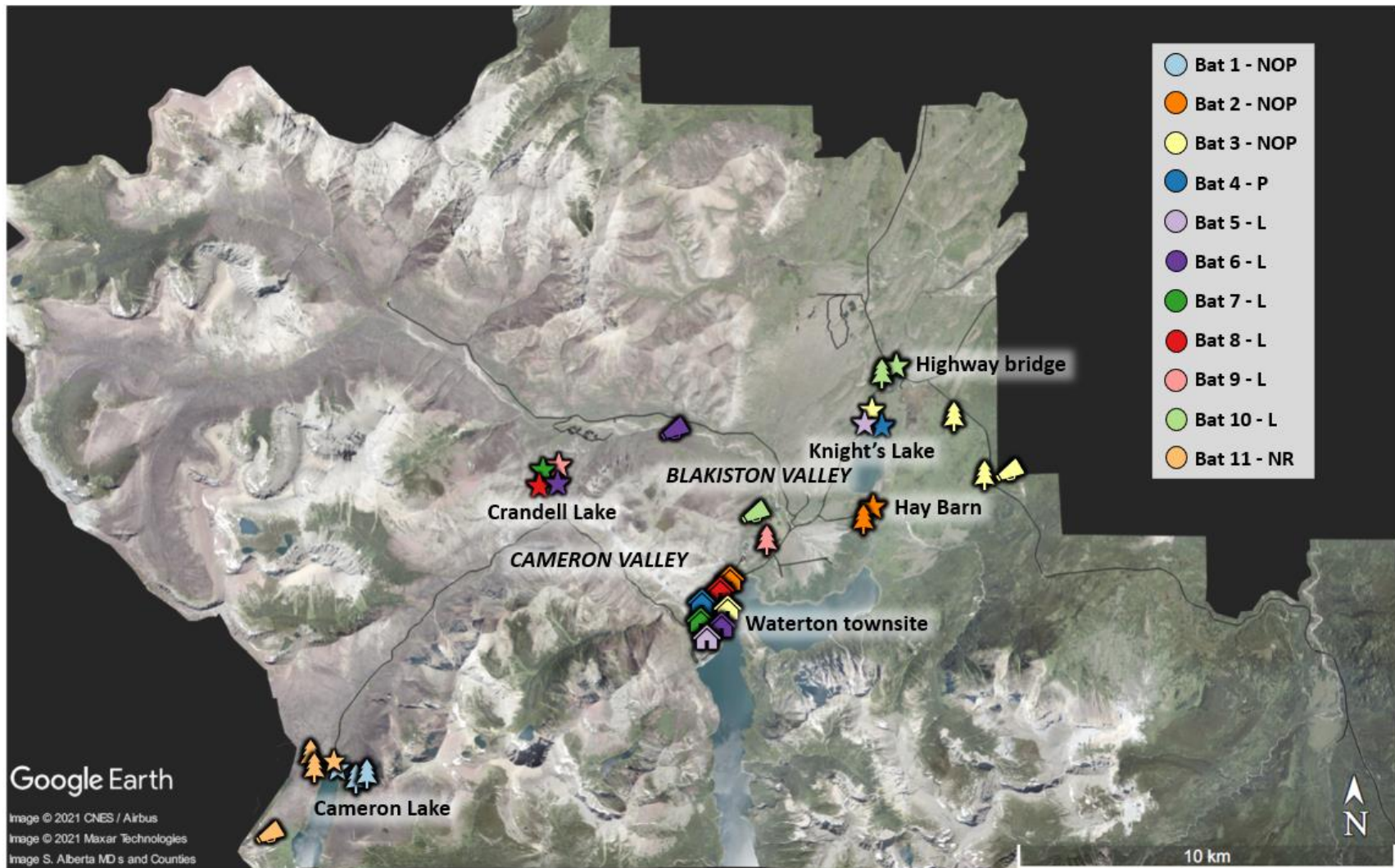


Fig. 2.2. The locations of the capture sites (star icons) and roosting locations of the female little brown *Myotis* tracked in Waterton Lakes National Park, Alberta in 2019 and 2020. The tree icons denote natural roosts (i.e. tree or rock roosts), the house icons denote building roosts and the megaphone icon denotes general areas where radio signals were detected. “NOP” refers to not obviously pregnant (i.e. caught early in the season and could not determine if pregnant), “P” refers to pregnant, “L” refers to lactating, and “NR” refers to females that were non-reproductive (i.e. not pregnant, lactating or post-lactating). The Cameron Valley and Blakiston Valley are shown, corresponding to the roads (i.e. black lines). Netting sites and the Waterton townsite are also labeled. The map layer was provided by Waterton Lakes National Park. The site and roost coordinates are listed in Appendix C.

One female (Bat 3), who was tagged early in the season (June 24, 2019) used tree roosts (likely balsam poplar [*Populus balsamifera*] or trembling aspen [*Populus tremuloides*], although no specific roost trees were confirmed), before switching to a building roost. Although she was caught too early in the season to determine her reproductive condition, I suspect she was reproductive and was originally using natural roosts before relocating to a maternity colony to raise her pup. Another female (Bat 2) who was caught too early in the season to determine her reproductive condition (July 16, 2020) was first tracked to a tree roost (a dying balsam poplar, dbh 42.7 cm and 18.2 m tall), before relocating the next night to a building. However, using a natural roost near to the netting site (i.e. within 200 m) the night she was caught may have been a result of stress associated with being handled.

Two lactating females (Bats 6 and 9) tagged at Crandell Lake likely dropped their tags while out foraging or night roosting in the Blakiston Valley. I believe that the tags were dropped due to radio signals appearing to come from the same location for more than one week. One of those females (Bat 6) was located roosting in a building in the Waterton townsite prior to dropping her tag. I was unable to locate the maternity roost of the other female (Bat 9) before she dropped her tag. One lactating female (Bat 10) caught later in the season (August 9, 2019) roosted in a tree (dying balsam poplar, dbh 11.8 cm and 35.0 m tall) near to the netting site the night she was captured (i.e. within 100 m), was then heard roosting in the Blakiston Valley two days later, and from the following day onwards, was never detected again. Four either pregnant or lactating females (Bats 4, 5, 7 and 8) were only ever detected in building roosts.

Two females (Bat 1 and 11), both likely non-reproductive, were tracked at Cameron Lake. The general roosting area was determined for Bat 1 (likely conifer tree roosts on the edge between burned and unburned forest), and confirmed roosts (bark roosts of burned conifer trees) were determined for Bat 11. Bat 11 roosted under the bark of a burned Engelmann spruce (*Picea engelmannii*; dbh 61.8 cm and 29.2 m tall) the night I tagged her, was not tracked the following day, and switched to a different bark roost of a burned subalpine fir (*Abies lasiocarpa*; dbh 23.2 cm and 22.4 m tall) which she used as a day roost for the following two days. Both tree roosts selected by Bat 11 were in early decomposition stages (i.e. very little decay, and most bark and branches were still present) as the trees had likely been killed in the

wildfire. The bark roosts were located on the southeast side of the trees. Although I was not able to confirm her roost location for any other days, based on the bearings I took of the radio-signal, she was roosting at a higher elevation at the far end of Cameron Lake.

Although I did not track the females throughout the night (i.e. track females from their foraging/capture sites to their diurnal roosts to determine the exact routes taken and thus maximum flight distance), locating an individual's foraging site and their diurnal roost informs roosting trends and minimum nightly flight distances. The Euclidean distance between Knight's Lake and the Waterton townsite is approximately seven kms. This suggests that the bats roosting in the townsite who were caught at Knight's Lake would have flown at least 14 kms in one night. The distance from Crandell Lake to the Waterton townsite is approximately six kms via the Cameron Valley (i.e. Euclidean distance) and approximately 10 to 12 kms (i.e. dependent if flying along the side of the mountain or following Blakiston Creek) via the Blakiston Valley. The female caught at Hay Barn was tracked to a building roost approximately 4.5 kms away, suggesting a nightly flight Euclidean distance of at least nine kms. For the confirmed/likely non-reproductive females caught at Cameron Lake, they were tracked to roosting locations much closer to their capture site. The female whose roost trees were confirmed, was found roosting 0.4 km away from where she was tagged. However, she would have been roosting further from her capture site during the days when she was not tracked and only the signal bearing was determined (likely roosting within two to three kms of her capture site). Although a confirmed roost tree was never located for the other female, the general roosting areas located were 0.7 km away from where she was tagged. The metadata for the above capture sites, bats tagged, and roost specifics are provided in Appendix C.

*2.3.3 Male little brown *Myotis**

In 2020, 15 male little brown *Myotis* were radio-tagged; no males were tracked in 2019. In total, I located 37 roosts or roosting areas for twelve of the fifteen males I radio-tagged. Three males disappeared before I could confirm roost sites. The locations of the unburned and burned capture sites and roosting locations of the males tracked in WLNP in 2020 are illustrated in Fig. 2.3 and Fig. 2.4, respectively.

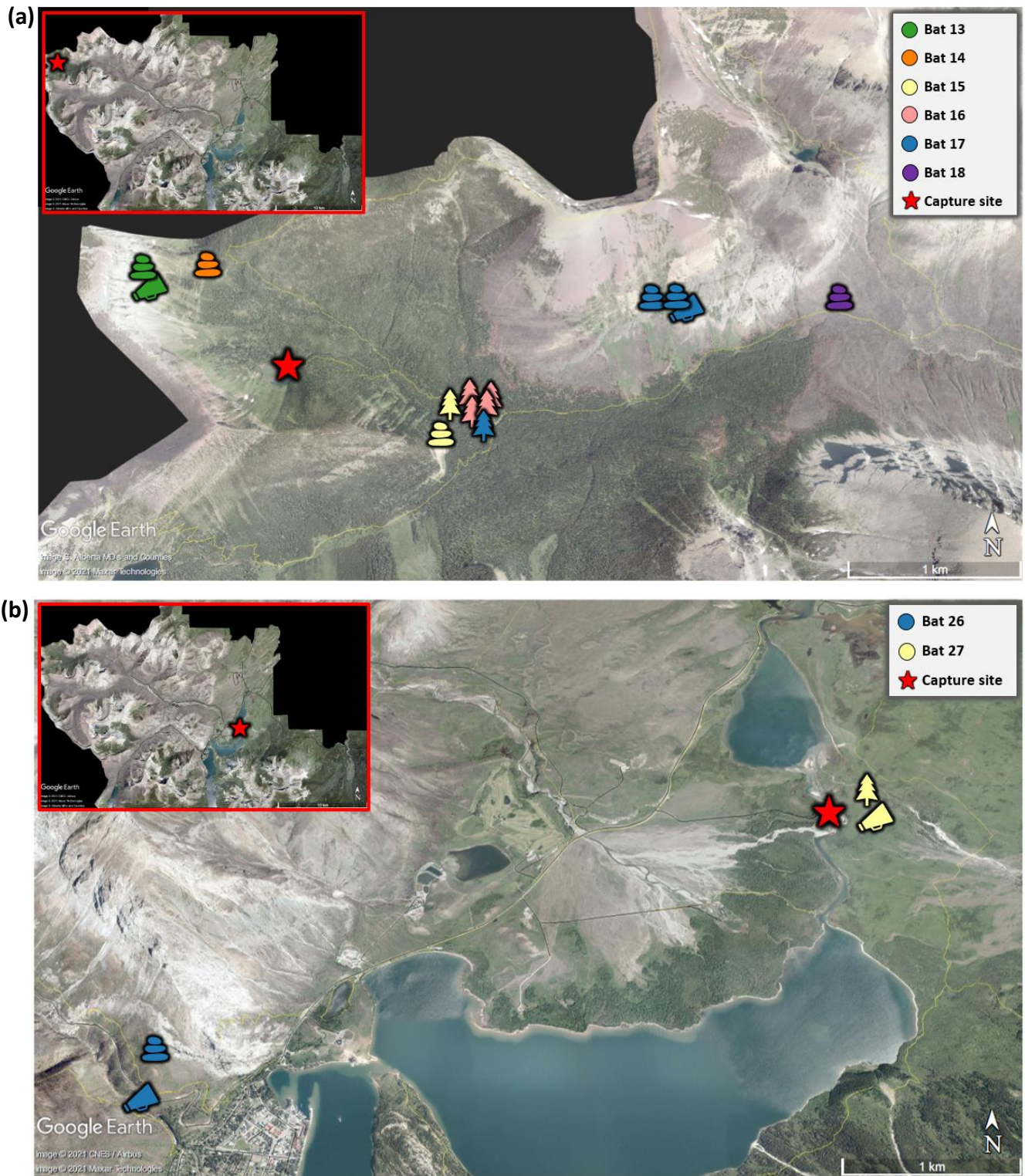


Fig. 2.3. The locations of the unburned capture sites (star icon) and roosting locations of the male little brown *Myotis* tracked in Waterton Lakes National Park, Alberta in 2020. The Lost Lake area is illustrated in (a) and the Hay Barn area is displayed in (b). In this and subsequent figures, the tree icons denote tree roosts, the rock pile icons denote rock crevices or boulder roosts, and the megaphone icon denotes general areas where radio signals were detected. The map layer was provided by Waterton Lakes National Park. The site and roost coordinates are listed in Appendix C.

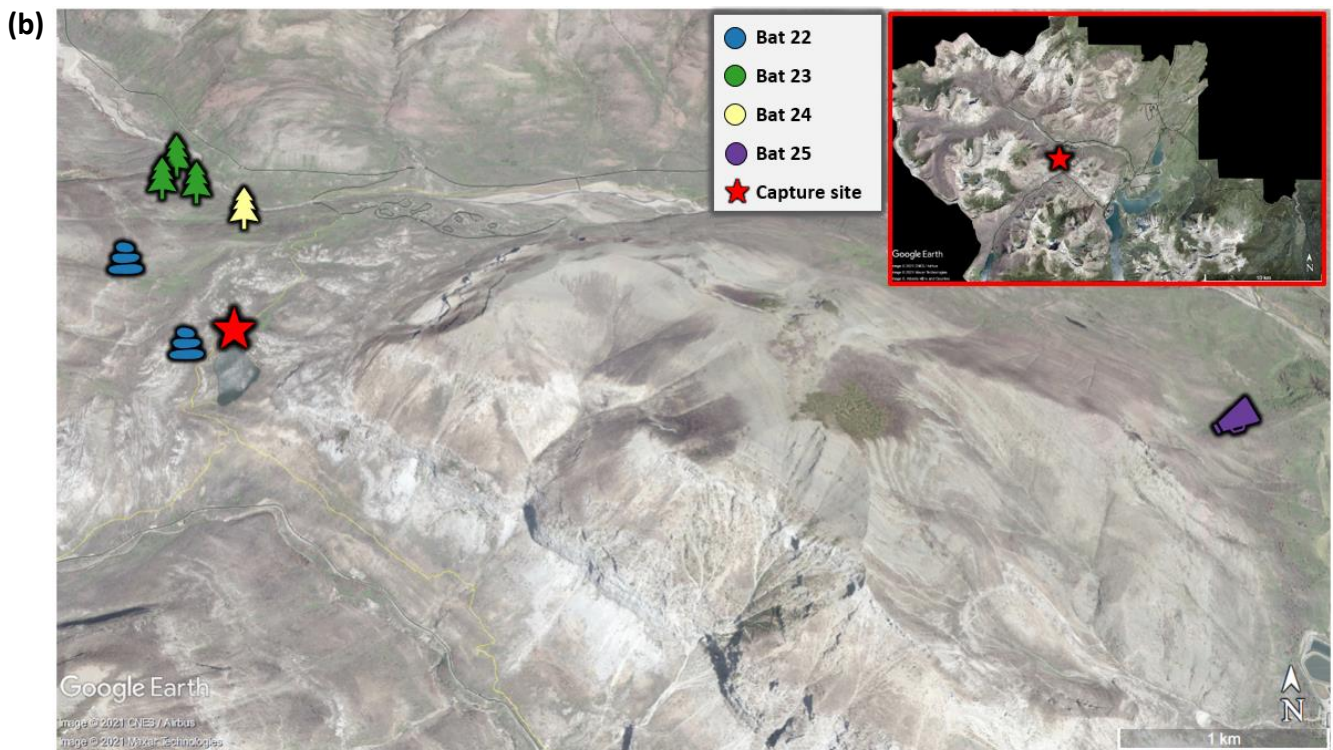
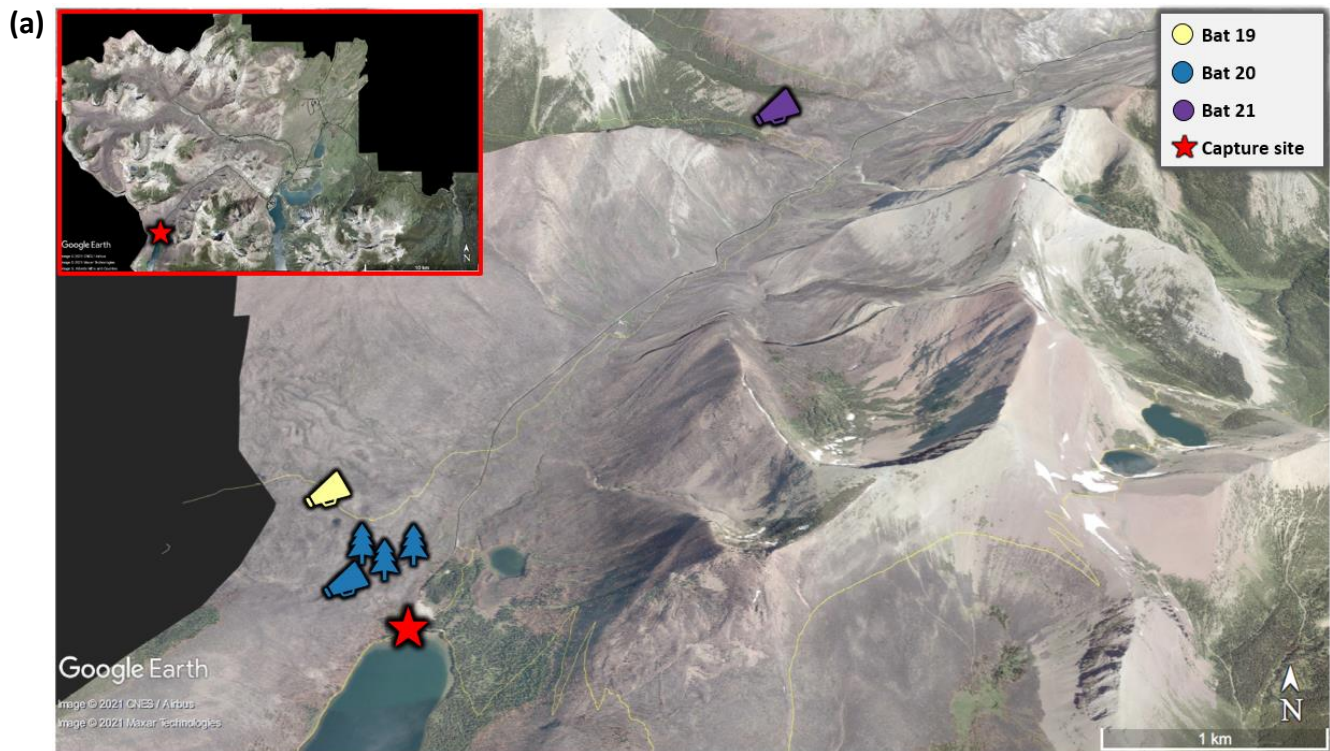


Fig. 2.4. The locations of the burned capture sites (star icon) and roosting locations of the male little brown *Myotis* tracked in Waterton Lakes National Park, Alberta in 2020. The Cameron Lake area is illustrated in (a) and the Crandell Lake area is displayed in (b). The map layer was provided by Waterton Lakes National Park. The site and roost coordinates are listed in Appendix C.

The males tracked in 2020 roosted in trees (n=22), in rock crevices along the side of mountains (n=12), or in cracks in boulders (n=3). Examples of the natural roosts selected by male little brown *Myotis* in WLNP in 2020 are illustrated in Fig. 2.5. Male bats were only ever observed roosting alone (n=13). Bats tracked to trees were either confirmed or suspected to be using bark roosts only. Confirmed bark roosts (i.e. where a bat was either observed during the day or was seen emerging at night) were primarily on the southwest to southeast side of the tree (n=10). The located rock roosts (i.e. rock crevices and boulders) were predominantly south to east facing (13 out of 15).

Male bats were tracked for between one and five days (mean 3.1 ± 0.5 SE). I located 19 roosts in burned areas (from seven individual bats); 52.6% in trees, 31.6% in rock crevices and 15.8% in boulders. I located 18 roosts in unburned areas (from six individual bats); 66.7% in trees and 33.3% in rock crevices. The radio-tracked bats showed preferences for specific roost types, with only two out of the twelve tracked individuals ever switching between trees and rocks. The Euclidean distance from where bats were captured to where they roosted ranged from 0.3 to 5.6 km (n=33), with a mean of 1.6 ± 0.2 SE km. The bat roosting in the boulder traveled greater distances compared to the mean distances for tree and rock crevice roosting bats (Fig. 2.6). All males caught at the burned sites (i.e. Cameron Lake or Crandell Lake) were found roosting in burned areas (n=20 roosts). Males predominately roosted in unburned areas when caught at unburned sites (i.e. Hay Barn and Lost Lake; n=28 roosts). However, 28.6% (n=8) of located roosts were found in burned areas as well. Males switched roost sites on average every 1.8 ± 0.2 SE days (includes eight different individuals who were tracked to confirmed roosts for \geq two days). Males roosting in the burned areas switched roosts on average every 1.7 ± 0.5 SE days (n=4), with males in the unburned areas switching on average every 1.9 ± 0.4 SE days (n=3). There was also one male who switched from an unburned tree (day one) to rock crevices in a burned area (days two to four) and then back to an unburned tree (day five).

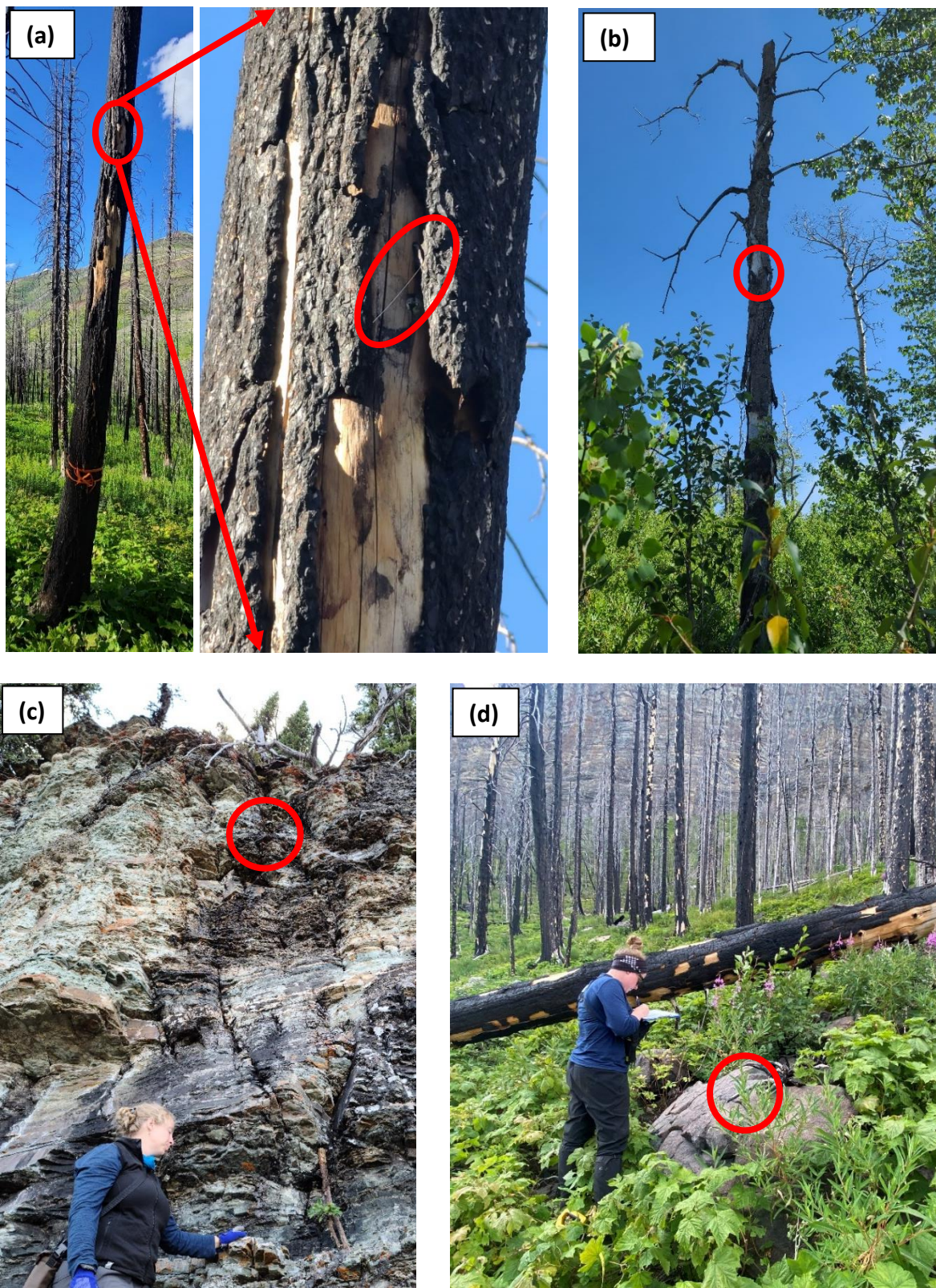


Fig. 2.5. Natural roosts selected by male little brown *Myotis* in Waterton Lakes National Park, Alberta in 2020. Included are examples of burned (a) and unburned (b) bark roosts, as well as a rock crevice (c) and boulder roost (d; K. Low). The red circles denote where on the roost structure the bats were located. The antenna of the bat in (a) can be seen sticking out from under the bark.

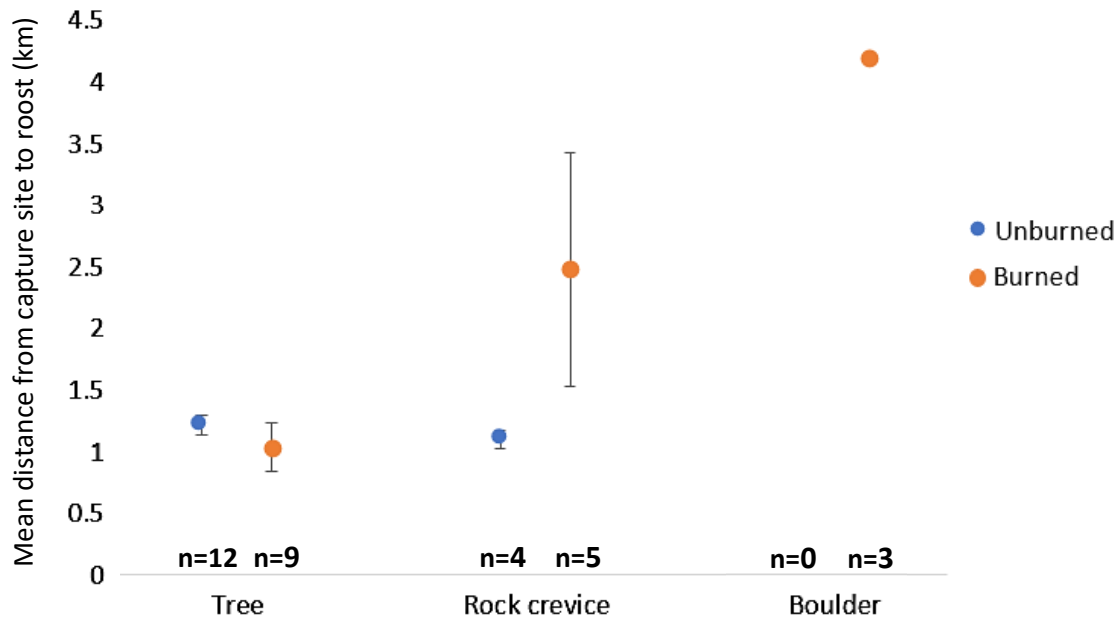


Fig. 2.6. The mean (\pm SE) Euclidean distance male little brown *Myotis* traveled between capture sites and roosts (km) for trees, rock crevices and boulders in the unburned and burned areas in Waterton Lakes National Park, Alberta in 2020.

Males that roosted in trees selected a number of different tree species including: subalpine fir, Engelmann spruce, lodgepole pine (*Pinus contorta*), and trembling aspen. The species of tree selected for roosting was always one of the predominant species in the area. The dbh of roost trees ranged from 12.6 to 49.4 cm ($n=16$), with a mean of 34.8 ± 2.5 SE cm. Roost trees ranged in height from 6.9 to 28.8 m ($n=16$), with a mean of 21.0 ± 1.7 SE m. The trees selected as roosts were always dead, with the trees in the unburned areas typically being more decayed compared to the burned trees. Unburned roost trees ($n=7$) typically had few branches, the bark was either loose or lost, and the top was often broken. Burned roost trees ($n=9$) were likely alive prior to the wildfire and were in earlier stages of decomposition (i.e. very little decay and many branches and twigs were still present). The percent bark remaining on roost trees ranged from 5 to 95% ($n=16$), with a mean of $66.6\% \pm 6.4$ SE. There was no significant difference in the percent bark remaining on roost trees between burned ($n=9$, mean 60.0 ± 10.9 SE %) and unburned areas ($n=7$, mean 75.0 ± 2.7 SE %; $W=22.5$, $P=0.359$).

2.3.4 Tree roost availability surveys

Although I recorded all potential roost types in 2019 (e.g. sloughing bark, crack, cavity, broken top), I only included sloughing bark for the potential roost availability analysis, as tree-roosting bats were only ever located under bark throughout this study. As different methods were used in assessing bark roost availability in 2019 and 2020, I examined the potential bark-roost availability (i.e. number of potential bark-roosts/number of trees surveyed) in the burned versus unburned areas in 2019 separately from 2020, as well as in 2019/2020 combined. In 2019, the proportion of potential bark-roosts ranged from 0 to 0.4 (mean 0.2 ± 0.07 SE, $n=5$ sites) in the burned sites, and from 0 to 0.2 (mean 0.1 ± 0.04 SE, $n=5$ sites) in the unburned sites. In 2020 the proportion of potential bark-roosts ranged from 0 to 0.3 (mean 0.2 ± 0.02 SE, $n=10$ sites) in the burned sites, and from 0 to 0.3 (mean 0.2 ± 0.04 SE, $n=7$ sites) in the unburned sites. There was no significant difference in the number of potential bark roosts between the burned and unburned areas in 2019 ($W= 16.0$, $P = 0.511$), 2020 ($W= 26.0$, $P = 0.405$), or 2019/2020 ($W= 93.0$, $P = 0.903$). The locations of the vegetation surveys conducted in 2019 and 2020 are illustrated in Fig. 2.7.

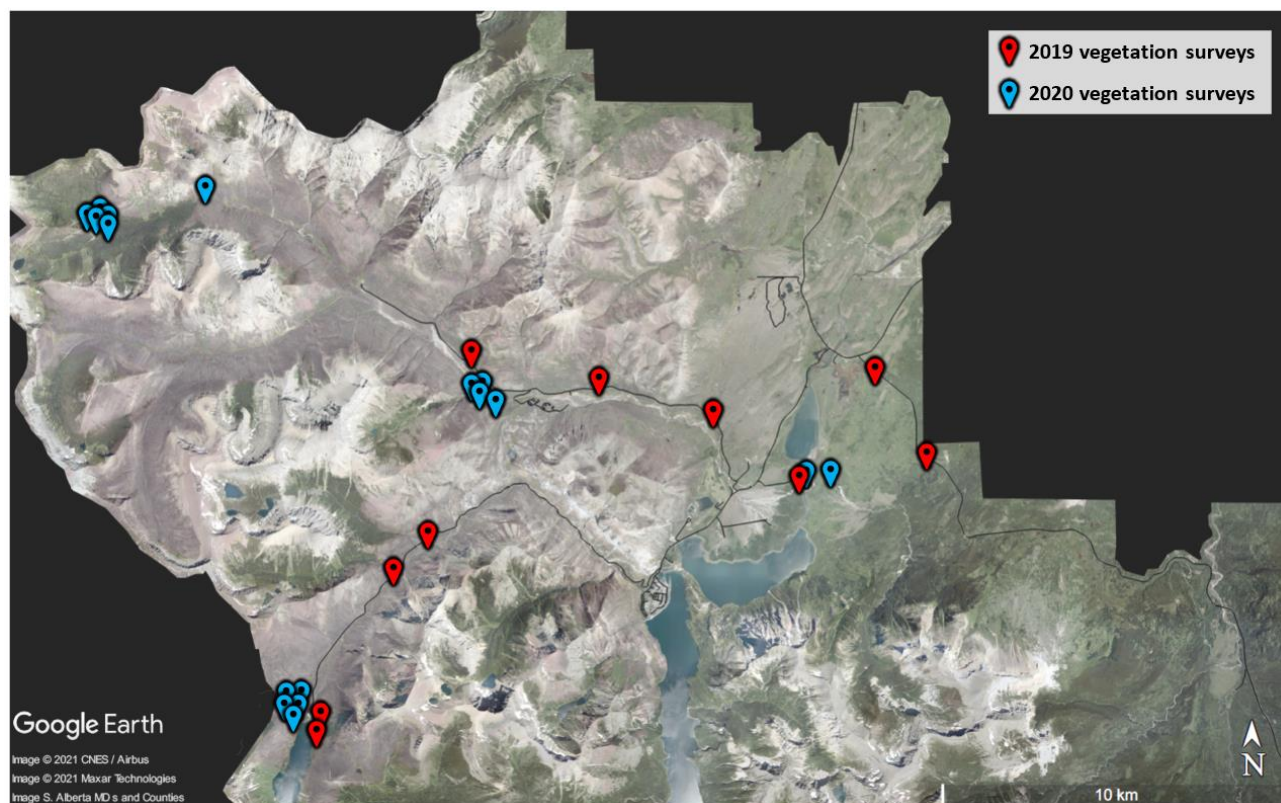


Fig. 2.7. The locations of the 2019 and 2020 vegetation surveys completed in Waterton Lakes National Park, Alberta. The vegetation surveys assessed the number of potential tree roosts (i.e. trees with sloughing bark) in each survey area. The map layer was provided by Waterton Lakes National Park.

2.4 Discussion

2.4.1 2019/2020 captures

From my capture data, I determined that female (reproductive and non-reproductive) little brown *Myotis* foraged in habitats at lower elevations compared to males. This was not surprising due to the prevalence of maternity colonies in the buildings in the Waterton townsite. The Waterton townsite is situated at approximately the same elevation as my low elevation capture sites and is surrounded by a network of lakes and watercourses. Thus, the reproductive females have appropriate roosting habitat (i.e. buildings) and suitable foraging opportunities (i.e. riparian areas) to meet the high energetic requirements of raising a pup. Although non-reproductive females do not have the same energy requirements as pregnant and lactating females, it is possible that some of the captured non-reproductive females were

still associated with the maternity colonies. Past research on other bat species has suggested that non-reproductive females may roost with reproductive females for the purpose of social thermoregulation (Ransome 1973, Kerth and König 1999). It is also possible that non-reproductive females chose to roost near to the maternity colonies to maintain social relationships with members of the colony (Fukui et al. 2010), with sociality perhaps being related to kin selection (Eberhard 1975, Kerth et al. 2002).

Sex-specific variation in geographic distribution patterns has been observed in previous studies (e.g. Barclay 1991, Grindal et al. 1999, Cryan et al. 2000, Saunders 2015). Past research has suggested that reproductive females are likely limited to areas with lower thermoregulatory costs and energy requirements (Barclay 1991, Grindal et al. 1999, Cryan et al. 2000). Males are likely able to inhabit areas with increased energy demands as they are able to use torpor under adverse conditions such as in cold weather and when prey resources are limited (Barclay 1991). Conversely, any torpor use by reproductive females will slow the development of the pups and reduce the pups' overwintering survival rates (Racey 1973, Racey and Swift 1981, Grindal et al. 1992). Reproductive females' energy demand nearly doubles from early lactation to peak lactation (Kurta et al. 1989), emphasizing the importance of inhabiting areas with reduced energetic costs.

As all maternity colonies located were in the Waterton townsite and males roosted near to where they were captured, I examined whether there was a difference in the mean distance from the capture site to the townsite between males and females. I found there was no significant difference in distance for females versus males, which suggests that females are likely traveling further distances in a night compared to males. My results from radio-tracking males and females also support this. Most of the reproductive females tracked were caught at least six kms away from their maternity colonies in the Waterton townsite, whereas most males roosted within 1.5 km of where they were caught. In Yukon, female little brown *Myotis* also travelled further between their roost sites and foraging areas compared to males (Randall et al. 2014).

As reproductive females in WLNP displayed a strong fidelity to building roosts in the townsite, it is possible that they are traveling long distances nightly due to the network of lakes

and watercourses surrounding the townsite. This water network may provide flyways with reduced energetic demands, coupled with abundant foraging opportunities. It is also likely that due to the concentration of female little brown *Myotis* in the Waterton townsite, each individual needs to have a larger home range due to intraspecific competition for food. Although some males were also caught near the Waterton townsite, more were associated with higher elevation sites, presumably areas with reduced competition for food sources. My results from radio-tracking males in 2020 suggest that natural roosts are not limited at these higher elevation sites.

2.4.2 Female little brown *Myotis*

Despite the limited sample size of the radio-tracked females, I determined that buildings are critical habitat for reproductive female little brown *Myotis* in WLNP. I also found that non-reproductive females behaved more similarly to males compared to reproductive females, choosing natural roosts rather than buildings. I tracked seven females to seven different maternity colonies in buildings in the Waterton townsite. Past studies have also found that buildings are important habitat for maternity colonies of little brown *Myotis* (e.g. Davis and Hitchcock 1965, Burnett and August 1981, Henry et al. 2002), especially in high elevation areas (Johnson et al. 2019). Buildings are often selected as roosting structures as they provide stable warm microclimates, allow a larger number of individuals to roost together, thus reducing energy costs of thermoregulation, have low predation risks, and are more permanent compared to tree roosts (Lewis 1995, Lausen and Barclay 2006, Willis and Brigham 2007, Johnson et al. 2019).

Although the behaviour was only observed in one bat, using tree roosts early in the season prior to relocating to buildings (presumably to give birth) could be a way to minimize energy expenditure during periods of poor weather (e.g. cold spring days). Males and non-reproductive females are regularly associated with tree and rock roosts (e.g. Jung et al. 2004, Psyllakis and Brigham 2006, Randall et al. 2014, Shively and Barboza 2017) as natural roosts are cooler compared to buildings and thus more suitable for using torpor (Barclay and Kurta 2007, Johnson et al. 2019). Although using torpor saves energy for the female, it delays fetal development and milk synthesis (Racey and Swift 1981, Wilde et al. 1999). Late parturition

dates reduce juvenile overwintering survival rates (Frick et al. 2010). This is especially true in high elevation locations that have late springs and early falls and thus a shortened time to acquire the fat reserves required for hibernation (Johnson et al. 2017). It is likely that buildings have enabled reproductive females to successfully raise offspring in locations such as WLNP that would otherwise not be possible if natural roosts were the only roosting option (Johnson et al. 2019). It has been suggested that female little brown *Myotis*' frequent use of buildings may be due to their preference for roosting structures that can accommodate large colony sizes and last for relatively long periods of time (Bergeson et al. 2015). Buildings can support large colonies for many years and have become increasingly available since European colonization when anthropogenic structures started to replace trees (Bergeson et al. 2015).

Due to the locations of the tags that I believe were dropped by the reproductive females while foraging or night roosting (Bats 6 and 9), I suspect accessing Crandell Lake via the Blakiston Valley (rather than Cameron Valley) is more energy efficient and/or has greater insect availability. This suggests that some reproductive females are likely flying more than 20 kms per night. All females radio-tagged at Crandell Lake (n=4) were lactating, an energetically demanding reproductive state that also requires frequent returns to the maternity roost to nurse the pups (Kurta et al. 1989, Henry et al. 2002). Although I was unable to assess foraging distance differences between pregnant (n=1) and lactating (n=6) females due to my small sample sizes, a past study found that lactating females had smaller home ranges compared to pregnant females (Henry et al. 2002). As the energetic demand for a lactating female to commute large distances between a foraging site and their maternity roost is high, it suggests that roost sites and/or foraging areas may be a limiting resource in WLNP. Alternatively, it could be that the benefit of roosting in a building outweighs the cost of foraging at sites that are far away.

2.4.3 Male little brown Myotis

Contrary to my prediction that burned areas would have more confirmed roost trees compared to unburned areas, I found that males did not preferentially select either burned or unburned trees, but rather used the habitat that was in close proximity to them. All males caught at the burned sites roosted in burned areas, and most males caught at unburned sites

roosted in unburned areas (20 out of 28; 71.4%). Bark roosts were selected more often compared to either rock crevices or boulders, in both burned and unburned areas. However, the radio-tracked bats appeared to have individual preferences for specific roost types, with only two out of the twelve tracked bats ever switching between trees and rocks.

Although my sample size was small, roost switching frequency was similar in burned (mean 1.7 ± 0.5 SE days, $n=4$) and unburned (mean 1.9 ± 0.4 days, $n=3$) areas. The mean distances traveled from capture sites to roosts were also similar for tree roosts between the burned (mean 1.0 ± 0.2 SE kms, $n=9$) and unburned (mean 1.2 ± 0.1 SE kms, $n=12$) areas. For rock crevice roosts, bats traveled further in the burned areas (mean 2.5 ± 0.9 SE kms, $n=5$) compared to the unburned areas (mean 1.1 ± 0.1 SE kms, $n=4$). However, as rock crevice availability was likely not affected by the fire, this could be a result of the small sample size.

Past studies examining roost selection by male bats in temperate areas in post-fire landscapes, have yielded differing results. Similar to my results, roost switching frequency and distances moved between roost trees by male *Myotis sodalis* was similar between burned and unburned areas (Johnson et al. 2010). Another study found that male *M. sodalis* selected burned trees more often than unburned trees for roosting in one study area, but used both burned and unburned trees equally in two other study areas (Perry et al. 2015). Other studies have found that roosts in burned areas are selected by male (Boyles and Aubrey 2006, Torrey 2018) and female bats (Lacki et al. 2009) compared to roosts in unburned areas. The differences observed in roost selection in post-fire studies may be due to a variety of differences in the studies, including different study species, study sites, time of year, number of years since the fire, and fire severity. However, overall, the results highlight that fire likely has an overall neutral to positive effect on availability of roosting habitat.

Although my sample sizes are not large enough to elucidate why some males selected tree versus rock roosts, my findings highlight the individual-level variation among little brown *Myotis*. This variation includes differences in behavioural traits such as exploration, sociability, boldness, and aggressiveness (termed 'animal personality') (Toscano et al. 2016). Animal personality can influence an individual's spatial distribution and movement patterns within habitats (Nilsson et al. 2014, Schirmer et al. 2019), including their foraging and roosting

behaviour (e.g. site fidelity, home-range size) (Toscano et al. 2016). Individual-variation reduces intraspecific competition and may promote diversification and stability within a population (Bolnick et al. 2003).

From my tracking data in 2020, I found that only two out of the twelve male bats ever switched between tree and rock roosts, and one bat was only ever located in a boulder (i.e. was visually confirmed in the boulder on three consecutive days). Thus, male little brown *Myotis* in WLNP exhibit behavioural differences regarding roost selection. The boulder roosting bat especially highlighted this as he was also detected foraging at Lost Lake at night after locating him in the boulder that morning as well as the morning after. To roost in the boulder during the day and forage at Lost Lake at night, a distance of over four kms each way, he had to fly past both burned and unburned tree roosts as well as rock crevices. As flying is energetically costly, the rationale for choosing to repeatedly roost in a boulder over four kms away from a preferred foraging area is likely due to personality.

2.4.4 Tree roost availability surveys

As all bats tracked to tree roosts (both males and females) were either confirmed or suspected to be using bark roosts, I only considered sloughing bark as a potential roost in my analysis (i.e. did not include cracks, cavities, or broken tops). Contrary to my prediction that burned areas would have more potential roost trees compared to unburned areas, I found that there was no significant difference in bark roost availability between burned and unburned areas. However, this was for years two and three after the wildfire. I believe roost availability could change in future years, as bark roosts are temporary structures (Russo et al. 2005). I observed a bark roost that was occupied by a bat in July, and by August of the same year, the piece of bark the bat was roosting under appeared to have fallen off the tree. As WLNP receives a large amount of precipitation and wind (Parks Canada 2018, ACIS 2020), I expect that sloughing bark will continue to fall off trees at a relatively fast rate. In the burned areas, it will take decades for trees to regrow and become large enough to have bark roosting opportunities. This suggests that bark roosts may become limited in the burned areas in future years. However, I also observed a number of potential roosting cracks in trees (10% of 100 trees surveyed) in the burned areas in 2019. In addition to the existing cracks that are currently an

appropriate width for little brown *Myotis* to roost in, there were additional smaller cracks present that will likely widen over time. It is unknown whether little brown *Myotis* will switch to cracks in trees if sloughing bark becomes limited in the future.

Snags in severely burned areas are less likely to fall compared to trees in lightly burned areas (Angers et al. 2011). In addition, burned and unburned trees are lost at similar rates (Ford et al. 2021). However, this does not necessarily translate to sustained roost availability for bats if bark roosts are still actively selected for over cracks in the future. Although snags are still standing, the amount of bark remaining on dead trees decreases over time (Barclay and Brigham 2001). Overall, tree roost availability in WLNP did not appear to change in the burned areas immediately post-fire. Projecting to the intermediate-term trend, the loss of trees can be a gradual process in a post-fire landscape (Angers et al. 2011, Ford et al. 2021). The long-term effects of fire on roost availability are less known as it will depend on both the recruitment of new and loss of old roost trees (Ford et al. 2021).

2.4.5 Study limitations

In 2019 I faced challenges due to intermittent field support and in 2020 my field season was limited due to COVID-19. Sample sizes for the number of bats tracked and vegetation surveys completed were smaller than I had planned for both years. Radio-tracking bats in the mountains also raised challenges. An ongoing issue was the terrain causing both signal bounce (i.e. unreliable bearings due to the mountains reflecting the signal) or interference (i.e. mountains limiting the range of the tags). Some tagged bats were never detected and/or tracked to a roost, and as such, my results are from the easily tracked bats and potentially the ones with predictable behaviour. It is unknown if the bats that were never detected had issues with their radio-tag (e.g. tag stopped working, fell off), were roosting in areas with high interference (e.g. deep in rock crevices, behind mountains) or left the study area entirely. In addition, as often multiple bats were tracked and located each day, it was not possible to confirm exit points (i.e. watch the roost at emergence time) for each located roost. It is possible that some of the roosts were misidentified.

For the data analysis, roost characteristics and roosting behaviours were pooled even though multiple data points were associated with the same individual bat. Not all bats were

tracked for equal amounts of time, meaning that the calculated means are skewed in the direction of bats who were located more often. Any potential bias is likely exacerbated by the small size. However, my results do highlight roosting behaviours and overall trends. For the vegetation surveys, different methods were used in the two years and the sample sizes were limited for both. I only included sloughing bark as a potential roost type. However, it is possible that some bats also use cracks, cavities and broken tops of trees. I also included sloughing bark from any side of the tree. My results from confirmed roosts suggest that bats predominately use bark roosts on the south side of trees. However, methods were consistent between the burned and unburned areas for each year. Therefore, the overall conclusions about potential roost availability between burned and unburned areas are likely unbiased.

2.4.6 Future research

An interesting observation I found from tracking a female which was likely pregnant in 2019, was the use of natural roosts early in the season. Although she was the only individual tracked in June, my findings suggest that reproductive females may use natural roosts while pregnant. This may allow for greater use of torpor and thereby delay parturition until environmental conditions improve. Reproductive females could also be taking advantage of the increasing night length (i.e. longer foraging times) that occurs after summer solstice (Reimer 2013) or timing lactation to when there is an increase in insect availability (Henry et al. 2002). Additional females suspected or confirmed to be in early pregnancy should be tracked early in the year to test this hypothesis.

Another area for future research could be examining tree and bark persistence (e.g. Barclay and Brigham 2001) over the intermediate to long-term. If bark roosts become limiting in future years, male little brown *Myotis* may switch to cracks, cavities and broken tops of trees, roost in rock crevices more often, or be forced to roost more predominately in unburned areas. Understanding the long-term succession of trees in burned areas could help explain roosting behaviour of male little brown *Myotis* in WLNP.

2.4.7 Management recommendations

My results highlight the importance of building roosts to reproductive female little brown *Myotis* in WLNP. As each female tagged was tracked to a different building, it suggests

that there are more building roosts in the Waterton townsite than I located. To maintain the population of this endangered species in WLNP, building roosts should be protected. If there are human-wildlife concerns and building owners need to evict the bats, timing can be used to prevent unnecessary mortality of the mothers and pups. Bat exclusions should be planned for the late fall to early spring, when bats are hibernating and not in the buildings (Neilson and Fenton 1994, Brittingham and Williams 2000). Renovations to buildings with maternity colonies should also be completed from late fall to early spring.

My findings from tracking male bats highlight their adaptability and use of a variety of roosting structures in WLNP. As the study area is located within a national park and is therefore a protected area, I do not believe that male little brown *Myotis* require additional management. Although it is unknown what the long-term effect of the fire will be on tree roost availability in the burned areas, the unburned areas may offer sufficient roosting habitat. As long as forest succession continues and trees are able to mature, availability of roost trees should not become limiting.

2.5 Conclusion

Past studies have found that fire has both positive and negative effects on roosting opportunities for bats. Fire can enhance roosting habitat by creating snags and increasing the occurrence of potential roosts (Johnson et al. 2009, Perry 2012, Buchalski et al. 2013, O'Keefe and Loeb 2017). However, fire can also consume or weaken roost trees (Randall-Parker and Miller 2002, Jung 2020). Despite the small sample sizes, my results suggest that the Kenow wildfire did not change roosting availability for little brown *Myotis* in WLNP two to three years post-fire.

Reproductive females were strongly associated with buildings in the Waterton townsite, structures that were primarily unaffected by the wildfire. Some reproductive females likely traveled more than 20 kms per night, suggesting that high quality roost sites and/or foraging areas may be limited in WLNP. As female little brown *Myotis* appear to require buildings to successfully raise their young in WLNP, buildings with maternity roosts need to be protected to maintain the little brown *Myotis* population in the park. The radio-tracked males and non-reproductive females did not preferentially select either burned or unburned roosts, but rather

used the habitat that was close in proximity. Bats used a variety of natural roosts in WLNP including bark roosts, rock crevices and boulders in both burned and unburned areas. Natural roosts do not appear to be limiting in the burned areas as rock roosts were likely unaffected and the wildfire did not appear to change the availability of bark roosts. I believe that the variation in natural roost selection by little brown *Myotis* was primarily due to individual preferences and personality differences in their preferred roost type. The little brown *Myotis* in WLNP also displayed sexual segregation in foraging and roosting habitat with males associated with higher elevation areas compared to females. My findings highlight the importance of considering both males and females to make informed management decisions that are not biased towards one or the other. Overall, my results highlight the adaptability and opportunistic nature of little brown *Myotis*. Although roost availability did not increase post-fire, this species shows resilience and appears to have adapted to the significant ecological changes caused by the Kenow wildfire.

Chapter 3: Changes in Bat Activity after the Kenow Wildfire

3.1 Introduction

3.1.1 *Fire and bats*

Fire is an important natural disturbance shaping forest communities (Waldrop et al. 1992, Brose et al. 2001, Krawchuk et al. 2020), creating a mosaic of habitat patches and altering species composition and forest structure (Dale et al. 2001, Johnstone et al. 2016). Although fire promotes landscape heterogeneity (Armitage and Ober 2012) and maintains suitable habitat for wildlife (Pausas and Parr 2018), it is unknown what the effects of rapidly changing fire regimes will be on forest biodiversity (Fontaine and Kennedy 2012, Blakey et al. 2019), including the bat community. The response by bats to wildfire is complex due to a variety of interactions between the fire regime (e.g. severity, frequency, season), regional climate, habitat type, and the individual species' biology (Perry 2012, Law et al. 2018, Steel et al. 2019). Overall, however, the effect of wildfire on bats is understudied and poorly understood (Perry 2012, Buchalski et al. 2013, Law and Blakey 2021).

Fire (both prescribed and wildfire) generally has an overall neutral or positive effect on most bat species (e.g. Austin et al. 2020, Ancillotto et al. 2021, Blakey et al. 2021, Taillie et al. 2021). However, species-specific negative responses have also been reported (e.g. Jung 2020, Starbuck et al. 2020, Ancillotto et al. 2021). Fire impacts insectivorous bat communities by changing the foraging habitat, insect communities (i.e. prey), and roosting opportunities (e.g. Buchalski et al. 2013, Lacki et al. 2017). Fires promote reduced vegetative complexity in the midstory and understory, and lower tree densities and more-open canopies, conditions that are conducive for less maneuverable bat species (Boyles and Aubrey 2006, Perry 2012, Jung 2020). Past studies have suggested that severe fires create pulses of insect productivity (e.g. Malison and Baxter 2010a, Buchalski et al. 2013), with bats adapting their foraging behaviours and habitats to the timing of insect outbreaks (Brigham et al. 1992, McCracken et al. 2012, Kalda et al. 2015). Fire has both positive and negative impacts on roost availability. Fire enhances roosting habitat (e.g. creates snags, increases occurrence of loose bark, reduces structural clutter), while also reducing roosting availability (e.g. consumes potential roost trees) (Perry 2012, Buchalski et al. 2013, O'Keefe and Loeb 2017, Jung 2020).

3.1.2 Acoustically monitoring bats

Researchers have studied bats using a variety of techniques including capture methods (e.g. mist nets, harp traps and hand nets) and acoustic surveys (Zamora-Gutierrez et al. 2021). Bat capture surveys provide information that can only be determined through physically handling bats (Vonhof 2006). However, capture surveys are biased towards bat groups that are easier to catch, involve a significant disturbance to the animal, and are only feasible in certain habitat types (e.g. over small water bodies, forest trails) (Vonhof 2006, Zamora-Gutierrez et al. 2021). Acoustic surveys are not as invasive and typically have less biases compared to capture surveys (Zamora-Gutierrez et al. 2021). Acoustic surveys involve using detectors with high frequency microphones capable of recording bat echolocation passes (Zamora-Gutierrez et al. 2021). A bat echolocation pass is a series of calls (individual echolocation 'clicks'), recorded as the bat passes near the microphone (Fenton 1970). Ultrasonic detectors have become increasingly affordable, with reasonable battery life and data storage capacities, making it possible to deploy multiple detectors over a large area for extended periods of time (Hill et al. 2019). However, definitive species identification is often impossible for many recorded passes as there is much overlap in echolocation frequency and pattern among different bat species (Barclay 1999, Walters et al. 2013). Bat echolocation passes vary based on whether the bat is searching, approaching, or attacking prey (Griffin et al. 1960), whether the bat is flying in an open or cluttered habitat (Findlay and Barclay 2020), and whether other bats are nearby (Obrist 1995). There is also regional variation among different populations, suggesting that call libraries and the resulting auto-identification software should be specific to a certain geographic area and habitat (Barclay 1999). Overall, despite some of the limitations, acoustic surveys have provided a cost-effective and consistent method to collect long-term data on bat populations (Zamora-Gutierrez et al. 2021).

Bat echolocation calls vary depending on whether the bat is searching, approaching, or attacking prey resulting in three different call types: search phase, approach phase, and terminal phase (Griffin et al. 1960, Fenton 2013). A search phase call consists of calls that have an approximately consistent interpulse interval (i.e. time between calls) and a relatively low call repetition rate (Griffin et al. 1960). Search phase calls are used when bats are commuting and

searching (Schnitzler and Kalko 2001, Russo et al. 2018a). Search phase calls are the call type that should be used when analyzing bat passes as it is the most diagnostic call type to species or species group (Reichert et al. 2018). Approach phase calls are used when bats are approaching an insect or obstacles such as vegetation (Griffin et al. 1960, Schnitzler and Kalko 2001). These calls are typically steeper in frequency change, shorter in duration, have a faster repetition rate, and may be higher in frequency (Griffin et al. 1960, Russo et al. 2018a). Terminal phase calls consist of a burst of calls that are very steep (i.e. rapidly drop in frequency), short, fast, and often of low amplitude (Griffin et al. 1960). Terminal phase calls provide the final, precise localization information of the target (Jakobsen and Surlykke 2010, Schmieder et al. 2010), and are used when the bat is close to and capturing an insect (Griffin et al. 1960, Russo et al. 2018a). The feeding buzz (i.e. the collection of the terminal phase calls) has an abrupt change in frequency (Kalko 1995, Schnitzler and Kalko 2001, Jakobsen and Surlykke 2010). This drop in frequency is thought to be a result of the bat increasing the width of their biosonar sound beam, thereby increasing the area of detection (Jakobsen and Surlykke 2010, Matsuta et al. 2013). The increased call rate and broad bandwidth assist in successfully tracking maneuvering insects (Schmieder et al. 2010, Elemans et al. 2011, Ratcliffe et al. 2013). Bats also emit buzzes when drinking (Griffiths 2013, Russo et al. 2016) or landing (Russo et al. 2007), but the drop in frequency associated with feeding buzzes is typically absent in drinking and landing buzzes (Russo et al. 2018a).

Social calls are an additional call type and consist of complex sounds that are variable in frequency, shape, and length (Limpens 2004, Szewczak 2018). Social calls are used when bats are communicating with each other (Limpens 2004). Fig. 3.1 illustrates a bat pass consisting of search, approach and terminal phase calls, and an example of a bat pass containing social calls.

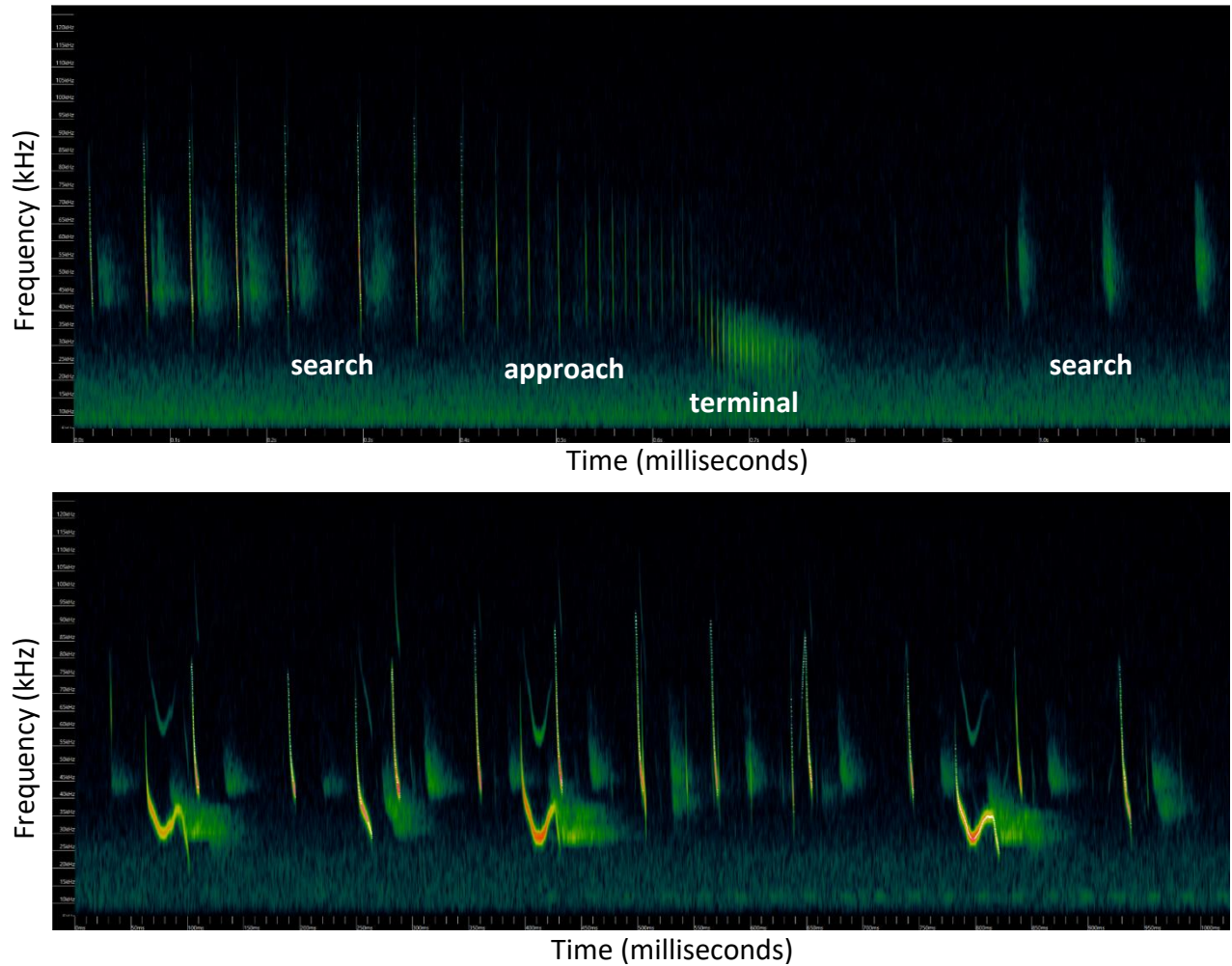


Fig. 3.1. Examples of bat echolocation passes consisting of the three different call types and social calls. The search phase, approach phase, and terminal phase (i.e. feeding buzz) calls are displayed in the top figure and examples of social calls are displayed in the bottom figure. The sonograms are displayed in Wildlife Acoustics' Kaleidoscope Lite bat analysis software (version 5.4.0) in true time.

3.1.3 Hypothesis and predictions

From 2015 to 2020, bat acoustic surveys were conducted by Parks Canada staff in Waterton Lakes National Park (WLNP), providing the unique opportunity to compare bat activity from before the Kenow wildfire (2015 to 2017) to levels after the natural disturbance (2018 to 2020). Based on information from Chapter 1, I hypothesized that the wildfire affected the species' distribution and abundance of bats in WLNP due to changes in the forest structure, foraging opportunities, and roosting availability. Despite the different life histories of the bat species in WLNP, I hypothesized that the fire created a heterogeneous landscape (i.e. created a variety of burned, unburned and edge habitats) that diversified the forest structure, and increased foraging and roosting options. As most of the species in the study area are edge or open specialists, I hypothesized that the reduced forest clutter improved the habitat. For the clutter-adapted species, there were local unburned habitats to relocate to. Although roost trees were consumed by the wildfire, I hypothesized that there would have been an overall increase in the number of potential roost trees (see Chapter 2). For the foliage-roosting species, there were nearby unburned habitats that provided suitable roosting options. Despite the fire likely having both positive and negative effects on the local insect population, I hypothesized that some insect species would have a strong positive response and the local bats, being opportunistic predators, would capitalize on this. Therefore, overall, I hypothesize that the fire would have a positive effect on all species in the study area.

To test my hypotheses, I predicted that: 1) between pre- (2015 to 2017) and post- (2018 to 2020) wildfire years:

- a) post-fire years would have a greater amount of bat activity, the highest in 2018 (1-year post-fire), with a decreasing trend in subsequent years; and
- b) post-fire years would have a higher rate of feeding buzzes, the highest in 2018 (1-year post-fire), with a decreasing trend in subsequent years.

I also predicted that: 2) between burned and unburned sites after the fire:

- a) burned sites would have a greater amount of bat activity; and
- b) burned sites would have a higher rate of feeding buzzes.

3.2 Methods

3.2.1 Study species

Based on capture records from netting surveys conducted in 2011/2012 (Lausen 2012) and 2019/2020 (see Chapter 2), there are at least seven species that have been confirmed in WLNP. These species include: the little brown *Myotis* (*Myotis lucifugus*), long-eared *Myotis* (*Myotis evotis*), long-legged *Myotis* (*Myotis volans*), big brown bat (*Eptesicus fuscus*), silver-haired bat (*Lasionycteris noctivagans*), eastern red bat (*Lasiurus borealis*), and hoary bat (*Lasiurus cinereus*) (Lausen 2012, see Chapter 2). Acoustic surveys have suggested that an additional three species may be found in WLNP (Lausen 2012). These include the western small-footed *Myotis* (*Myotis ciliolabrum*), Yuma *Myotis* (*Myotis yumanensis*) and California *Myotis* (*Myotis californicus*) (Lausen 2012). However, none of these species have been caught during any past trapping surveys.

In WLNP, the bats in the *Myotis* genus include the bat species with the smallest body size and highest echolocation frequency. All *Myotis* in the study area hibernate during the winter, with the reproductive females forming maternity colonies in summer to raise their offspring (Nagorsen and Brigham 1993, Naughton 2012). In western Canada, *Myotis* roost in buildings, bat houses, trees and rock crevices (Nagorsen and Brigham 1993). All of the *Myotis* in WLNP except the long-eared *Myotis*, primarily consume their prey mid-flight (i.e. aerial hawking) (Nagorsen and Brigham 1993). Long-eared *Myotis* also glean prey, hovering in flight and picking their prey off surfaces (Faure and Barclay 1994, Naughton 2012).

The little brown *Myotis* is a medium-sized *Myotis* (mass 6.2 – 10.2 g, forearm length 33.0 – 40.3 mm) (Nagorsen and Brigham 1993) which has a characteristic echolocation frequency (i.e. the frequency of the call at its lowest slope toward the end of the call) between 35 and 40 kHz (Lausen 2016, pers. comm.). Currently, the little brown *Myotis* is provincially listed as “May Be at Risk” (AEP 2020). However, it has been recommended by Alberta's Endangered Species Conservation Committee to be listed as “Endangered” under the Alberta Wildlife Act (L. Wilkinson, pers. comm.). It is federally listed under the Species at Risk Act as “Endangered” (GOC 2021). Individuals forage over water, along forest edges, and in openings, feeding on a variety of insect orders including Diptera, Ephemeroptera, Trichoptera and

Lepidoptera (Nagorsen and Brigham 1993, Clare et al. 2011, 2014a, Nelson and Gillam 2017). The long-eared *Myotis* (mass 4.2 – 8.6 g, forearm length 36.0 – 42.0 mm; Nagorsen and Brigham 1993) has a characteristic echolocation frequency between 30 – 35 kHz (Lausen 2016, pers. comm.). It is provincially listed as “Sensitive” (AEP 2020). Individuals forage in cluttered forest habitats and feed extensively on Lepidoptera (Barclay 1991, Nagorsen and Brigham 1993, Naughton 2012). The long-legged *Myotis* (mass 5.5 – 10.0 g, forearm length 34.0 – 43.0 mm; Nagorsen and Brigham 1993) has a characteristic echolocation frequency of approximately 40 kHz (Lausen 2016, pers. comm.). It is provincially listed as “Undetermined” (AEP 2020). Individuals forage over the forest canopy and water, as well as along forest and cliff edges, feeding extensively on Lepidoptera (Nagorsen and Brigham 1993, Johnson et al. 2007).

In WLNP, the big brown, silver-haired, eastern red, and hoary bats comprise the species with the larger body sizes and lower echolocation frequencies. Both big brown bats (mass 8.8 – 21.9 g, forearm length 43.0 – 52.0 mm) and silver-haired bats (mass 5.8 – 12.4 g, forearm length 39.1 – 43.9 mm) are medium-sized bats (Nagorsen and Brigham 1993) with a characteristic echolocation frequency between 25 – 30 kHz (Lausen 2016, pers. comm.). Big brown bats hibernate during the winter, with reproductive females forming maternity colonies in summer to raise their offspring during the active season (Nagorsen and Brigham 1993, Naughton 2012). Big brown bats roost in buildings, bat houses, trees and rock crevices (Nagorsen and Brigham 1993, Brittingham and Williams 2000). They are provincially listed as “Secure” (AEP 2020). They are aerial hawkers, foraging along forest edges and over the forest canopy and water, feeding extensively on Coleoptera (Brigham and Saunders 1990, Nagorsen and Brigham 1993, Hamilton and Barclay 1998). Silver-haired bats migrate out of Alberta during the winter. Reproductive females form maternity colonies in summer to raise their offspring (Nagorsen and Brigham 1993, Naughton 2012). Silver-haired bats roost in trees (Nagorsen and Brigham 1993) and are provincially listed as “Sensitive” (AEP 2020). They are aerial hawkers, foraging in clearings and over water, feeding extensively on Lepidoptera (Barclay 1986, Nagorsen and Brigham 1993, Reimer et al. 2010).

Both species of *Lasiurus* migrate out of Alberta during the winter, and roost solitarily amongst tree foliage (Nagorsen and Brigham 1993, Naughton 2012). Both species are also aerial

hawkers, foraging above the forest canopy and in open areas, feeding extensively on Lepidoptera (Shump and Shump 1982, Nagorsen and Brigham 1993, Reimer et al. 2010). Eastern red bats are a medium-sized bat (mass 7.2 – 18.5 g, forearm length 34.0 – 42.0 mm; Nagorsen and Brigham 1993) with a characteristic echolocation frequency between 30 – 35 kHz (Lausen 2016, pers. comm.). They are provincially listed as “Sensitive” (AEP 2020). Hoary bats are a large-sized bat (mass 20.1 – 37.9 g, forearm length 50.3 – 57.4 mm; Nagorsen and Brigham 1993) with a characteristic echolocation frequency between 16 – 18 kHz (Lausen 2016, pers. comm.). They are provincially listed as “Sensitive” (AEP 2020).

3.2.2 Acoustic surveys

Parks Canada staff conducted acoustic surveys of bats in WLNP from late June to early August from 2015 to 2020. Surveys for each site were conducted at approximately the same time each year to ensure that sites either captured the pre-volancy or post-volancy period (Parks Canada 2017a). Originally, Parks Canada had planned to do three years of baseline monitoring (2015 to 2017) to better understand the distribution and abundance of bats in WLNP prior to the arrival of white-nose syndrome. However, after the Kenow wildfire went through the park, a unique opportunity to examine the effects of a severe wildfire on the bat community presented itself, and surveys were continued from 2018 to 2020. There were 13 different survey sites (Fig. 3.2) located throughout the Subalpine, Montane and Foothills Parkland natural subregions of the park (Fig. 1.2). The same sites were surveyed in all six years. For each survey site, detectors were deployed continuously for between two and 13 nights. However, due to acoustic detector issues, some nights were excluded from the analysis. In total, I included 380 nights in the analysis, with a mean of 5.1 nights ($SE \pm 0.2$) for each site each year. Sites were situated at elevations between 1284 and 2033 m, located between two and 20 kms from the Waterton townsite, and distances of five to 350 meters from the nearest water feature (Fig. 3.3). The Kenow wildfire burned approximately half of the survey sites (Fig. 1.1), resulting in both burned (six sites) and unburned (seven sites) data, as well as pre-fire (three years) and post-fire (three years) data. The metadata for each acoustic survey site and weather station (used for compiling daily environmental data) are provided in Appendix D.



Fig. 3.2. Examples of six of the thirteen acoustic detector sites monitored from 2015 to 2020 in Waterton Lakes National Park, Alberta. Included are: Bison Paddock (top left), Blakiston Roadside (top right), Dipper at Rowe (middle left), Lost Lake (middle right; K. Low), Sewage Lagoon (bottom left), Wishbone (bottom right).

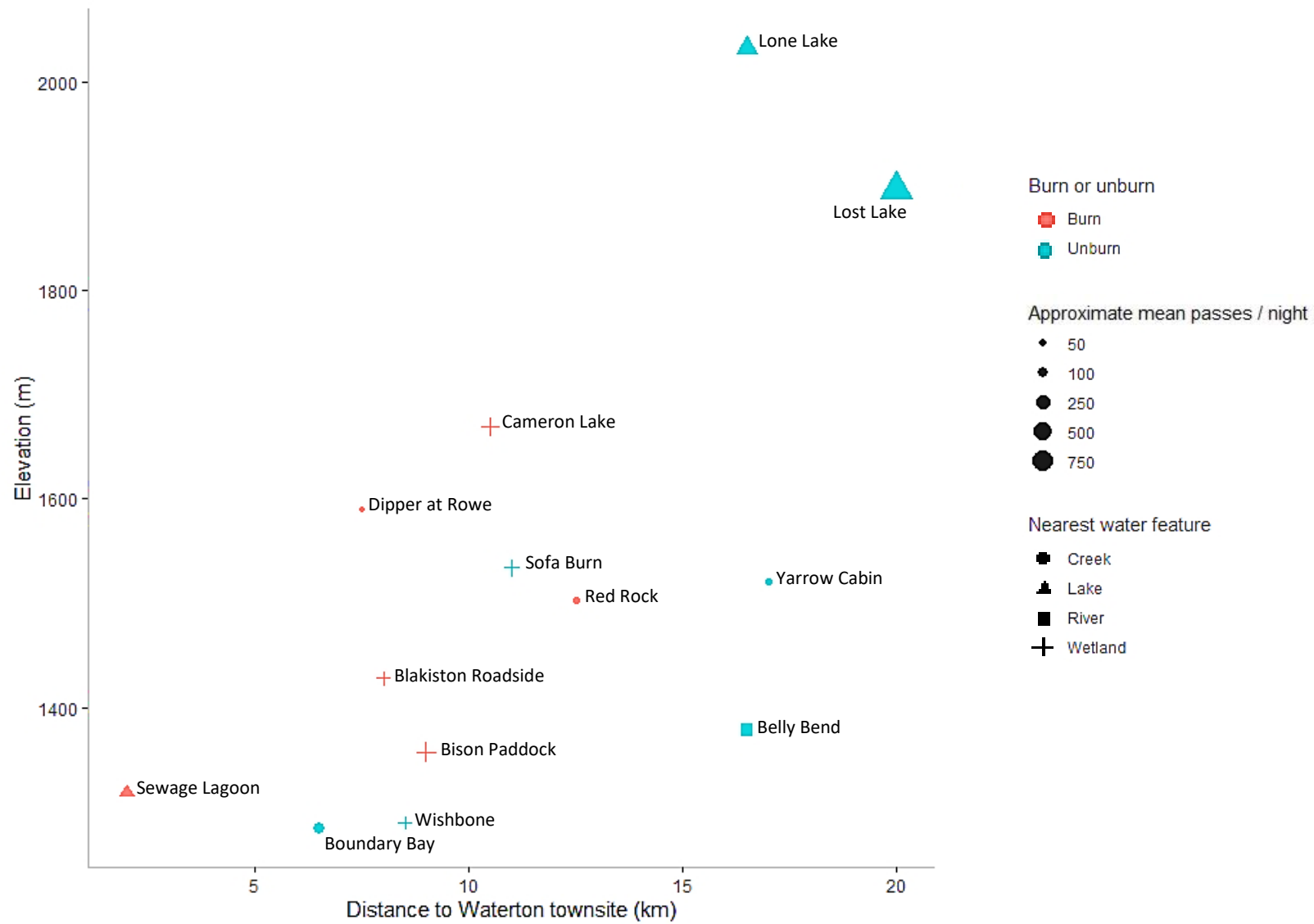


Fig. 3.3. The site variation among the 13 acoustic sites monitored from 2015 to 2020 in Waterton Lakes National Park, Alberta.

3.2.3 Detector deployment

The locations of the acoustic detectors were selected based on the North American Bat Monitoring Program (NABat) probability-based sampling structure (Parks Canada 2017a). The sites were selected from a grid-based finite-area sampling frame, with the exact locations suggested by a bat biologist (Lausen 2012, Parks Canada 2017a). NABat is a collaborative monitoring program across North America that uses standardized protocols to collect acoustic bat data (Loeb et al. 2015). These data help assess population trends and inform conservation-based strategies to support bats due to their vulnerability to a variety of threats (e.g. white-nose syndrome, wind energy development, climate change, habitat loss) (Loeb et al. 2015). Wildlife Acoustics' Song Meter SM2BAT+ detectors and omnidirectional ultrasonic microphones were used throughout the six-year study. The microphones were raised five to six meters from the ground (Fig. 3.4) and were calibrated prior to the start of each field season to ensure they were working within normal parameters. The detectors were programmed to record from 30 minutes before sunset to 30 minutes after sunrise. Additional information regarding detector settings is discussed in Appendix E.

3.2.4 Acoustic call processing

At the end of each field season, the raw acoustic files were processed using Wildlife Acoustics' Kaleidoscope Pro bat analysis software (version 3.1.1, www.wildlifeacoustics.com/products/kaleidoscope-pro), providing both full-spectrum and zero-cross files. Full spectrum recordings are a digitized representation of the entire soundscape, which includes frequency, time, harmonic and amplitude information for each call (Szewczak 2010, Agranat 2013). A zero-cross recording is created by measuring the amount of time it takes a soundwave to cross the 'zero' mark a given number of times (Szewczak 2010, Agranat 2013). The time between the oscillations, or typically the time it takes for eight zero-crossings (called the division ratio), is recorded (Szewczak 2010). These data are then plotted as the average frequencies per time (Szewczak 2010). Zero-cross files can only display the dominant frequency at any one time, and only the frequency and time of each call (i.e., no amplitude or harmonic information) (Szewczak 2010, Agranat 2013).

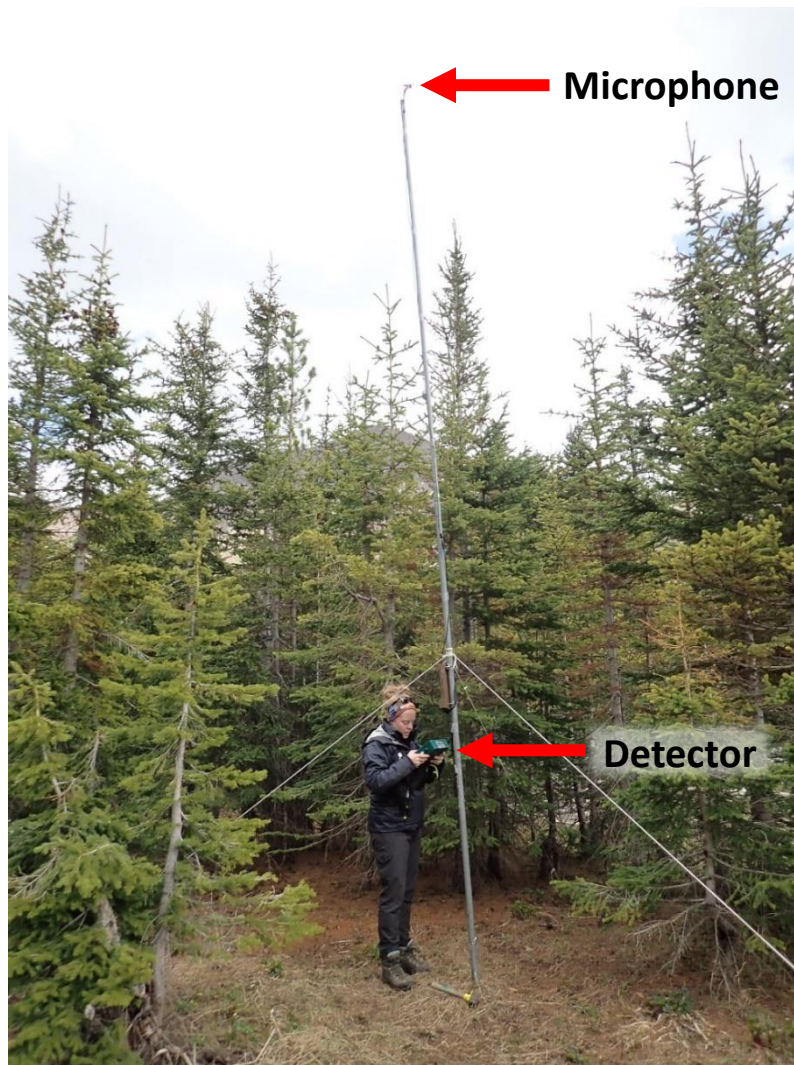


Fig. 3.4. The acoustic detector deployment set-up at Cameron Lake, Waterton Lakes National Park, Alberta (K. Melrose).

All files were run through the Kaleidoscope noise scrubber and species classifier to move noise files to a separate folder and assign an auto-identified species label to each bat file. Once the bat files were processed in Kaleidoscope Pro, I inspected all of the data log-files for each site and year to ensure detectors were recording during the entire deployment period. I then excluded nights when the detector was not recording normally. I compiled metadata for the surrounding habitat for each detector location (e.g. natural subregions, elevation, distance to the Waterton townsite, nearest water feature), as well as environmental conditions (e.g. sunset temperature, nightly precipitation) for each night a detector was deployed (Appendix D).

To analyze the acoustic files, I used Titley Scientific's AnalookW bat analysis software (version 4.5s) and filters (J. Rae and C. Lausen, pers. comm.). The filters provided a secondary auto-identification method to increase efficiency and consistency when I was manually analyzing the acoustic files. Once all bat files had been auto-identified by both Kaleidoscope Pro and the filters in AnalookW, I manually identified all files. Manual species identification was based on resources (Lausen 2016, pers. comm., Szewczak 2018) provided from bat acoustic training courses I attended. As there is no WLNP specific bat echolocation reference library, I was conservative in assigning species-specific identification and thus grouped species with similar echolocation characteristics (e.g. high frequency, 40kHz *Myotis*, big brown/silver-haired bat, low frequency). The echolocation call characteristics I used for each species/species group and sonogram examples are illustrated in Appendix F. Some files had multiple species recorded. Each different species/species group present in a file was identified and accounted for. Files that appeared to have multiple individuals of the same frequency class were not corrected for, and therefore only counted once.

Although I analyzed files for species analysis using the zero-cross files created from Kaleidoscope Pro, I also used Wildlife Acoustics' Kaleidoscope Lite bat analysis software (version 5.4.0) to examine a subset of the processed full spectrum files created from Kaleidoscope Pro. AnalookW only recognizes zero-cross files, but Kaleidoscope Lite is able to load both zero-cross and full spectrum files. For full spectrum files to be zero-crossed, the signal to noise ratio (i.e. the difference between the detected signal and background noise) needs to be above a certain threshold to be converted. Therefore, many of the low amplitude calls that appear in full-spectrum are absent in zero-cross files. Due to feeding buzzes having low intensity, they rarely appeared in the zero-cross files. I used Kaleidoscope Lite to determine the number of feeding buzzes in a subsample of files (i.e. random sample of 20 individual files or 10% of all files, whichever was greater, for each date for every site for each year; n=6,891) that I manually identified in AnalookW as 40 kHz *Myotis*. 40 kHz *Myotis* were not detected every night, therefore I excluded nights with no detections (i.e. 40 kHz *Myotis* were detected on 361 of the 380 detector nights). I then determined the proportion of files with a feeding buzz (i.e. number of files with a feeding buzz/total number of files subsampled) for each date for every site for

each year. Additional information regarding the software settings, workflow, site metadata compilation, and echolocation requirements are discussed in Appendix E.

In addition to feeding buzzes not being zero-crossed, there were also low-amplitude bat passes that were missed from the zero-cross conversion or incompletely zero-crossed, thus creating an ambiguous echolocation pulse. These bat calls appeared as additional species in the full spectrum bat files, as well as in the files that were auto-identified as “noise” by Kaleidoscope Pro, and in the files I manually identified as “noise” in AnalookW. I manually inspected a subsample of these files in Kaleidoscope Lite using the corresponding full spectrum file. For each analysis (i.e. the number of full spectrum bat files containing additional species [n=3,626], the number of “noise” files from Kaleidoscope Pro containing bats [n=3,800], and the number of “noise” files from AnalookW containing bats [n=2,055]), ten individual files were selected for each date for every site for each year. Some dates had less than 10 files, therefore I selected however many files were available for that date (i.e. one to nine). Additional information regarding the subsampled files, the proportions of missed species, and the overall limitations of using the Kaleidoscope noise scrubber and analyzing the zero-cross files instead of the full spectrum files, is discussed in Appendix G.

3.2.5 Statistical analysis

To determine the effect of the wildfire on bats in WLNP, I compared activity levels (i.e. the number of echolocation passes per night) between pre- and post-wildfire years, as well as between burned and unburned sites. I tested the number of echolocation passes per night for each species/species group for normality using a Shapiro-Wilk test (function `shapiro.test()` in package *stats*, R Core Team 2020) and by examining histograms (function `hist()` in package *graphics*, R Core Team 2020) of the plotted data in RStudio (version 1.3.1093, R Core Team 2020). I conducted statistical analyses on four different species/species groups; 40 kHz *Myotis* (i.e. little brown *Myotis*, long-legged *Myotis* and non-diagnostic long-eared *Myotis* passes), big brown/silver-haired bats, hoary bats, and long-eared *Myotis* (i.e. diagnostic long-eared *Myotis* passes). I also examined the proportion of 40 kHz *Myotis* passes that contained feeding buzzes (i.e. 40 kHz *Myotis* feeding buzz). All species/species groups were from non-normal distributions.

For species/species groups with sufficient sample sizes to model (i.e. models were able to converge), I modeled the effects of environmental variables on bat activity using generalized linear mixed models (GLMMs), with a negative binomial distribution (function `glmer.nb()` in package *lme4*, Bates et al. 2015). I created separate GLMMs for 40 kHz *Myotis*, big brown/silver-haired bat, and hoary bat. Within each species/species group, I created three different models to predict the echolocation counts for each fire metric: 1) among the years before the wildfire to each year after the wildfire (PrePost123); 2) between the burned and unburned sites both before and after the wildfire (PrePostUnburnBurn); and 3) between burned and unburned sites, and considering how many years since the wildfire occurred (PrePost123UnburnBurn). Although there was variation in seasonal activity from year to year due to differences in the weather (e.g. cold, wet spring, dry, hot summer), having three years of pre-fire data likely captured much of the yearly variation that was present. Therefore, I combined all pre-fire data (i.e. 2015 to 2017), thus representing the baseline activity of bats in WLNP prior to the Kenow wildfire.

Possible explanatory variables originally included in the models included the fire metric (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn), natural subregion, elevation (m), townsite distance (km), water distance (km), type of nearest water feature, total nightly precipitation (mm), forest type, sunset temperature (°C), and a random effect of the acoustic site (Table 3.1). Although including “site” as the random effect would have accounted for some of the characteristics I included as additional explanatory variables (e.g. natural subregion, elevation, forest type), I was interested in specifically how each of the included covariates affected bat abundance (i.e. which habitat features were associated with the greatest bat activity and the direction of the effect). All numerical variables were scaled for better model convergence using the *scale()* function in base R, except for the responding variable (i.e. the number of echolocation passes per night). An information-theoretic approach based on Akaike's information criterion corrected for small sample size (AICc) was used to select the most parsimonious model (function `dredge()` in package *MuMIn*, Barton 2020). I ranked models by AICc where smaller values represent better models, and for models within two $\Delta AICc$, the most simple model was selected (Burnham and Anderson 2002).

Table 3.1. The possible explanatory variables originally included in the generalized linear mixed models (GLMMs) that examined the effect of the Kenow wildfire on bats in Waterton Lakes National Park, Alberta. I used the number of echolocation passes per night to explore activity differences between pre- and post-wildfire years, as well as between the burned and unburned sites. Separate models were created for 40 kHz *Myotis* (40kMyotis), big brown/silver-haired bats (EPFULANO), and hoary bats (LACI).

Variable name	Fixed/random effect or response variable	Level names	Variable description
PrePost123	Fixed	Pre, Post1, Post2, Post3	“Pre” = Unburned and burned sites from 2015 – 2017, “Post1” = Unburned and burned sites in 2018, “Post2” = Unburned and burned sites in 2019, “Post3” = Unburned and burned sites in 2020
PrePostUnburnBurn	Fixed	PreUnburn, PreBurn, PostUnburn, PostBurn	“PreUnburn” = Unburned sites from 2015 – 2017, “PreBurn” = Burned sites from 2015 – 2017, “PostUnburn” = Unburned sites from 2018 – 2020, “PostBurn” = Burned sites from 2018 - 2020
PrePost123UnburnBurn	Fixed	PreUnburn, PreBurn, PostUnburn1, PostUnburn2, PostUnburn3, PostBurn1, PostBurn2, PostBurn3	“PreUnburn” = Unburned sites from 2015 – 2017, “PreBurn” = Burned sites from 2015 – 2017, “PostUnburn1” = Unburned sites in 2018, “PostUnburn2” = Unburned sites in 2019, “PostUnburn3” = Unburned sites in 2020, “PostBurn1” = Burned sites in 2018, “PostBurn2” = Burned sites in 2019, “PostBurn3” = Burned sites in 2020
Natural subregions	Fixed	Foothills Parkland, Lower Subalpine, Montane, Upper Subalpine	The different natural subregions found in the park
Elevation	Fixed	1284 – 2033	Elevations of the 13 sites in meters
Townsite distance	Fixed	2 – 20	Distance from each acoustic station to the townsite in kms
Water distance	Fixed	0.005 – 0.35	Distance from each acoustic station to the nearest water source in kms
Nearest water feature	Fixed	Creek, Lake, River, Wetland	The type of water feature nearest to each acoustic station
Night sum precipitation	Fixed	0.0 – 19.7	The total precipitation in mm for each detector night
Forest type	Fixed	Coniferous, Deciduous, Mixed	The different forest types found in the park
Sunset temperature	Fixed	3.8 – 25.3	The temperature at sunset in °C for each detector night
Site name	Random	Belly Bend, Bison Paddock, Blakiston Roadside, Boundary Bay, Cameron Lake, Dipper at Rowe, Lone Lake, Lost Lake, Red Rock, Sewage Lagoon, Sofa Burn, Wishbone, Yarrow Cabin	The names of the acoustic stations
40kMyotis, EPFULANO, LACI	Response variable	N/A	The number of echolocation passes per night for each species/species group

I initially ran models with all possible explanatory variables. However, some models would not converge when all variables were included. Therefore, I tested for correlation to determine which variables should be removed from the model. I tested for correlation by calculating the variance inflation factor (VIF, function `vif()` in package *car*, Fox and Weisberg 2019), generating a correlation matrix (function `cor()` in package *ggcorrplot*, Kassambara 2019), and calculating Cramer's V (function `assocstats()` in package *vcd*, Meyer et al. 2020). For the VIF test (collinearity between all variables), I removed all variables with values greater than three, the threshold value suggested by Zuur et al. (2009). For the correlation matrix (collinearity between numerical variables; e.g. elevation, townsite distance, sunset temperature), I removed variables with correlation coefficients (positive or negative) that had "moderate" or higher correlation (i.e. $\geq \pm 0.5$) (Mukaka 2012). For the Cramer's V calculation (collinearity between categorical variables; e.g. natural subregions, nearest water feature, forest type), I removed variables with correlation coefficients that had "very strong" correlation (i.e. > 0.25) (Akoglu 2018). All correlation tests showed similar trends; natural subregions, elevation and forest type were correlated. For each model generated, I also checked that the residuals were normally distributed (function `qqnorm()` in package *stats*, Fox and Weisberg 2019), and checked for overfitting (i.e. singular fit; function `getME()` in package *lme4*, Bates et al. 2015), and overdispersion (Bolker 2008).

I completed post-hoc tests (function `summary()` in package *base*, R Core Team 2020 and function `emmeans()` in package *emmeans*, Lenth 2021) to further examine the relationships between the different fire-metric variables. The estimated marginal means (EMMs) post-hoc tests had lower statistical power compared to the generated model summary output due to how many groups were compared. Therefore, I used the summary output when available. The R^2 value was calculated for each model using Nakagawa's R^2 for mixed models (function `r2_nakagawa()` in package *performance*, Lüdtke et al. 2020). Both the marginal (i.e. variance explained by fixed effects) and conditional (i.e. variance explained by fixed and random effects) R^2 values were calculated to measure the strength of each model (Nakagawa et al. 2017). I created prediction plots based on the GLMMs using the package *ggeffects* (Lüdtke 2018).

For two of the species/species groups, long-eared *Myotis* and 40 kHz *Myotis* feeding buzz, the models would not converge or the model diagnostics (i.e. residuals, singularity, and overdispersion) were not within acceptable parameters. I tried various model types, optimizers, distributions, packages and data transformations. However, I was unable to create a model that was appropriate. Therefore, I conducted Kruskal-Wallis rank sum tests (function `kruskal.test()` in package *stats*, R Core Team 2020) to examine the effect of each fire metric (i.e. Prepost123, PrePostUnburnBurn, and PrePost123UnburnBurn) on the number of echolocation passes per night for long-eared *Myotis* and 40 kHz *Myotis* feeding activity. When the results were significant, I performed a Dunn's test of multiple comparisons (function `dunnTest()` in package *FSA*, Ogle et al. 2021) with a Bonferroni correction to control for the experiment-wise error rate. Results were displayed graphically in boxplots (function `boxplot()` in package *graphics*, R Core Team 2020). The hoary bat PrePostUnburnBurn GLMM would also not converge, therefore I performed a Kruskal-Wallis rank sum test and Dunn's test of multiple comparisons on those data. In addition to testing the effect of each fire metric (i.e. Prepost123, PrePostUnburnBurn, and PrePost123UnburnBurn) on the proportion of 40 kHz *Myotis* files with a feeding buzz (i.e. number of files with a feeding buzz/total number of files subsampled), I also tested if there was a difference in the raw number of feeding buzzes in the burned sites between pre- and post-fire years. To assess this, I conducted a Wilcoxon rank sum test with a continuity correction (functions `shapiro.test()` and `wilcox.test()` in package *stats*, R Core Team 2020).

3.3 Results

3.3.1 Species analysis

In total, after raw acoustic files were processed in Kaleidoscope Pro and days with suspected detector issues were discarded, 380 detector-nights were included in the analysis over the six years (54 detector-nights discarded). Acoustic survey dates ranged from late June to early August (Fig. 3.5). Detectors were deployed for between two and 13 nights for each site during each year, with a mean of 5.1 nights (SE \pm 0.2). This resulted in 75,225 zero-cross files during the six-year study. Of the total zero-cross files, 10,035 files (13.3%) were manually identified as noise, with 65,190 files manually identified as a bat species/species group. Of the bat files, 6,031 (9.3%) had two or more species groups recorded in the file. In total, including

correcting for files with more than one species group, 71,353 passes were detected. For each detector night, between 0 – 2,994 bats were detected, with a mean of 187.8 (SE \pm 18.2) passes/night. High frequency bats were detected more commonly than low frequency bats, with 40 kHz *Myotis* detected most often (Table 3.2). Of the low frequency bats, big brown/silver-haired bat was the most common species group (Table 3.2). The mean passes/night for each site for the pre- and post-fire years are illustrated in Table 3.3. The two sites with the highest bat activity, Lost Lake and Lone Lake, had much higher counts than any of the other sites (978.9 passes/night and 333.0 passes/night, respectively). All of the remaining sites had counts of 176.7 passes/night or less. The mean, excluding Lost Lake and Lone Lake, was 91.2 (SE \pm 8.2) passes/night. The mean passes/night for each species/species group (i.e. 40 kHz *Myotis*, big brown/silver-haired bat, hoary bat, and long-eared *Myotis*) for each site and year (i.e. 2015 to 2020) are illustrated in Appendix H.

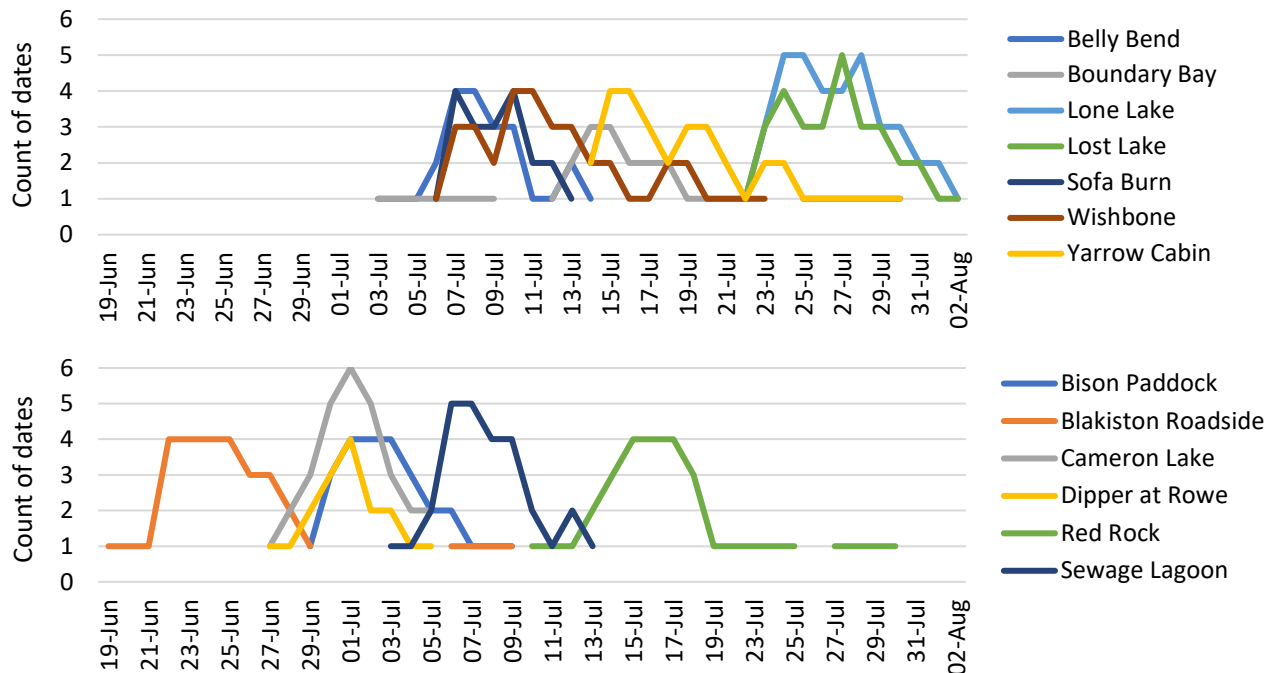


Fig. 3.5. The survey date range for each acoustic detector site over the six-year study in Waterton Lakes National Park, Alberta. Sites that span a small date range and have larger values for “counts of dates” are sites with less variation in the deployment dates over the six years. Conversely, sites that span a large date range and have smaller values for “counts of dates” are sites that were monitored across a larger time period. The top figure displays the unburned sites, the bottom figure displays the burned sites.

Table 3.2. The percentage of total bat activity for each species/species group detected, and the total number of bat echolocation passes from 2015 to 2020 in Waterton Lakes National Park, Alberta.

Bat species/group	Percentage of detections	Total passes
High frequency	73.1%	52,184
40 kHz <i>Myotis</i>	50.8%	36,236
HighF*	20.7%	14,761
Long-eared <i>Myotis</i>	1.4%	1,031
Eastern red bat	0.2%	156
Low frequency	26.6%	19,001
Big brown/silver-haired bat	15.4%	11,015
LowF†	7.0%	5,000
Hoary bat	4.2%	2,986
30 kHz‡	0.2%	168
Total	100.0%	71,353

*non-diagnostic echolocation passes with a characteristic frequency *above* 30 kHz

†non-diagnostic echolocation passes with a characteristic frequency *below* 30 kHz

‡non-diagnostic echolocation passes with a characteristic frequency *of* 30 kHz

Table 3.3. The total number of echolocation passes, the total detector nights monitored, and the mean passes/night \pm SE for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta. Results are displayed grouping burned and unburned sites, as well as pre- (2015 to 2017) and post-fire (2018 to 2020) years.

Site name	Total passes	Total detector nights	Mean passes/night \pm SE	
			PRE-FIRE	POST-FIRE
Burn	16,290	164	81.8 \pm 10.0	116.1 \pm 25.9
Bison Paddock	4,595	26	143.3 \pm 23.1	205.4 \pm 126.5
Blakiston Roadside	1,894	32	78.0 \pm 29.4	40.4 \pm 14.4
Cameron Lake	4,037	29	85.8 \pm 36.9	182.6 \pm 60.2
Dipper at Rowe	605	17	57.2 \pm 18.3	11.2 \pm 1.9
Red Rock	1,336	32	45.8 \pm 10.8	37.8 \pm 4.1
Sewage Lagoon	3,823	28	86.5 \pm 11.0	186.6 \pm 40.5
Unburn	55,063	216	258.6 \pm 40.5	251.6 \pm 42.4
Belly Bend	2,941	24	134.8 \pm 25.2	110.3 \pm 18.1
Boundary Bay	2,827	26	225.8 \pm 47.3	56.7 \pm 10.7
Lone Lake	12,653	38	364.9 \pm 70.7	301.1 \pm 35.4
Lost Lake	30,346	31	696.6 \pm 175.1	1369.8 \pm 140.1
Sofa Burn	2,804	26	132.3 \pm 37.9	92.6 \pm 35.4
Wishbone	2,021	37	72.4 \pm 10.8	33.7 \pm 4.5
Yarrow Cabin	1,471	34	47.5 \pm 7.4	39.9 \pm 8.5
Total (burn and unburn)	71,353	380	180.9 \pm 24.0	194.1 \pm 27.1

3.3.2 40 kHz *Myotis* feeding buzz analysis

To assess changes in 40 kHz *Myotis* feeding activity (i.e. from pre- to post-fire and between burned and unburned areas), I examined a subsample of 40 kHz *Myotis* files to determine the proportion of files with a feeding buzz for each date for every site for each year. In total, there were 36,224 full spectrum 40 kHz *Myotis* files, detected across 361 of the 380 detector nights. By subsampling, I manually analyzed 6,891 files, identifying 787 feeding buzzes. For each detector night, between 0 – 100% of the 40 kHz *Myotis* files contained feeding buzzes. The mean percentage of files containing feeding buzzes across the six years was 9.2%. The percent of subsampled 40 kHz *Myotis* files that contained feeding buzzes for each site is illustrated in Appendix I.

3.3.3 Generalized linear mixed model selection

For most of the species/species groups (i.e. 40 kHz *Myotis*, big brown/silver-haired, and hoary bat) the fire metric (i.e. PrePost123, PrePostUnburnBurn, and PrePost123UnburnBurn), nearest water feature, and sunset temperature influenced nightly echolocation counts. Total

nightly precipitation, townsite distance, and water distance were also variables in some of the top-ranked models (i.e. models with the lowest AICc value and highest AICc weight; Burnham et al. 2011). In most models, the conditional R^2 values of the top-ranked models were ≥ 1.3 times the marginal R^2 values, emphasizing the importance of site as a random effect. The conditional R^2 values of all models were ≥ 0.665 , with a mean of 0.687.

The top-ranked model for 40 kHz *Myotis* PrePost123 included nearest water feature, total nightly precipitation, and sunset temperature. The conditional R^2 value of the top-ranked model was 0.665. The top-ranked model for 40 kHz *Myotis* PrePostUnburnBurn included nearest water feature, total nightly precipitation, PrePostUnburnBurn, and sunset temperature. The conditional R^2 value of the top-ranked model was 0.686. The top-ranked model for 40 kHz *Myotis* PrePost123UnburnBurn included nearest water feature, PrePost123UnburnBurn, sunset temperature, and water distance. The conditional R^2 value of the top-ranked model was 0.701. The top and candidate models (i.e. models within $\Delta AICc < 2$; Burnham and Anderson 2002) for 40 kHz *Myotis* are illustrated in Table 3.4.

The top-ranked model for big brown/silver-haired bat PrePost123 included nearest water feature, PrePost123, sunset temperature, and townsite distance. The conditional R^2 value of the top-ranked model was 0.681. The top-ranked model for big brown/silver-haired bat PrePostUnburnBurn included nearest water feature, PrePostUnburnBurn, sunset temperature, and townsite distance. The conditional R^2 value of the top-ranked model was 0.680. The top-ranked model for big brown/silver-haired bat PrePost123UnburnBurn included nearest water feature, PrePost123UnburnBurn, sunset temperature, and townsite distance. The conditional R^2 value of the top-ranked model was 0.690. The top and candidate models for big brown/silver-haired bat are illustrated in Table 3.5.

The top-ranked model for hoary bat PrePost123 included nearest water feature, PrePost123, and sunset temperature. The conditional R^2 value of the top-ranked model was 0.676. The model for hoary bat nightly echolocation count for PrePostUnburnBurn failed to converge, therefore, no GLMM was created for this group. The top-ranked model for hoary bat PrePost123UnburnBurn included nearest water feature, PrePost123UnburnBurn, and sunset temperature. The conditional R^2 value of the top-ranked model was 0.715. The top and candidate models for hoary bat are illustrated in Table 3.6.

Table 3.4. Model selection factors influencing the echolocation counts for 40 kHz *Myotis* (40kMyotis) in Waterton Lakes National Park, Alberta from 2015 to 2020. The three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) were modeled separately in negative binomial generalized linear mixed models, including site as a random effect. The top candidate models according to the AICc are included with the top-ranked model bolded; I picked the most simple model within $\Delta AICc < 2$. “K” refers to the degrees of freedom.

Response variable	Fire metric	Model	K	AICc	$\Delta AICc$	AICc weight
40kMyotis echolocation count	PrePost123	NearestWaterFeature + NightSumPrecip + SunsetTemp	8	3736.9	0.00	0.271
		NearestWaterFeature + NightSumPrecip + SunsetTemp + WaterDistance	9	3737.8	0.85	0.178
		NearestWaterFeature + NightSumPrecip + SunsetTemp + TownsiteDistance	9	3738.1	1.13	0.154
		NearestWaterFeature + NightSumPrecip + SunsetTemp + TownsiteDistance + WaterDistance	10	3738.8	1.84	0.108
40kMyotis echolocation count	PrePostUnburnBurn	NearestWaterFeature + NightSumPrecip + PrePostUnburnBurn + SunsetTemp + WaterDistance	12	3723.2	0.00	0.439
		NearestWaterFeature + NightSumPrecip + PrePostUnburnBurn + SunsetTemp	11	3725.0	1.81	0.178
		NearestWaterFeature + NightSumPrecip + PrePostUnburnBurn + SunsetTemp + TownsiteDistance + WaterDistance	13	3725.4	2.14	0.151
		NearestWaterFeature + PrePostUnburnBurn + SunsetTemp + WaterDistance	11	3726.3	3.06	0.095
40kMyotis echolocation count	PrePost123UnburnBurn	NearestWaterFeature + NightSumPrecip + PrePost123UnburnBurn + SunsetTemp + WaterDistance	16	3719.3	0.00	0.284
		NearestWaterFeature + PrePost123UnburnBurn + SunsetTemp + WaterDistance	15	3720.2	0.91	0.180
		NearestWaterFeature + NightSumPrecip + PrePost123UnburnBurn + SunsetTemp	15	3720.6	1.38	0.143
		NearestWaterFeature + NightSumPrecip + PrePost123UnburnBurn + SunsetTemp + TownsiteDistance + WaterDistance	17	3721.4	2.16	0.096

Table 3.5. Model selection factors influencing the echolocation counts for big brown/silver-haired bats (EPFULANO) in Waterton Lakes National Park, Alberta from 2015 to 2020. The three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) were modeled separately in negative binomial generalized linear mixed models, including site as a random effect. The top candidate models according to the AICc are included, with the top-ranked model bolded; I picked the most simple model within $\Delta AICc < 2$. “K” refers to the degrees of freedom.

Response variable	Fire metric	Model	K	AICc	$\Delta AICc$	AICc weight
EPFULANO echolocation count	PrePost123	NearestWaterFeature + PrePost123 + SunsetTemp + TownsiteDistance	11	2901.6	0.00	0.310
		NearestWaterFeature + NightSumPrecip + PrePost123 + SunsetTemp + TownsiteDistance	12	2901.8	0.23	0.276
		NearestWaterFeature + PrePost123 + SunsetTemp + TownsiteDistance + WaterDistance	12	2903.7	2.10	0.108
		NearestWaterFeature + NightSumPrecip + PrePost123 + SunsetTemp + TownsiteDistance + WaterDistance	13	2904.0	2.36	0.095
EPFULANO echolocation count	PrePostUnburnBurn	NearestWaterFeature + PrePostUnburnBurn + SunsetTemp + TownsiteDistance	11	2897.2	0.00	0.352
		NearestWaterFeature + NightSumPrecip + PrePostUnburnBurn + SunsetTemp + TownsiteDistance	12	2897.8	0.56	0.266
		NearestWaterFeature + PrePostUnburnBurn + SunsetTemp + TownsiteDistance + WaterDistance	12	2899.4	2.13	0.121
		NearestWaterFeature + NightSumPrecip + PrePostUnburnBurn + SunsetTemp + TownsiteDistance + WaterDistance	13	2899.9	2.69	0.092
EPFULANO echolocation count	PrePost123UnburnBurn	NearestWaterFeature + PrePost123UnburnBurn + SunsetTemp + TownsiteDistance	15	2899.4	0.00	0.361
		NearestWaterFeature + NightSumPrecip + PrePost123UnburnBurn + SunsetTemp + TownsiteDistance	16	2900.3	0.84	0.237
		NearestWaterFeature + PrePost123UnburnBurn + SunsetTemp + TownsiteDistance + WaterDistance	16	2901.6	2.17	0.122
		PrePost123UnburnBurn + SunsetTemp + TownsiteDistance	12	2902.3	2.93	0.084

Table 3.6. Model selection factors influencing the echolocation counts for hoary bats (LACI) in Waterton Lakes National Park, Alberta from 2015 to 2020. The three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) were modeled separately in negative binomial generalized linear mixed models, including site as a random effect. The top candidate models according to the AICc are included, with the top-ranked model bolded; I picked the most simple model within $\Delta AICc < 2$. Note, five candidate models are shown for PrePost123, as there were five models with $\Delta AICc < 2$. “K” refers to the degrees of freedom.

Response variable	Fire metric	Model	K	AICc	$\Delta AICc$	AICc weight
LACI echolocation count	PrePost123	NearestWaterFeature + PrePost123 + SunsetTemp + TownsiteDistance	11	1973.2	0.00	0.206
		NearestWaterFeature + PrePost123 + SunsetTemp + TownsiteDistance + WaterDistance	12	1973.9	0.75	0.142
		NearestWaterFeature + PrePost123 + SunsetTemp	10	1974.2	0.95	0.128
		PrePost123 + SunsetTemp + TownsiteDistance	8	1974.2	1.01	0.124
		NearestWaterFeature + PrePost123 + SunsetTemp + WaterDistance	11	1975.1	1.93	0.079
LACI echolocation count	PrePostUnburnBurn	Model failed to converge				
LACI echolocation count	PrePost123UnburnBurn	NearestWaterFeature + PrePost123UnburnBurn + SunsetTemp	14	1947.0	0.00	0.481
		NearestWaterFeature + PrePost123UnburnBurn + SunsetTemp + TownsiteDistance	15	1949.0	1.95	0.181
		NearestWaterFeature + PrePost123UnburnBurn + SunsetTemp + WaterDistance	15	1949.1	2.06	0.171
		NearestWaterFeature + NightSumPrecip + PrePost123UnburnBurn + SunsetTemp	15	1949.2	2.12	0.166

3.3.4 Fire metrics' results

Activity levels for almost all species/species groups (i.e. 40 kHz *Myotis*, big brown/silver-haired bat, hoary bat, long-eared *Myotis*, and 40 kHz *Myotis* feeding buzz) were influenced by year (i.e. pre- versus post-fire years and the number of years since the fire occurred) as well as whether the site burned or remained unburned after the fire. For 40 kHz *Myotis* (Appendix J), there was no significant change in overall activity pre- versus post-fire (Fig. 3.6a), but activity increased in burned areas and decreased in unburned areas after the fire (Fig. 3.6b). The increase in activity in the burned areas was most apparent in the year immediately following the fire, with the largest decrease in the unburned areas occurring in the third year after the fire (Fig. 3.6c). Prior to the fire, activity was higher in unburned areas compared to burned areas, while after the fire there was no significant difference (Fig. 3.6b).

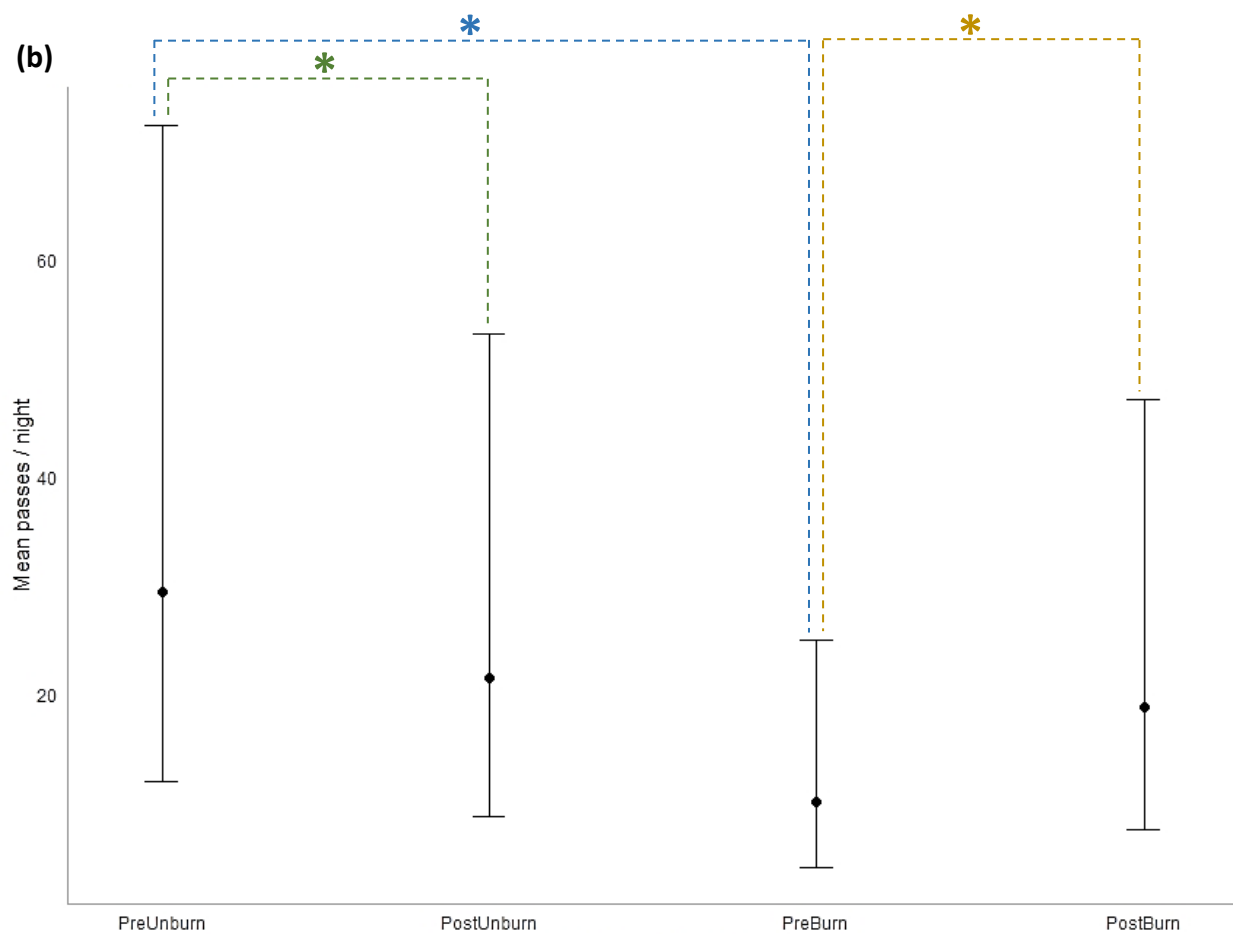
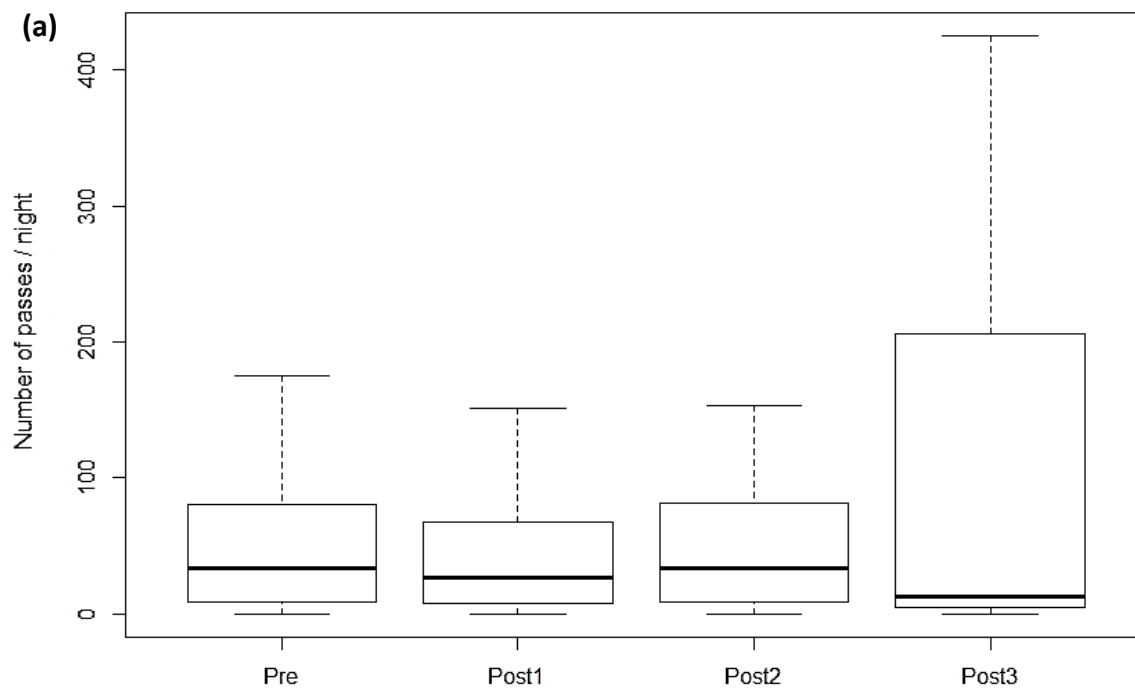
Big brown/silver-haired bat (Appendix K) activity decreased post-fire (Fig. 3.7a), but this trend was predominantly in the unburned areas (Fig. 3.7b). Activity levels did not change significantly in the burned areas pre- versus post-fire (Fig. 3.7b), and there were no significant differences between the three years post-fire for the unburned sites (Fig. 3.7c). Big brown/silver-haired bats were associated with both burned and unburned sites equally, pre- and post-fire (Fig. 3.7b). Hoary bat (Appendix L) activity decreased significantly after the fire (Fig. 3.8a) in both burned and unburned areas (Fig. 3.8b). However, activity levels increased to similar pre-fire levels by the third-year post-fire in both burned and unburned areas (Fig. 3.8c). For both pre- and post-fire years, hoary bats were associated more with unburned sites compared to burned sites (Fig. 3.8b).

Long-eared *Myotis* (Appendix M) activity decreased significantly after the fire (Fig. 3.9a), primarily in the burned areas in the second and third years (Fig. 3.9b, Fig. 3.9c). Long-eared *Myotis* were more closely associated with sites in the burned areas of the park before the fire, but there was no significant difference between the burned versus unburned areas after the fire (Fig. 3.9b). For 40kHz *Myotis* feeding buzzes (Appendix I), there was no significant difference in the proportion of echolocation calls that contained buzzes pre- versus post-fire (Fig. 3.10a). However, activity was higher in the unburned sites compared to the burned sites

before the fire, and there was no significant difference between the sites after the fire (Fig. 3.10b, Fig. 3.10c).

3.3.5 Site and environmental covariates' results

For 40 kHz *Myotis* (Appendix J), big brown/silver-haired bat (Appendix K), and hoary bat (Appendix L), increased sunset temperatures had a positive effect on nightly echolocation counts. Increased total nightly precipitation had a negative effect on nightly echolocation counts. 40 kHz *Myotis* and big brown/silver-haired bat were most associated with sites near to lakes, while hoary bat activity levels were highest near the river site. Townsite distance was significant in all big brown/silver-haired bat models, showing a positive relationship between townsite distance and activity levels (i.e. as the distance from town increased, nightly echolocation counts increased). Water distance was only significant in the 40 kHz *Myotis* PrePost123UnburnBurn model. There was a negative relationship between water distance and activity levels (i.e. as the distance from water increased, nightly echolocation counts decreased).



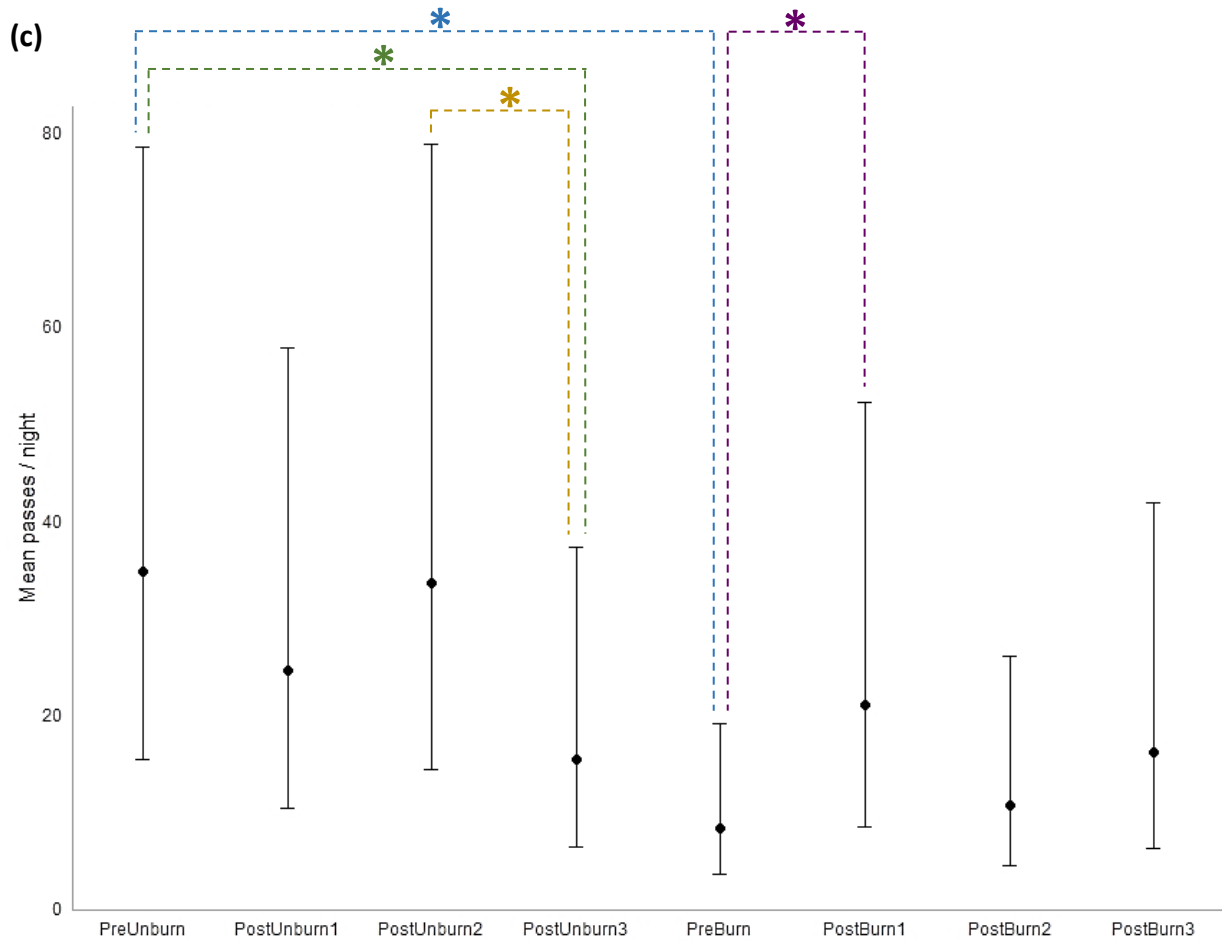
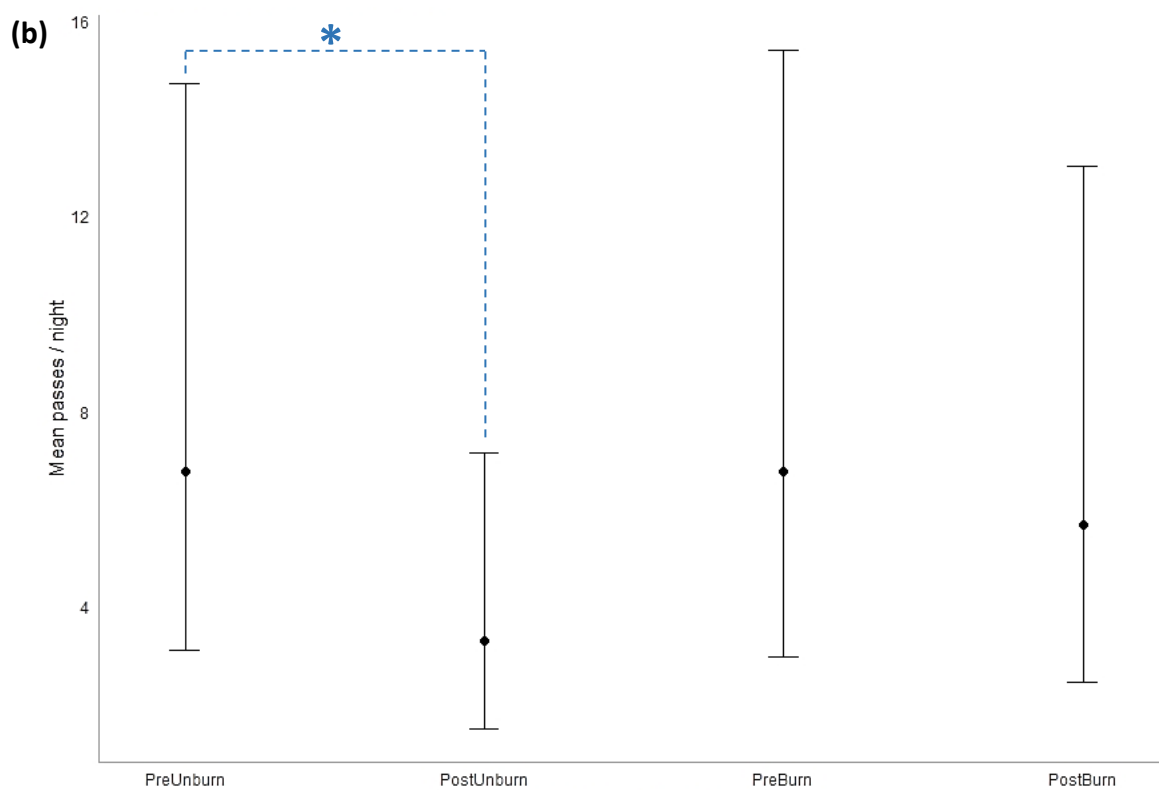
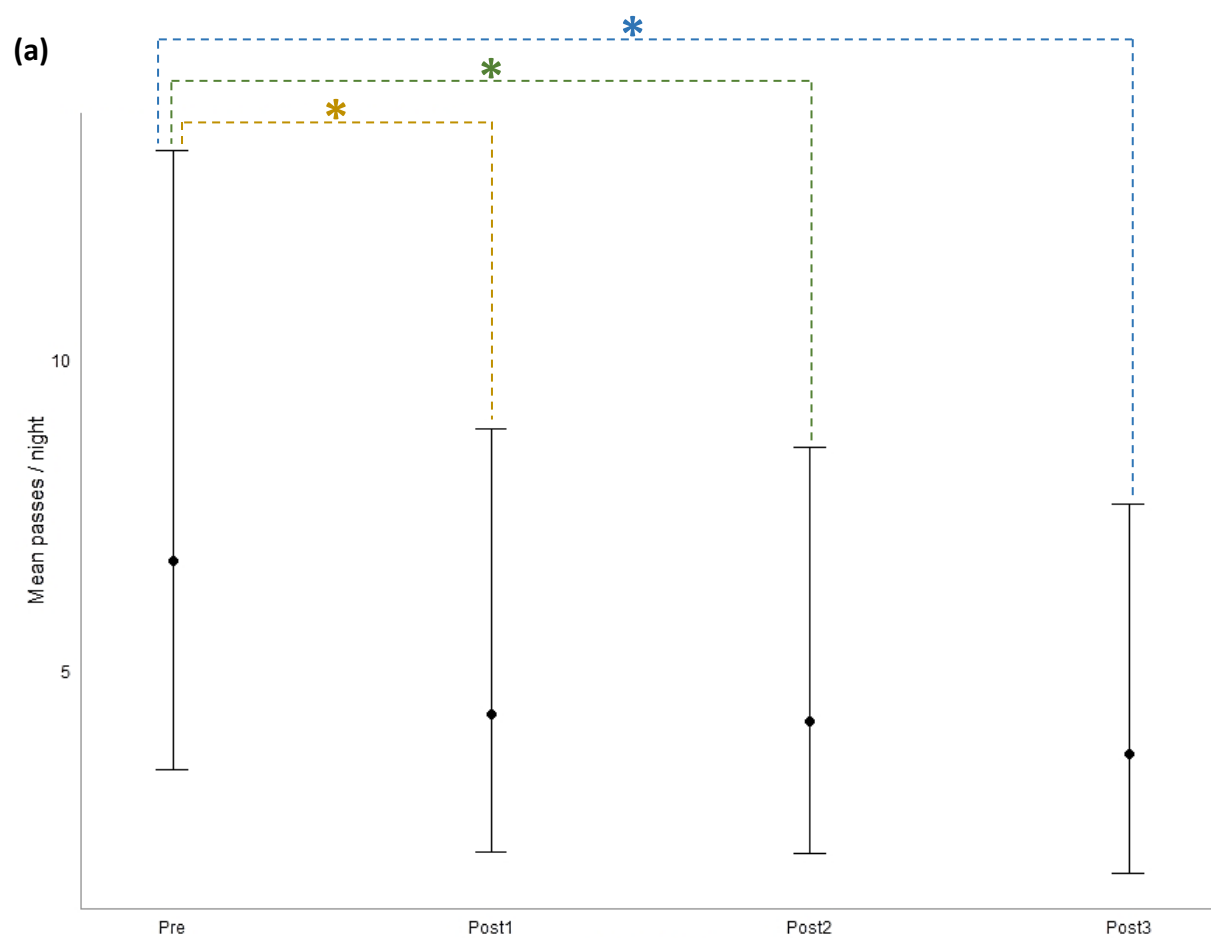


Fig. 3.6. The 40 kHz *Myotis* nightly activity plotted for each fire metric. The number of passes/night for PrePost123 is shown in (a), and the mean passes/night predicted by the associated generalized linear mixed model (i.e. controlling for the effects of the covariates) for PrePostUnburnBurn and PrePost123UnburnBurn are shown in (b) and (c), respectively. For the boxplot in (a) and all subsequent boxplots, the values displayed represent the minimum, first quartile, median, third quartile, and maximum of the data. The values displayed in the boxplot in (a) are based on the fire metric only (i.e. PrePost123) and do not control for the effects of any covariates. The outliers have been removed in the boxplot and the error bars in the prediction plots show the 95% confidence interval. The asterisks denote a significant difference ($P < 0.05$), with each colour indicating a separate comparison.



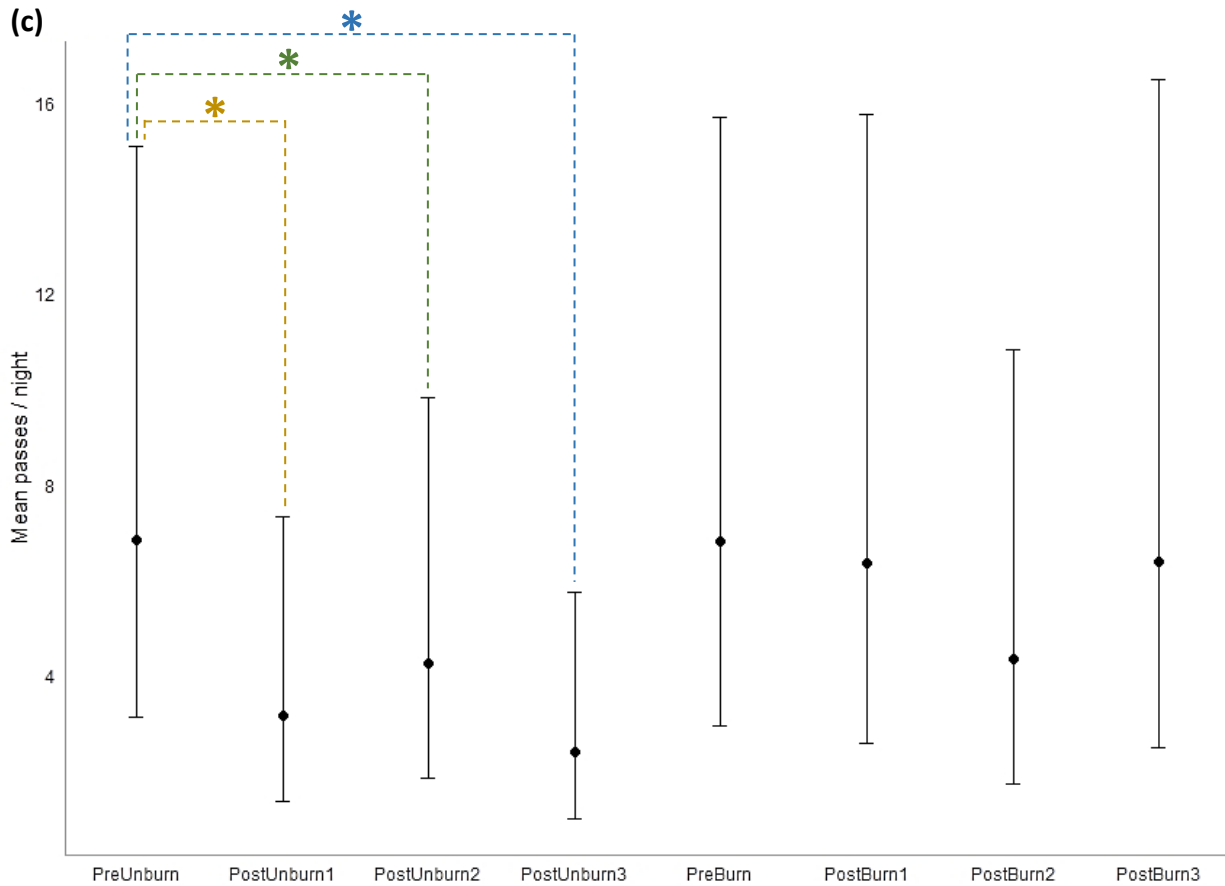
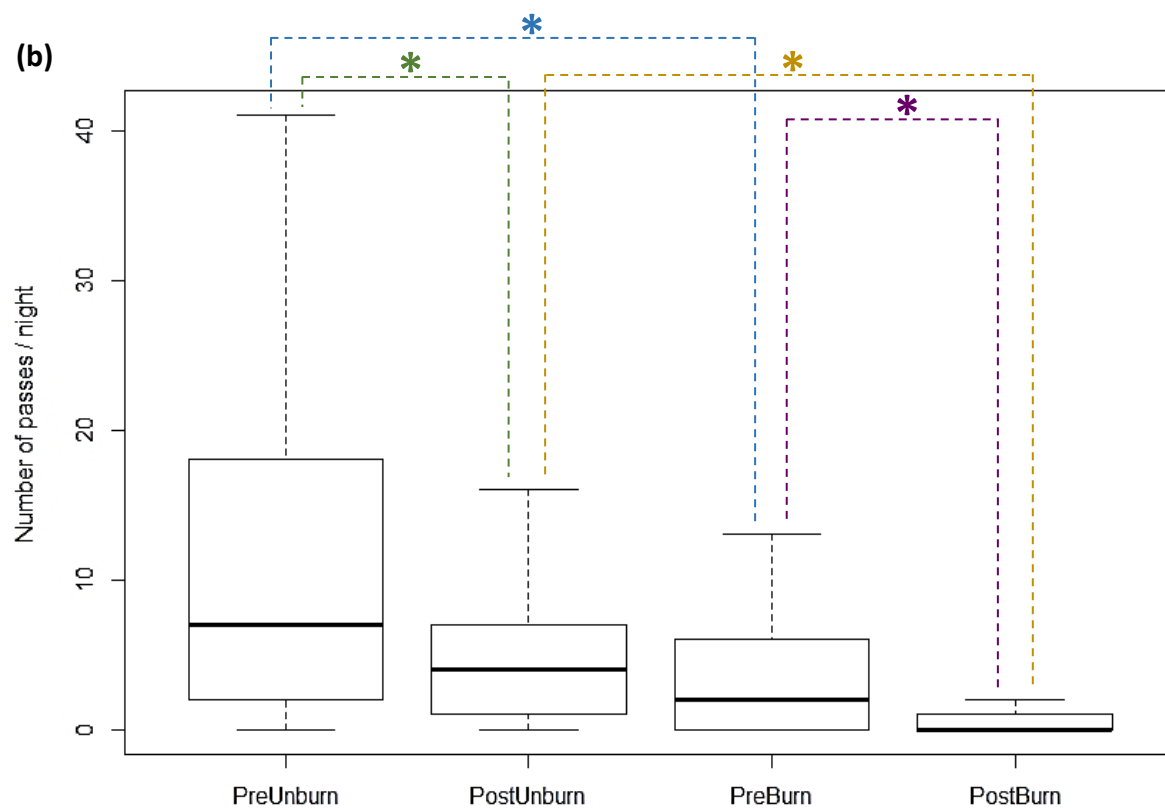
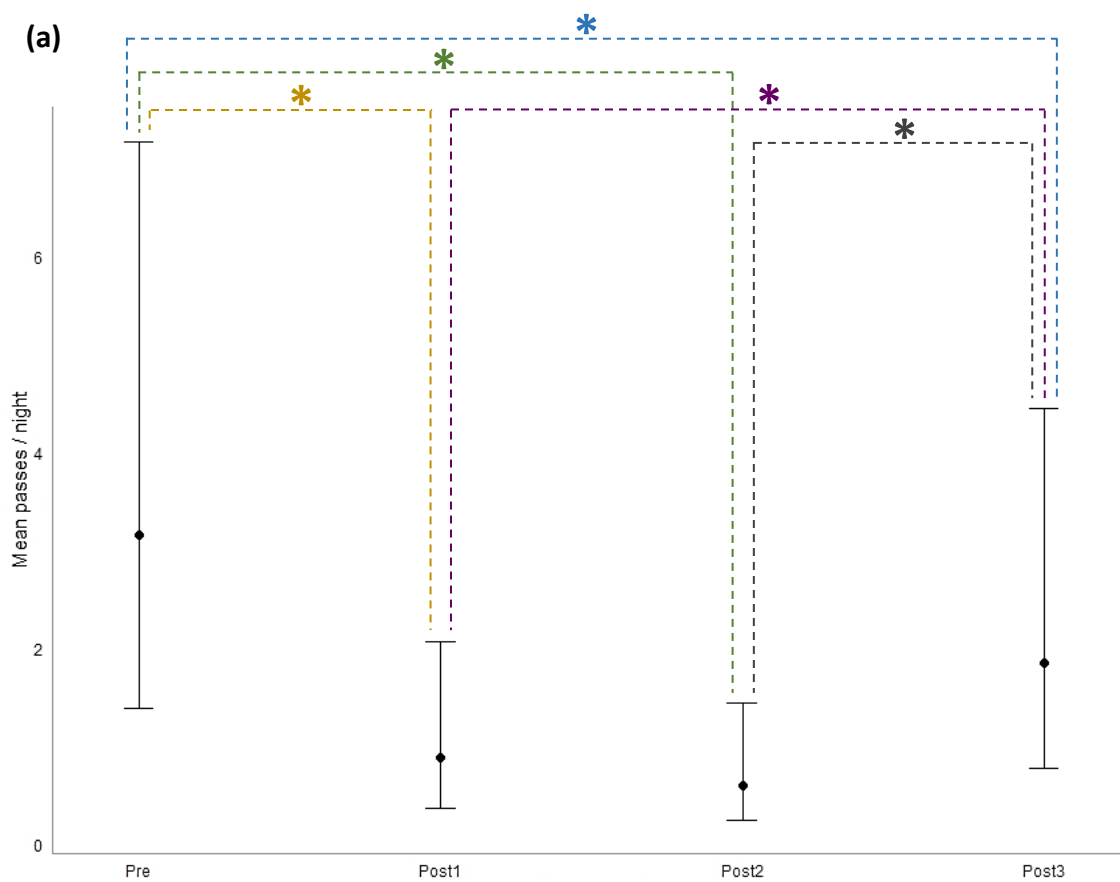


Fig. 3.7. The big brown/silver-haired bat nightly activity plotted for each fire metric. The mean passes/night predicted by the associated generalized linear mixed model (i.e. controlling for the effects of the covariates) for PrePost123, PrePostUnburnBurn, and PrePost123UnburnBurn are shown in (a), (b) and (c), respectively. The error bars in the prediction plots show the 95% confidence interval. The asterisks denote a significant difference ($P < 0.05$), with each colour indicating a separate comparison.



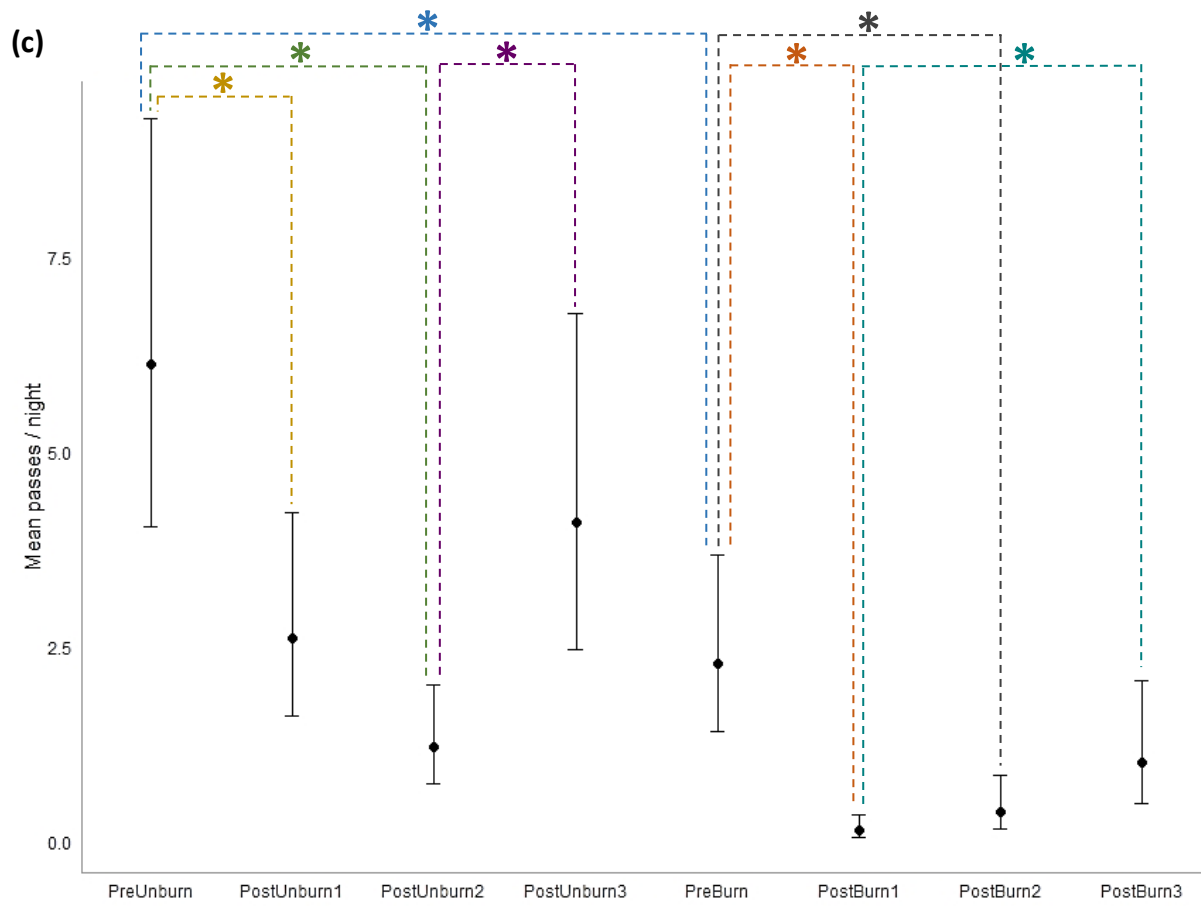
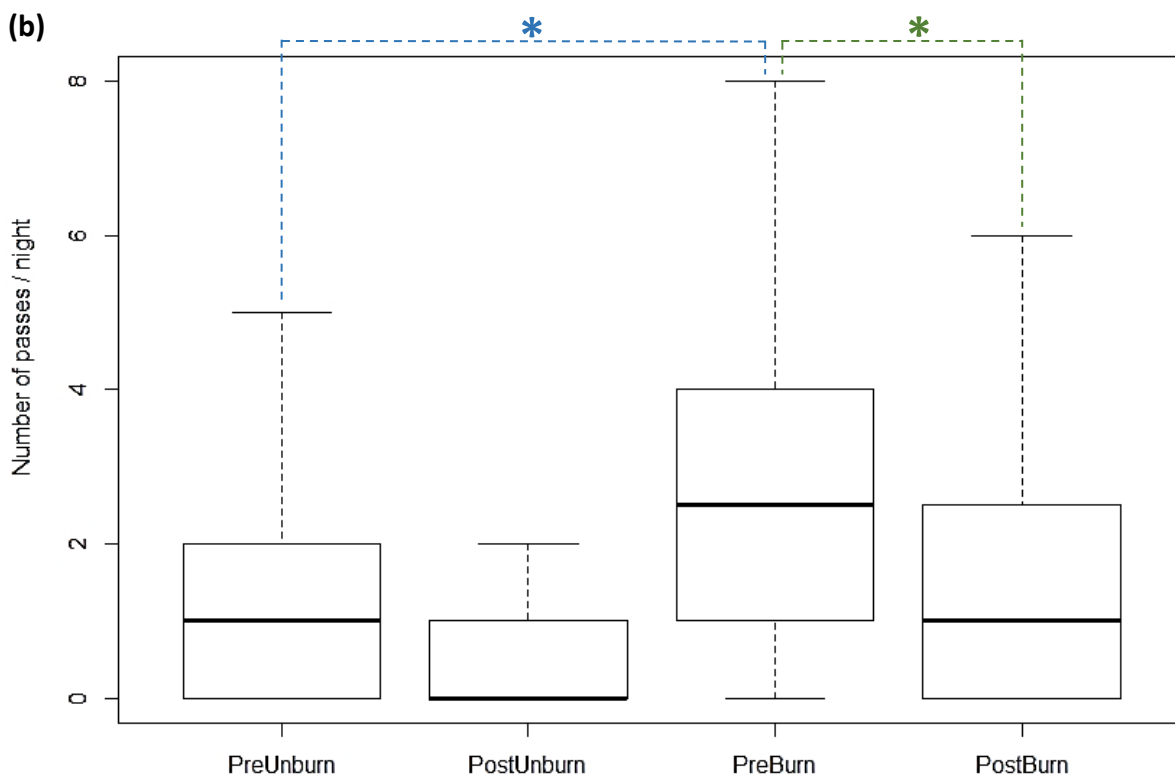
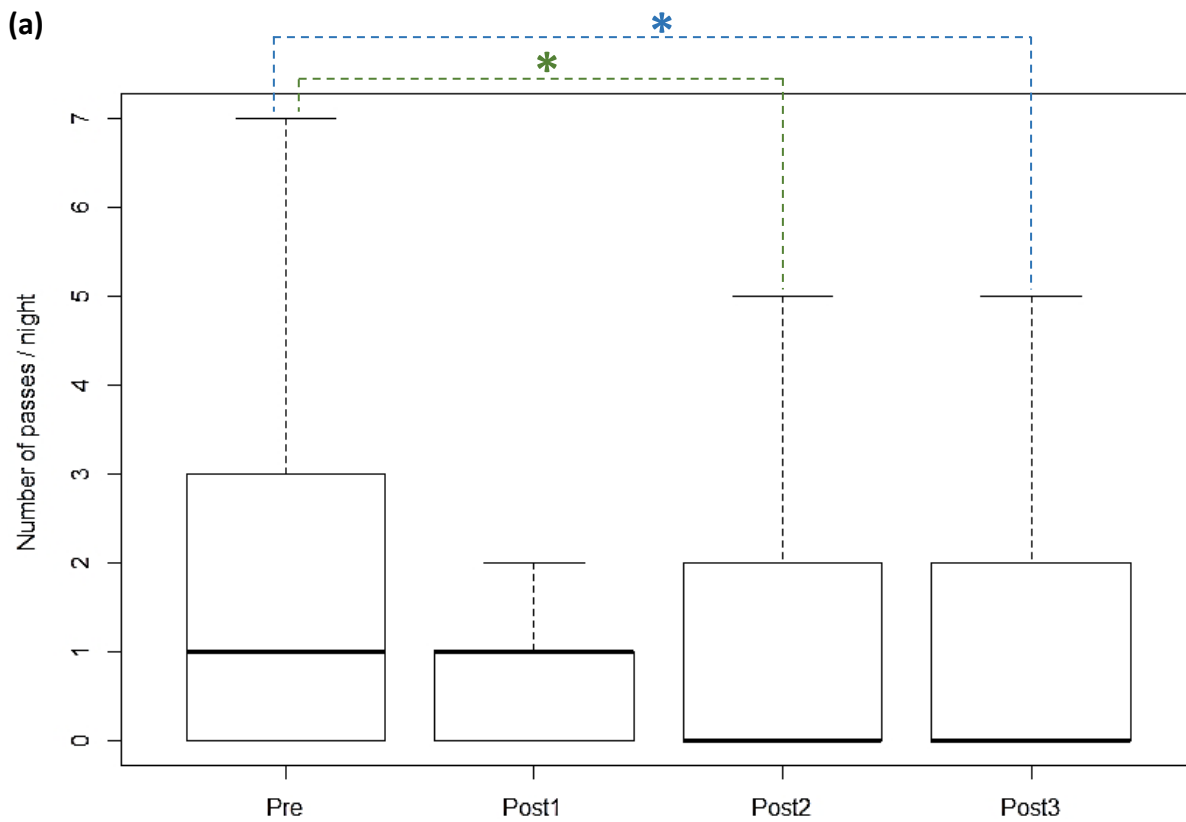


Fig. 3.8. The hoary bat nightly activity plotted for each fire metric. The mean passes/night predicted by the associated generalized linear mixed model (i.e. controlling for the effects of the covariates) for PrePost123 and PrePost123UnburnBurn are shown in (a) and (c), respectively, and the number of passes/night for PrePostUnburnBurn is shown in (b). The boxplot in (b) is based on the associated Kruskal-Wallis rank-sum test, and therefore does not control for the effects of any covariates. The outliers have been removed in the boxplot and the error bars in the prediction plots show the 95% confidence interval. The asterisks denote a significant difference ($P < 0.05$), with each colour indicating a separate comparison.



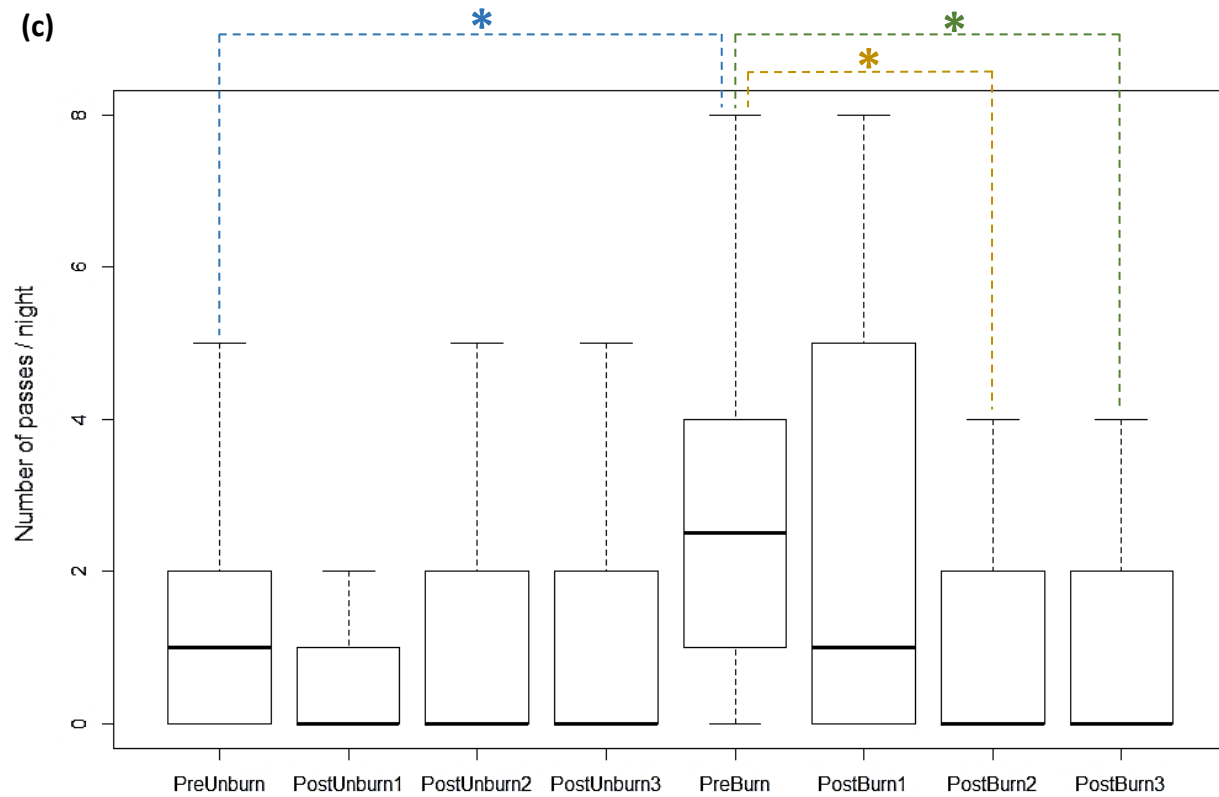
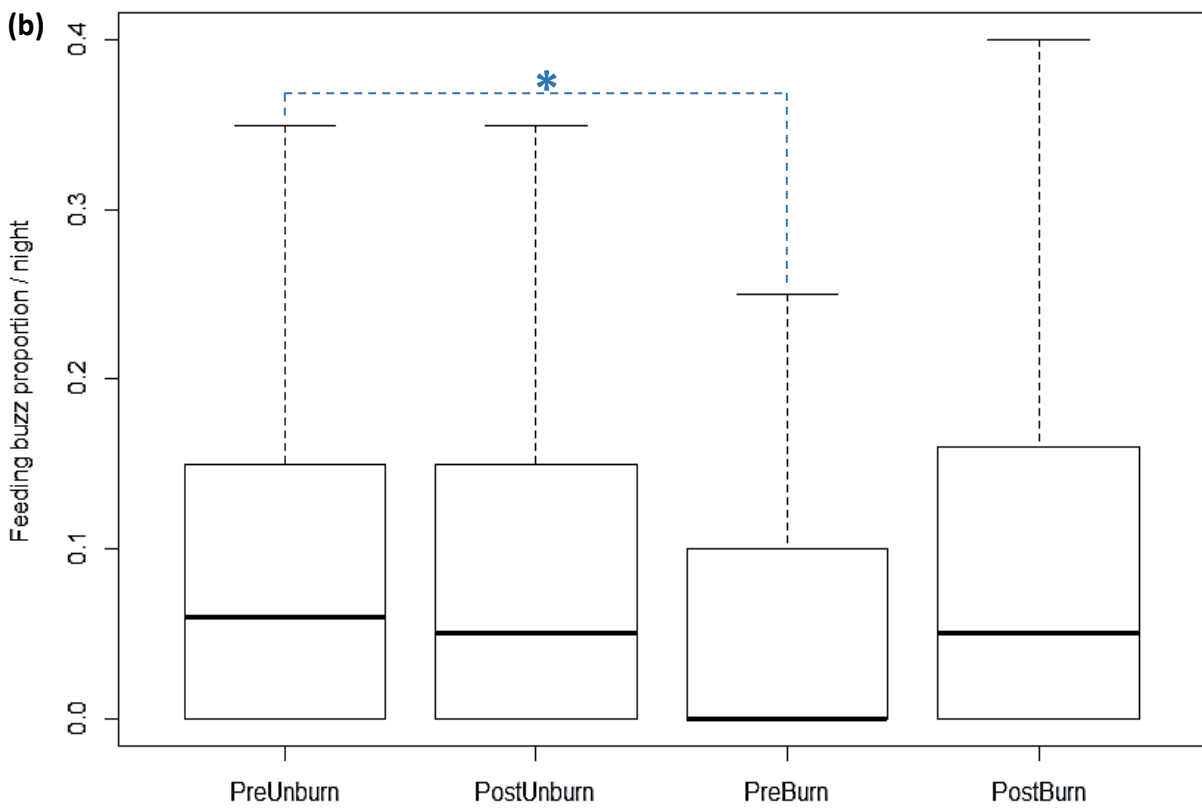
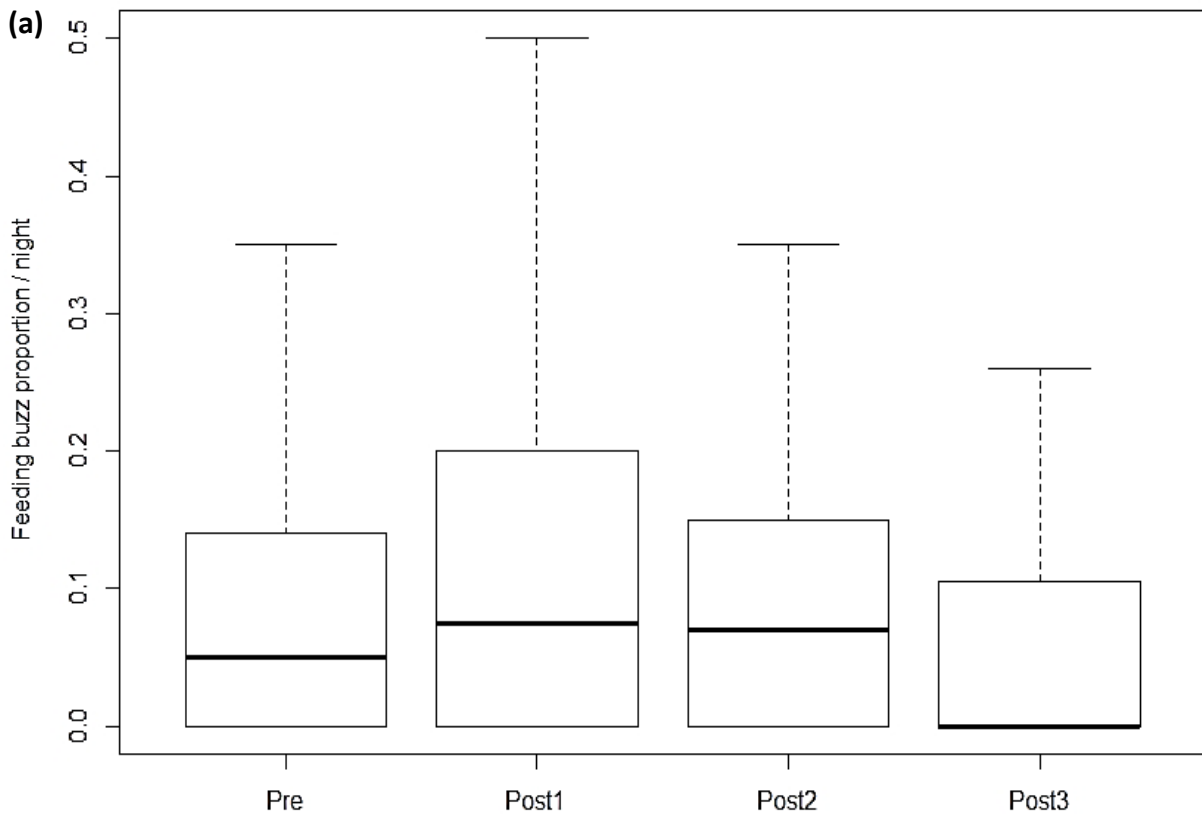


Fig. 3.9. The long-eared *Myotis* nightly activity plotted for each fire metric. The number of passes/night for PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn are shown in (a), (b) and (c), respectively. The boxplots are based on the associated Kruskal-Wallis rank-sum tests, and therefore do not control for the effects of any covariates. The outliers have been removed in the boxplots. The asterisks denote a significant difference ($P < 0.05$), with each colour indicating a separate comparison.



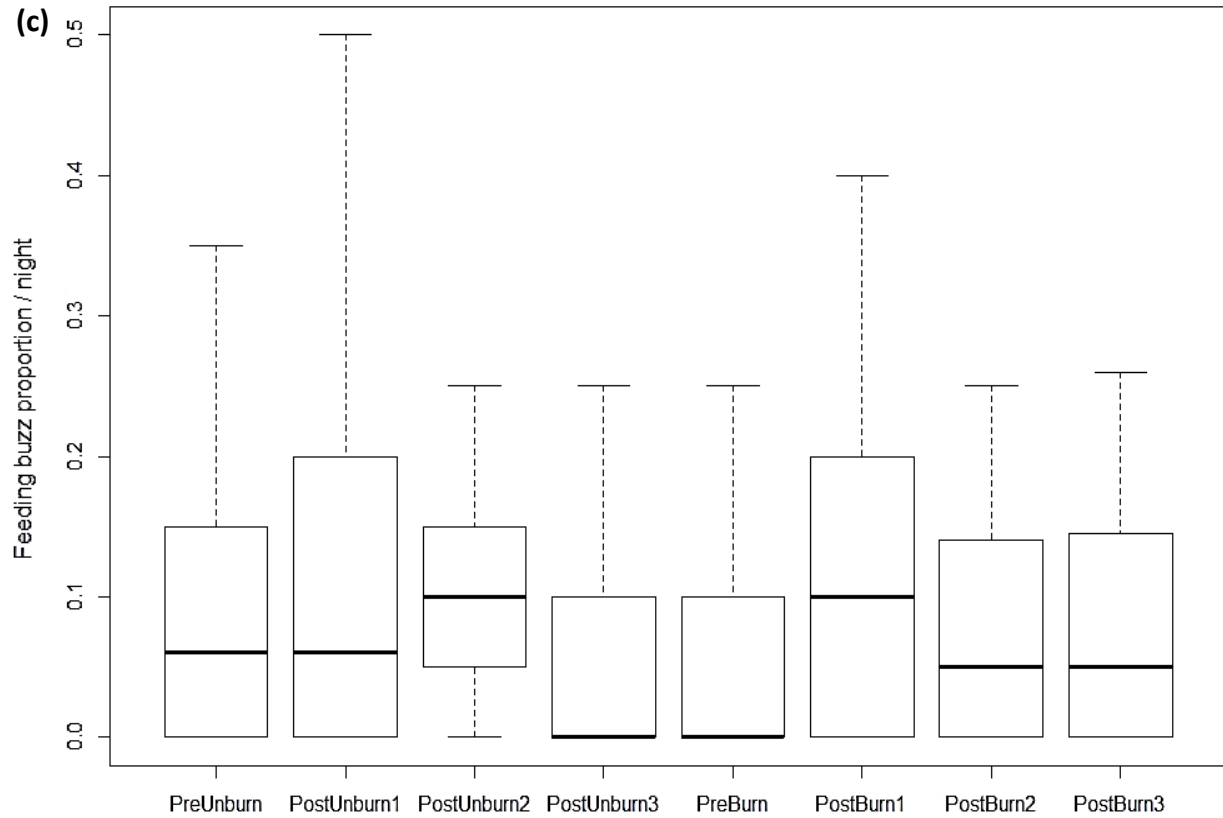


Fig. 3.10. The nightly proportion of feeding buzzes for 40kHz *Myotis* plotted for each fire metric. The feeding buzz proportion/night for PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn are shown in (a), (b) and (c), respectively. The boxplots are based on the associated Kruskal-Wallis rank-sum tests, and therefore do not control for the effects of any covariates. The outliers have been removed in the boxplots and the asterisk denotes a significant difference ($P < 0.05$).

3.4 Discussion

I predicted that bat activity would vary between pre- and post-fire years as well as between post-fire burned and unburned areas due to changes in forest structure, foraging opportunities, and roosting availability. Specifically, I predicted that post-fire years and burned areas would have greater bat activity compared to pre-fire years and unburned areas. My results demonstrated that species/species groups were affected differently by the fire, with some having increased abundance and others reduced abundance. I discuss the effect of the fire on each species/species group below, and the significant site and environmental covariates in Appendix N.

3.4.1 40 kHz *Myotis* activity and feeding

40 kHz *Myotis* made up the majority of acoustic detections during the six-year study (50.8% of overall activity, 95.4 ± 11.0 SE passes/night), highlighting the importance of this group in WLNP. Although I was not able to determine species-specific identifications for this group, capture records from 2011/2012 (Lausen 2012) and 2019/2020 (see Chapter 2) strongly suggest that most detections were of little brown *Myotis*. In 2011/2012, 95.4% of all 40 kHz *Myotis* (i.e. little brown *Myotis*, long-legged *Myotis*, and long-eared *Myotis*) captures were of little brown *Myotis* (Lausen 2012) and in 2019/2020, 97.9% were little brown *Myotis* (see Chapter 2).

As I predicted, 40 kHz *Myotis* activity increased in burned areas and decreased in unburned areas after the fire. As well, in the burned areas, activity was highest in 2018 (the year after the fire). However, contrary to my prediction, overall, there was no significant difference in mean activity per night between pre- and post-fire years. This suggests that while resident bats were moving from the unburned areas of the park to the burned areas of the park, the fire did not change survival rates of resident bats or promote immigration into or emigration out of the park. When examining 40 kHz *Myotis* feeding activity, I predicted that feeding activity would be highest in the burned areas after the fire, with the greatest number in 2018 and decreasing afterward. There was no significant difference between the percentage of 40 kHz *Myotis* echolocation calls that contained feeding buzzes between the burned areas before and after the fire, including in 2018. However, even though not significant, feeding activity did increase post-fire in the burned areas (nightly mean of 6.7 ± 1.2 SE % in burned

areas pre-fire versus 10.4 ± 1.7 SE % post-fire). In addition, as overall 40 kHz *Myotis* activity significantly increased in the burned areas post-fire, there was also a corresponding significant increase in the raw number of feeding buzzes in the burned areas after the fire (mean of 2.0 ± 0.4 SE buzzes/night) compared to before the fire (mean of 1.0 ± 0.2 SE buzzes/night; $W=3378.5$, $P= 0.023$). This suggests that the local movement of 40 kHz *Myotis* from unburned areas to burned areas could be due to increased foraging opportunities in the burned areas.

Although previous studies have emphasized the relationship between fire, reduced habitat clutter, and increased activity levels of edge and open-habitat specialist bat species, such as 40 kHz *Myotis* (e.g. Loeb and Waldrop 2008, Armitage and Ober 2012, Buchalski et al. 2013, Inkster-Draper et al. 2013, Cox et al. 2016, Blakey et al. 2019, Taillie et al. 2021), I argue that increased foraging opportunities were the predominant cause of the changes observed before and after the Kenow wildfire. Little brown *Myotis* are opportunistic, generalist, predators that are capable of taking prey on the wing and gleaning insects in a variety of habitats (Ratcliff and Dawson 2003, Clare et al. 2011, Kunz et al. 2011). Past research has emphasized that various species of bats likely capitalize on insect outbreaks (Brigham et al. 1992, Malison and Baxter 2010a, McCracken et al. 2012, Muller et al. 2012, Gonsalves et al. 2013); little brown *Myotis* may also exhibit this behaviour, congregating in areas with concentrated prey sources. Past studies have suggested that severe fires are associated with increased numbers of insects that are generalist primary-consumers, such as Chironomidae (Minshall et al. 1997, Minshall 2003, Malison and Baxter 2010b), a family of dipterans that are commonly preyed upon by little brown *Myotis* (Clare et al. 2014a). In addition, studies that have concurrently examined insect abundance and bat activity pre- and post-fire have also seen local movements of bats into areas experiencing increased foraging opportunities (Lacki et al. 2009, Malison and Baxter 2010a).

Although it has been suggested that roosting availability may also influence bat distribution (Lacki et al. 2017), I argue that potential changes in roosts were not a significant cause of the trends observed after the wildfire. While tracking little brown *Myotis* to their roost sites in 2019 and 2020, I examined the relative proportion of potential trees that had bark roosts between the burned and unburned areas (see Chapter 2). There was no significant

difference between the burned and unburned areas, suggesting that even though trees suitable for roosting were lost in the fire, roosting opportunities were created in comparable proportions.

While some studies found that the activity of edge-adapted species of bats often increases in habitats that have experienced high-severity fires, compared to areas that remained unburned (Malison and Baxter 2010a, Buchalski et al. 2013, Ancillotto et al. 2021), others have found the opposite (e.g. Jung 2020). One study found that little brown *Myotis* activity was lower in burned areas compared to unburned areas eight years after a severe wildfire (Jung 2020). Others have also found that as the number of years since fire increases, little brown *Myotis* occupancy probability decreases (Blakey et al. 2019). This suggests that as time since the Kenow wildfire increases, burned areas may be less selected by 40 kHz *Myotis*. As the burned areas regenerate, different areas may become more energetically favourable to forage in, causing bats to either undergo local redistribution within the park, or emigrate to new areas. In addition, roost trees may become limited in the park in the coming years (Jung 2020). In WLNP in 2019 and 2020, all of the little brown *Myotis* that I tracked to natural roosts were found roosting under loose bark (see Chapter 2). Bark roosts are temporary structures (Russo et al. 2005). I observed a bark roost used by a little brown *Myotis* in July, and by August of the same year, the piece of bark the bat was roosting under appeared to have fallen off the tree.

3.4.2 Big brown and silver-haired bat activity

The big brown/silver-haired bat species group made up 15.4% of overall activity, with a mean of 29.0 ± 2.8 SE passes/night. Big brown/silver-haired bat activity decreased after the fire. However, although activity levels were lower in both burned and unburned areas, the decrease was significant in the unburned sites only. Furthermore, post-fire unburned areas did not return to pre-fire activity levels in the three years monitored. These results were mostly contradictory to my prediction, as big brown/silver-haired bat activity levels were lower post-fire, and the possible local movements between burned and unburned areas after the fire were not clear.

Due to their similar echolocation characteristics, most acoustic studies have grouped big brown bats and silver-haired bats together (e.g. Cox et al. 2016, Austin et al. 2018a, Burns et al.

2019). The majority of past studies have found that fire typically has either a neutral or positive effect on big brown and silver-haired bats (Armitage and Ober 2012, Cox et al. 2016, Silvis et al. 2016, Austin et al. 2018b, 2020, Burns et al. 2019). In addition, two studies that examined big brown and silver-haired bat occupancy separately, found that occupancy probabilities of both species increased with burn severity (Steel et al. 2019) and decreased with canopy cover (Blakey et al. 2019). The positive relationship between fire and big brown/silver-haired bat activity has often been attributed to a less cluttered post-fire forest structure, thereby making foraging more efficient (Cox et al. 2016, Austin et al. 2018b, Blakey et al. 2019, Steel et al. 2019). Even though past studies have suggested that big brown/silver-haired bats respond neutrally or positively to fire, most of these studies were based on prescribed fires (e.g. Armitage and Ober 2012, Cox et al. 2016, Silvis et al. 2016, Austin et al. 2018a, Burns et al. 2019). The different results I found in big brown/silver-haired bat responses could be due to the severity of the Kenow wildfire and the corresponding level of ecological change that occurred post-fire.

Although it is unclear why there was a significant decrease in the unburned areas only after the fire in WLNP, there are several possible explanations. One is that the fire affected big brown bats and silver-haired bats differently. If one of the species responded positively to the fire, they may have moved from the unburned areas to the burned areas to capitalize on the increased resources potentially present. If the other species responded negatively, they may have emigrated out of the study area entirely. Thus, this would appear as a decrease in the unburned area (as both species left these habitats) and no change in the burned areas (as one species increased, and one species decreased). The 95% confidence intervals were also much larger for the post-burned sites compared to the post-unburned sites indicating that there was a large amount of variation in activity levels in the burned areas after the wildfire. Trapping surveys from 2011/2012 and 2019/2020 yielded few captures of big brown and silver-haired bats and therefore provide limited information on the relative proportions of big brown versus silver-haired bats in WLNP. In 2011/2012 2.2% of overall captures were big brown bats, and <1% were silver-haired bats (Lausen 2012). In 2019/2020 <1% of overall captures were big brown bats, and 2.7% were silver-haired bats (see Chapter 2).

Big brown and silver-haired bats have similar natural histories, with both species being considered edge-specialists (Jantzen and Fenton 2013), tree-roosters (Nagorsen and Brigham 1993), and predators of a variety of insect orders including Lepidoptera, Coleoptera, Diptera, Hemiptera, and Ephemeroptera (Carter et al. 2003, Reimer et al. 2010, Clare et al. 2014b). However, big brown bats feed extensively on Coleoptera (Hamilton and Barclay 1998) while silver-haired bats feed extensively on Lepidoptera (Black 1974). This suggests that if one species was negatively affected by the fire and one positively, it would likely be due to changes in prey availability. Although results vary, Lepidoptera typically responds negatively to fire, while Coleoptera increases after fire (Kral et al. 2017). This could suggest that the wildfire created optimal foraging habitats for big brown bats and forced silver-haired bats to use areas outside of WLNP. However, in a study assessing Coleoptera abundance after the Kenow wildfire, the families of Coleoptera examined did not increase post-fire (Langor 2019).

Carabid beetle abundance decreased significantly in both grassland and forest sites after the Kenow wildfire (Langor 2019). Wood-boring beetles (i.e. Cerambycidae) responded negatively to the fire, likely due to the severity of the fire causing a low source population and a deficit of available phloem in the burned trees (Langor 2019). Although the coleopterans in WLNP appeared to respond negatively immediately after the Kenow wildfire, it will likely take time for the insect assemblages to settle and show responses reflective of the renewed post-fire habitat (Langor 2019). In addition, grassland versus forest habitats respond differently to fire, with insect assemblages recovering more quickly in grassland sites (Langor 2019). An insect study was also conducted after the 1998 Sofa Mountain Burn wildfire in WLNP (Kinsella 2003). Carabid beetle richness and diversity was highest in burned sites two to three years after the wildfire. However, their abundance did not differ between burned and unburned areas (Kinsella 2003). While Kinsella (2003) examined burned and unburned sites post-fire and Langor (2019) observed pre- and post-fire trends, these studies highlight that the response of coleopterans to wildfire varies and is likely dependent on the habitat type and number of years since the fire. While both Carabidae and Cerambycidae are preyed on by big brown bats (Clare et al. 2014b), it is possible that other families of beetles, not surveyed in WLNP, increased after the Kenow wildfire and were fed on by big brown bats.

An alternative explanation to the activity patterns I observed for big brown/silver-haired bats (i.e. significant decrease in the unburned areas only) could be the effect of when the burned sites versus the unburned sites were monitored for bats. By chance, burned sites were monitored earlier in the season (approximately late June to mid July) and the unburned sites were monitored later (approximately early July to early August). While big brown bats hibernate in caves, mines or buildings during the winter, silver-haired bats migrate to their overwintering locations in the United States (Nagorsen and Brigham 1993). Previous acoustic studies conducted near WLNP suggest that the migratory period in southwestern Alberta begins in mid to late July for silver-haired bats (Baerwald and Barclay 2011). It is possible that the start of the migratory period was captured for the unburned sites only due to the timing of the acoustic surveys, and the migrating silver-haired bats either chose alternative migratory routes or spent less time in WLNP while flying through on their way south.

3.4.3 Hoary bat activity

Compared to the other species/species groups, hoary bats made up a small percentage of the WLNP bat community (4.2% of overall activity, mean of 7.9 ± 1.0 SE passes/night). Hoary bat activity was strongly negatively affected by the fire in all years, in both unburned and burned areas. However, activity levels increased significantly in the third year after the fire, compared to the first two post-fire years. Although there was an increase in activity in early July (corresponding to when the site with the “river” as the nearest water feature was monitored), activity was also higher at the end of July in most years, coinciding with the start of the migratory period in southwestern Alberta (Baerwald and Barclay 2011). It is therefore difficult to conclude if the fire reduced the survival of resident bats, forced the species to emigrate to areas outside the park, or caused them to choose alternative migratory routes. The significant increase by the third year, however, suggests that while the immediate effects of fire may originally be negative, hoary bats are resilient to wildfire.

Past research on the effect of fire on hoary bats has been largely opposite to my results in WLNP. Studies have found that fire generally has a neutral or positive effect on hoary bat activity as the fire-altered habitat becomes less cluttered with better foraging conditions (Cox et al. 2016, Austin et al. 2018a, 2020, Blakey et al. 2019, Burns et al. 2019, Steel et al. 2019). As

the Kenow wildfire reduced the vegetative complexity in the burned areas (i.e. caused forest structure changes that are conducive for less-maneuverable bat species such as the hoary bat), I argue that the significant decrease in activity levels were due to decreased foraging opportunities. Although hoary bats are opportunistic foragers, they feed extensively on Lepidoptera (Carter et al. 2003, Reimer et al. 2010, Valdez and Cryan 2013). Although past studies have been contradictory on the effect of fire on Lepidoptera, many have found that they are negatively affected (Reed 1997, Huebschman and Bragg 2000, Powell et al. 2007, Swengel and Swengel 2007, Vogel et al. 2010, Elia et al. 2012, Kral et al. 2017, Carbone et al. 2019). Overall, the different lepidopteran families may respond differently, and the severity and amount of time since the fire may also influence abundance (Perry 2012, Kral et al. 2017).

Although I did not quantify abundance of larval-stage bedstraw hawkmoths (*Hyles gallii*) during fieldwork surveys completed in the second and third year after the Kenow wildfire, I observed large numbers of these insects in the burned areas. The bedstraw hawkmoth larvae were strongly associated with fireweed (*Chamaenerion angustifolium*) (pers. obs.), a plant that responds positively to fire (Bartos and Mueggler 1981, Klimesova et al. 2009). Moths in the family Sphingidae (e.g. bedstraw hawkmoths) overwinter underground as pupae, emerging the following summer (Shalaway 2004). Although to my knowledge, there have been no studies looking specifically at whether hoary bats consume Sphingidae, this family is present in the diet of a number of other bat species (Aguiar and Antonini 2008, Bohmann et al. 2011, Clare et al. 2011, Lacki and Dodd 2011), including one in the same genus (Clare et al. 2009). Due to the severity of the Kenow wildfire, much of the ground vegetation, duff layer and coarse woody debris (all of which provide insect habitat) were consumed by the fire in the burned areas (Langor 2019). I suggest that the Kenow wildfire caused significant lepidopteran larval and pupal mortality, requiring the recolonization of insects and the reestablishment of the plant community. As plant availability and abundance is crucial for insect populations (Blanco and Garrie 2020), and plant regrowth increased each year (e.g. fireweed), there was likely a corresponding increase in insect abundance. The significant increase in hoary bat activity levels by the third year after the fire might be explained by this.

Although the fire consumed the foliage of trees (i.e. potential roosting sites for hoary bats), over 60% of the park did not burn (Parks Canada 2021a). WLNP encompasses a small area (i.e. 505 km²) (Parks Canada 2010), with hoary bats flying at a mean speed of 7.7 m/s (or 27.7 km/hr) (Salcedo et al. 1995). As such, I suggest that changes in roost availability was not responsible for the decline in activity after the fire, as hoary bats are highly mobile and capable of covering large distances each night (Morningstar and Sandilands 2019).

WLNP is located south of Pincher Creek, Alberta, an area with considerable wind-turbine development (van Kooten et al. 2016). Although wind turbines are causing significant mortality to hoary bats (Frick et al. 2017), I am confident that the decrease in activity levels after the fire are predominantly due to the Kenow wildfire. The mean nightly counts ranged from 10.9 (\pm 2.7 SE) to 12.7 (\pm 3.2 SE) for 2015 to 2017 (pre-fire), and 2.0 (\pm 0.3 SE) to 6.5 (\pm 2.1 SE) for 2018 to 2020 (post-fire). This suggests that even though there was variability in activity levels before the fire, there was a significant disturbance after 2017 (i.e. Kenow wildfire) (Fig. 3.11).

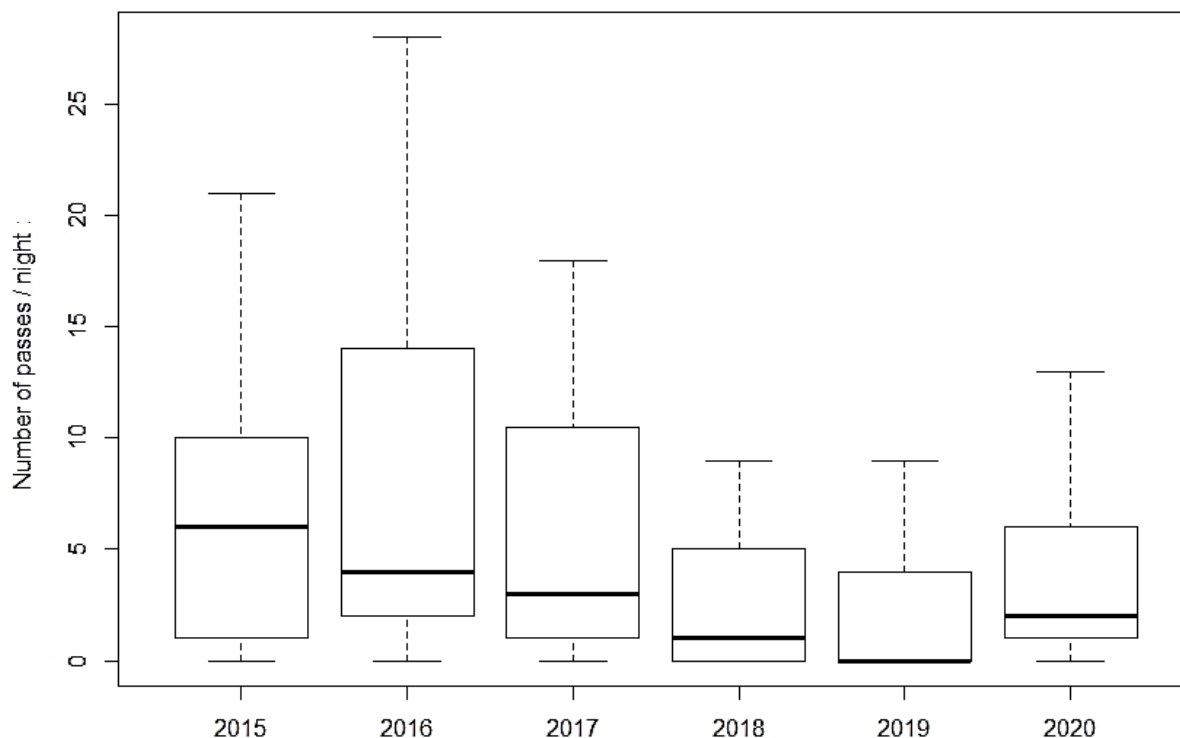


Fig. 3.11. Hoary bat nightly activity plotted for each year of the acoustic study in Waterton Lakes National Park, Alberta. Years 2015 to 2017 are pre-fire, and 2018 to 2020 are post-fire. The outliers have been removed.

3.4.4 Long-eared *Myotis* activity

My results suggest that the fire affected long-eared *Myotis* differently than the 40 kHz *Myotis* group (predominately little brown *Myotis*). Long-eared *Myotis* made up a small percentage of overall activity (1.4%), with few detections each night (mean 2.7 ± 0.4 SE passes/night). Unlike 40 kHz *Myotis*, long-eared *Myotis* activity decreased post-fire, primarily in the burned areas in the second and third years after the fire. As long-eared *Myotis* activity levels decreased after the fire and there was no indication of local movements between burned and unburned areas, it suggests that the fire either reduced survival rates or forced individuals to emigrate to areas outside the park.

The results from past research on the effects of fire on species adapted to cluttered environments, such as long-eared *Myotis*, have varied. Similar to my findings, some studies found that clutter specialists are negatively affected by fire (Armitage and Ober 2012, Blakey et al. 2019, Ancillotto et al. 2021). However, fire has also had neutral or positive effects on clutter-adapted species (Buchalski et al. 2013, Lacki et al. 2017, Austin et al. 2018b). Radio-tracking studies examining long-eared *Myotis* roosting sites and insect abundance between burned and unburned areas have also found contrasting results. One study found that in the first and second years after a fire, females roosted in rocks and trees equally, predominantly in areas that burned (Schwab 2006). While Diptera, Coleoptera and Trichoptera increased in abundance in the burned areas post-fire, Lepidoptera did not change (Schwab 2006). Contrary to this, Snider et al. (2013) found that female long-eared *Myotis* roosted predominantly in rock roosts in unburned areas. In the same area, insects were more abundant in the unburned areas compared to the burned areas in the four to five years after the fire (Snider 2009). Whether these differences are due to site-specific variations or represent different time points in the post-fire continuum, is difficult to determine. Overall, both studies found that long-eared *Myotis* roost sites are likely not limited in a burned/unburned landscape, suggesting that the decreases in long-eared *Myotis* activity I found were the result of something other than changing roost availability. While long-eared *Myotis* are capable of gleaning as well as aerial hawking (Faure and Barclay 1994), they feed extensively on Lepidoptera (Barclay 1991). Although past studies have been contradictory on the effect of fire on Lepidoptera, many have

found that they are negatively affected (Reed 1997, Huebschman and Bragg 2000, Powell et al. 2007, Swengel and Swengel 2007, Vogel et al. 2010, Elia et al. 2012, Kral et al. 2017, Carbone et al. 2019). Therefore, opportunities to forage on Lepidoptera may have become limited after the Kenow wildfire, resulting in long-eared *Myotis* relocating to other areas.

3.4.5 Study assumptions and limitations

My study had a number of assumptions and potential limitations. For example, it is unknown whether the detectability of bats changed pre- versus post-wildfire due to changes in the forest structure. Some studies have reported that structural clutter has a minimal effect on the detectability of bat echolocation calls (Yates and Muzika 2006, Obrist et al. 2011) and is therefore assumed to be of minor concern (Titchenell et al. 2011, Cox et al. 2016). However, others have found that the probability of detection is greater for high frequency bats in sites with reduced vegetation density (Burns et al. 2019, Bender et al. 2021) and/or low frequency bats (Patriquin et al. 2003, Burns et al. 2019, Steel et al. 2019). Although I did not include detection probabilities in my models, I do not believe that post-fire forest structural changes would have significantly changed detectability. As most acoustic survey locations were situated near water features, the area directly surrounding the water feature, and thus the microphone, were often unburned even in the sites that were classified as burned. The microphones were also raised five to six meters off the ground, thereby reducing the potential influence any changes in understory height may have had on the detectability of bats. Although one of the species groups had results that would be consistent with detectability increasing in areas with reduced clutter (i.e. 40 kHz *Myotis* activity and feeding buzzes), it was not ubiquitous across all species/species groups. Big brown/silver-haired bat activity did not change significantly pre- to post-fire in the burned areas, and hoary bats and long-eared *Myotis* activity decreased in burned areas post-fire. However, if the fire caused an increase in the detection range in the burned areas, the effect of the fire may have been even greater than I have stated for some of the species/species groups.

An assumption of my study was that activity levels did not vary over the summer season. Survey dates were not completely consistent across years, due to equipment failures and detectors needing to be redeployed. As such, some sites were monitored on slightly

different dates each year. There was also likely variation in seasonal activity from year to year based on the annual weather (e.g. cold, wet spring, dry, hot summer), thus affecting the timing of both volancy and migration. Based on capture surveys conducted in 2019 and 2020, the majority of the bats that raise offspring in WLNP are little brown *Myotis*. No pregnant or lactating hoary, silver-haired or big brown bats were ever caught. Little brown *Myotis* pups are born in early to mid July (first lactating female was captured on July 14) and are volant by early to mid August (first juvenile was captured on August 9), with maternity colonies likely exclusively in buildings in the Waterton townsite (see Chapter 2). Only one site was located near to the Waterton townsite (i.e. within two kms), and it was always monitored relatively early in the survey period (early to mid July), before pups would be flying. The sites that were monitored in late July/early August were far from the Waterton townsite (mean distance of 13.1 ± 1.9 SE kms), arguably farther than newly volant pups would be capable of flying. Past research has found that little brown *Myotis* swarming events (i.e. when bats congregate at hibernacula to mate) occur in mid to late August throughout Canada (Schowalter 1980, McGuire et al. 2009, Burns et al. 2014, Gallant and Broders 2015). This would be consistent with my capture records from 2019 and 2020, indicating that males in WLNP are not reproductive until late August, well after the end of the acoustic detector deployment period.

Past capture and acoustic surveys in WLNP suggest that the park is likely on a migratory route for hoary, eastern red, and silver-haired bats (Lausen 2012). Consistent with other acoustic work conducted in southwestern Alberta (e.g. Baerwald and Barclay 2011), Lausen (2012) found that migratory bat activity increased in late July in WLNP. Although both hoary bat and big brown/silver-haired bat activity increased each year at the end of July, I do not believe that including the beginning of the migratory period would have caused misleading results for these species/species groups. The increase in activity levels of hoary bats and big brown/silver-haired bats was associated with two sites with particularly high activity, Lost Lake and Lone Lake. These sites were always monitored in late July (ranging from July 22 to August 2), and as such the beginning of the migratory period was captured consistently across the six years. Eastern red bats were not included in any of the analyses due to the low number of detections throughout the study (0.4 ± 0.1 SE passes/night).

There was also a difference in when burned versus unburned sites were monitored. This was not planned; survey dates were chosen at random prior to the wildfire. Detectors were deployed at what were eventually burned sites, earlier in the season (approximately late June to mid July), with unburned sites being monitored later (approximately early July to early August). Although the unburned sites may have captured possible activity changes that were missed by the burned sites (i.e. migration or volancy), I do not believe this would have caused biased results. The design of the study prioritized having the same sites monitored on approximately the same dates each year. Thus, even if volancy, swarming or migration did occur within the unburned sites' deployment period, the effects would have been consistently captured each year.

An additional limitation of my study was that to increase efficiency and consistency when I was manually analyzing the acoustic files, I used auto-identification software and analyzed the files in zero-cross file format rather than in full spectrum. I used the noise scrubber and species' classifiers in Kaleidoscope Pro, as well as filters in AnalookW as two methods of auto-identification prior to manual analysis. AnalookW is only able to read zero-cross file formats, requiring all full spectrum files to be converted to zero-cross files prior to analysis. As the same software settings and workflow were used for each year of the acoustic data, I do not believe that analyzing the zero-cross files instead of the full spectrum files and using the Kaleidoscope noise scrubber would have led to misleading results as each year would have been consistently biased. However, based on the subsample of files I examined for missed species, I suspect that hoary bat activity levels were underestimated across all years, and Lost Lake and Lone Lake had higher activity levels than was reported throughout all years of the study (Appendix G).

The final limitation was that even though GLMMs converged for most species/species groups, I was unable to fit a model for the hoary bat PrePostUnburnBurn fire metric, and all fire metrics (i.e. PrePost123, PrePostUnburnBurn, and PrePost123UnburnBurn) for long-eared *Myotis* and 40 kHz *Myotis* feeding buzzes. For the Kruskal-Wallis rank-sum tests, I only included the effect of the fire metrics on activity levels/feeding buzz proportion, and was unable to include other possible covariates (i.e. sunset temperature, total nightly precipitation, nearest

water feature, distance to the townsite, site). In addition, I did not include all the 40 kHz *Myotis* data for the 40 kHz *Myotis* feeding buzz analyses, but subsampled. It is possible that with a bigger sample size, a GLMM could be created for 40 kHz *Myotis* feeding buzzes, and perhaps provide stronger support for the conclusion that 40 kHz *Myotis* feeding activity increased in the burned areas post-fire.

3.4.6 Future research

Given that time since the Kenow wildfire influenced the activity levels for some of the bat species/species groups, acoustic monitoring of the same study sites should continue in the future. This would help determine whether the patterns I observed in the first three years after the wildfire, are indicative of ecosystems still in a state of flux, or habitats representative of the post-fire landscape. As time since the fire increases, the forest will become more dense with new plant growth (Perry 2012), different insect species will likely recolonize the area (Kral et al. 2017), and roost trees created by the fire may no longer be standing due to their fire-weakened bases (Morrison and Raphael 1993).

Bat trapping surveys should continue in WLNP, focused primarily on conducting ongoing general species inventories rather than targeting a specific species as I did in 2019/2020. It will be important to conduct the surveys in a variety of habitats, place mist nets at different heights, and conduct these surveys throughout the summer, to ensure any spatial or temporal trends are captured. In addition to collecting information such as sex, age and reproductive condition of the bats, the trapping surveys may also help inform acoustic survey results. Due to the overlap in echolocation characteristics between species such as big brown and silver-haired bats (Betts 1998), capture rates could elucidate how common each species is in the different areas of the park. This could indicate whether the big brown/silver-haired bat group represents both species equally, or if one species is more common than the other.

Future research should also focus on insect sampling. These surveys should be conducted throughout various post-fire successional stages and in different habitat types (e.g. unburned and burned sites in forested and grassland areas). The types of insect sampling methods used should be ones that accurately reflect the species available to bats (i.e. nocturnal flying insects) (Perry 2012), such as using black-light traps (Agosta et al. 2003, Johnson et al.

2007, Dodd et al. 2012). In addition, if bat capture surveys are conducted, guano samples should be collected from the bats for diet analysis. Researchers can then examine what species of insects bats are feeding on in relation to what is available and abundant in the different burned and unburned habitats.

3.4.7 Management recommendations

Wildfires were frequent in WLNP prior to the early 1900s, promoting complex multi-aged forest stands and landscape diversity (Barrett 1996). However, active fire exclusion efforts for approximately 50 years shifted the WLNP forest communities to high density, single-aged forests with uniform canopies and high fuel loads (Barrett 1996). Due to the severity of the Kenow wildfire, it likely caused the forests affected by the fire to transition from an unburned monoculture to a burned monoculture. However, mixed-severity edges were created and over 60% of the park remained unburned (Parks Canada 2021a), thus creating habitat diversification.

The Kenow wildfire removed much of the accumulated fuel load and provided the necessary catalyst for WLNP to return to a structurally diverse and complex landscape. Moving forward, there are now opportunities to use prescribed burning throughout the different areas of the park, with fewer concerns of the fire becoming out of control. Although WLNP has been using prescribed burning since the 1990s, most of these fires occurred in the Foothills Parkland subregion, with few in the Montane and Subalpine subregions (GOC 2019). Future fire management should focus on striving to mimic how the natural fire cycle would have looked in a pre-fire-suppression era. All subregions should be considered for prescribed burning, to ensure that all forest communities have a variety of age-classes and habitat diversity. Regular prescribed burning would also decrease the chances of WLNP experiencing large, extreme severity wildfires in the future, and instead promote lower intensity burning in small habitat patches.

Overall, I believe that the severity of the Kenow wildfire was a consequence of a landscape with an extended fire-suppressed history. The wildfire had immediate positive effects on 40 kHz *Myotis*, and I believe once the plant community is re-established, and insects recolonize the area, the remaining bat species in the WLNP bat community will benefit from the post-fire changes. Fire management decisions should strive to conduct regular prescribed

burning throughout all regions of the park to ensure that there is a mosaic of habitat types suitable for the WLNP bat community.

3.5 Conclusion

Previous studies have found that fire typically has an overall neutral or positive effect on most bat species (e.g. Buchalski et al. 2013, Steel et al. 2019, Ancillotto et. al 2021, Blakey et al. 2021, Taillie et al. 2021). However, apart from 40 kHz *Myotis*, activity levels of bat species/species groups (i.e. big brown/silver-haired bat, hoary bat, and long-eared *Myotis*) examined in this study declined post-fire. I argue that the differences between my results and past research are primarily due to the severity of the Kenow wildfire and its immediate negative impact on the insect community. The Kenow wildfire consumed much of the insect habitat (i.e. ground vegetation, duff layer and coarse woody debris; Langor 2019), and likely caused high mortality of insect larvae and pupae. As such it will take time for the reestablishment of the plant community and the corresponding recolonization of insects.

Studies have emphasized the importance of mixed severity fires as well as unburned refugia to promote landscape heterogeneity, thus ensuring that there is suitable habitat available to all species present in a bat community (Armitage and Ober 2012, Buchalski et al. 2013, Steel et al. 2019, Ancillotto et. al 2021, Blakey et al. 2021, Taillie et al. 2021). In my study, little brown *Myotis* behaved opportunistically and were likely able to adapt immediately to the changes in the forest structure, foraging opportunities and roost availability (see Chapter 2) after the fire. Species such as big brown bats, silver-haired bats, hoary bats and long-eared *Myotis* were likely less adaptable to the significant ecological changes caused by the severe wildfire. However, acoustic monitoring only captured the first three years post-wildfire. As has already been observed with hoary bat activity levels increasing by the third year after the fire, bats show resilience to wildfire (Buchalski et al. 2013, Steel et al. 2019, Ancillotto et. al 2021, Blakey et al. 2021) and it may take time for the positive effects of the Kenow wildfire to be seen in the bat community as a whole.

Chapter 4: General Conclusions

Studies have found that fire (both prescribed and wildfire) generally have an overall neutral or positive effect on most bat species (e.g. Malison and Baxter 2010a, Armitage and Ober 2012, Buchalski et al. 2013, Austin et al. 2020, Blakey et al. 2021). However, species-specific negative responses have also been reported (e.g. Blakey et al. 2019, Jung 2020, Starbuck et al. 2020). Fire impacts insectivorous bat communities by changing the foraging habitat, insect communities (i.e. prey), and roosting opportunities (Lacki et al. 2009, 2017, Armitage and Ober 2012, Buchalski et al. 2013). I hypothesized that the Kenow wildfire affected species' distribution and abundance of bats in Waterton Lakes National Park (WLNP) due to changes in the forest structure, foraging opportunities, and roost availability.

In Chapter 2, I examined differences in little brown *Myotis* (*Myotis lucifugus*) roost availability between burned and unburned areas of WLNP. I predicted that between burned and unburned sites, burned areas would have more confirmed and potential roost trees compared to unburned areas. In 2019 and 2020, I radio-tracked female and male little brown *Myotis* to their roost sites. The maternity colonies of reproductive females were found exclusively in buildings in the Waterton townsite, highlighting the importance of building roosts in the mountains. Males and non-reproductive females did not preferentially select either burned or unburned roosts, but rather used the habitat that was close in proximity. Bats used a variety of natural roosts in WLNP including bark roosts, rock crevices and boulders in both burned and unburned areas. Overall, my results suggest that the wildfire did not change the availability of roosts for little brown *Myotis*.

In Chapter 3, I examined bat activity differences between pre- (2015 to 2017) and post- (2018 to 2020) wildfire years, as well as between burned and unburned areas. Bat activity in WLNP was examined using acoustic detectors to record bat echolocation calls at thirteen sites within the park. The Kenow wildfire negatively affected big brown/silver-haired bat (*Eptesicus fuscus*/*Lasiurus noctivagans*), hoary bat (*Lasiurus cinereus*), and long-eared *Myotis* (*Myotis evotis*). Although activity levels for big brown/silver-haired bat, and long-eared *Myotis* never returned to pre-fire levels, hoary bat activity increased by the third year after the fire. 40 kHz *Myotis* activity, likely predominately little brown *Myotis*, did not change pre- to post-fire.

However, activity increased in burned areas and decreased in unburned areas which could have been due to increased foraging opportunities in the burned areas. My findings emphasized species-specific responses, and only captured the first three years after the wildfire. The intermediate and long-term effects may be different.

Having both pre- and post-fire bat acoustic data provided the unique opportunity to examine the effects of a severe, late season, wildfire on bat populations in the Rocky Mountains. Furthermore, my study is the first to examine the effect of a wildfire on bat populations in the Canadian Rockies. The most common species in WLNP, little brown *Myotis*, is listed as “May Be at Risk” provincially (AEP 2020) and “Endangered” federally (GOC 2021). Provincially long-eared *Myotis*, silver-haired bat and hoary bat are listed as “Sensitive” (AEP 2020). Understanding how wildfire influenced abundance, and the community structure in WLNP, contributes to baseline data for bats in Alberta.

While conducting an assessment of the burn history in WLNP, Barrett (1996) predicted that “given [the mid 1990s’] relatively old and more-uniform forest mosaic, the large pine beetle epidemic, and recurrent droughts, major stand replacement burning may be imminent for WLNP”. In line with Barrett’s (1996) predictions, the high severity Kenow wildfire occurred two decades later. Although the Kenow wildfire was likely a result of a number of factors, climate change is directly impacting fire regimes, including fire frequency, size and severity (Flannigan et al. 2000).

As the number, duration and severity of wildfires are predicted to increase in the coming years due to climate change (Gillett et al. 2004, Westerling et al. 2006, Rocca et al. 2014, Stephens et al. 2014), it is becoming increasingly important to understand the effect of fire on all organisms native to an area, including bats. By gaining a more holistic understanding of the effects of wildfire on bats, the findings can be incorporated into fire management plans and inform future research studies. Fire management decisions should strive to conduct regular prescribed burning to ensure that there is a mosaic of habitat types. Habitat complexity and heterogeneity is beneficial for bat communities (Armitage and Ober 2012, Blakey et al. 2019), but also for other mammal (e.g. Hanley 1996, Sullivan et al. 2000), bird (e.g. Berg 1997, Drapeau et al. 2000), amphibian (e.g. Johansson et al. 2005, Loehle et al. 2005) and insect (e.g.

Jonsen and Fahrig 1997, Fuller et al. 2008) species. Acoustic and roost-availability surveys of bats should continue in the future to assess the intermediate and long-term changes in abundance and habitat use after the Kenow wildfire. Overall, although some bat species in WLNP appeared to immediately adapt to, or recover from, the significant ecological changes caused by the wildfire, continued monitoring is required to understand the resilience of bats to a high severity wildfire.

References

- ACIS [Alberta Climate Information Service]. 2020. Government of Alberta. Current and historical Alberta weather station data viewer. Retrieved from <https://acis.alberta.ca/weather-data-viewer.jsp>. Accessed on December 7, 2021.
- Adams, A. M., M. K. Jantzen, R. M. Hamilton, and M. B. Fenton. 2012. Do you hear what I hear? Implications of detector selection for acoustic monitoring of bats. *Methods in Ecology and Evolution* 3:992-998.
- AEP [Alberta Environment and Parks]. 2020. Wild species status search. Retrieved from <https://extranet.gov.ab.ca/env/wild-species-status/default.aspx>. Accessed on December 7, 2021.
- Agosta, S. J., D. Morton, and K. M. Kuhn. 2003. Feeding ecology of the bat *Eptesicus fuscus*: 'preferred' prey abundance as one factor influencing prey selection and diet breadth. *Journal of Zoology* 260:169-177.
- Agranat, I. 2013. Bat species identification from zero crossing and full spectrum echolocation calls using Hidden Markov Models, Fisher scores, unsupervised clustering and balanced winnow pairwise classifiers. *Acoustical Society of America, Proceedings of Meetings on Acoustics* 19:1-9.
- Aguiar, L., and Y. Antonini. 2008. Diet of two sympatric insectivores bats (Chiroptera: Vespertilionidae) in the Cerrado of Central Brazil. *Revista Brasileira de Zoologia* 25:28-31.
- Akoglu, H. 2018. User's guide to correlation coefficients. *Turkish Journal of Emergency Medicine* 18:91-93.
- Alberta Parks. 2015. Natural Regions and Subregions of Alberta. A framework for Alberta's Parks. Alberta Tourism, Parks and Recreation. Edmonton, Alberta. Retrieved from <https://albertaparks.ca/media/6256258/natural-regions-subregions-of-alberta-a-framework-for-albertas-parks-booklet.pdf>. Accessed on December 7, 2021.
- Aldridge, H. D. J. N., and R. M. Brigham. 1988. Load carrying and maneuverability in an insectivorous bat: a test of the 5% "rule" of radio-telemetry. *Journal of Mammalogy* 69:379-382.
- Allagas, P. 2020. Assessing the impact of restored wetlands on bat foraging activity over nearby farmland. MSc thesis, East Tennessee State University, Johnson City, Tennessee.
- Ancillotto, L., L. Bosso, P. Conti, and D. Russo. 2021. Resilient responses by bats to a severe wildfire: conservation implications. *Animal Conservation* 24:470-481.
- Angers, V. A., S. Gauthier, P. Drapeau, K. Jayen, and Y. Bergeron. 2011. Tree mortality and snag dynamics in North American boreal tree species after a wildfire: a long-term study. *International Journal of Wildland Fire* 20:751-763.

- Anthony, E. L. P. 1988. Age determination in bats. In: Kunz, T. H. (ed.). *Ecological and Behavioral Methods for the Study of Bats*. Smithsonian Institution Press. Washington, D.C., p. 47-57.
- Anthony, E. L., and T. H. Kunz. 1977. Feeding strategies of the little brown bat, *Myotis lucifugus*, in southern New Hampshire. *Ecology* 58:775-786.
- Armitage, D. W., and H. K. Ober. 2012. The effects of prescribed fire on bat communities in the longleaf pine sandhills ecosystem. *Journal of Mammalogy* 93:102–114.
- Arnett, E. B., and E. F. Baerwald. 2013. Impacts of wind energy development on bats: Implications for conservation. In: Adams, R. A., S. C. Pedersen (eds.). *Bat Evolution, Ecology, and Conservation*. Springer. New York, New York, p. 435-456.
- Austin, L. V., A. Silvis, W. M. Ford, M. Muthersbaugh, and K. E. Powers. 2018a. Bat activity following restoration prescribed burning in the central Appalachian upland and riparian habitats. *Natural Areas Journal* 38:183–195.
- Austin, L. V., A. Silvis, W. M. Ford, and K. E. Powers. 2020. Effects of historic wildfire and prescribed fire on site occupancy of bats in Shenandoah National Park, Virginia, USA. *Journal of Forest Research* 31:1255–1270.
- Austin, L. V., A. Silvis, M. S. Muthersbaugh, K. E. Powers, and W. M. Ford. 2018b. Bat activity following repeated prescribed fire in the central Appalachians, USA. *Fire Ecology* 14:1-11.
- Axelson, J. N., B. C. Hawkes, L. van Akker, and R. I. Alfaro. 2018. Stand dynamics and the mountain pine beetle—30 years of forest change in Waterton Lakes National Park, Alberta, Canada. *Canadian Journal of Forest Research* 48:1159-1170.
- Baerwald, E. F., and R. M. R. Barclay. 2011. Patterns of activity and fatality of migratory bats at a wind energy facility in Alberta, Canada. *The Journal of Wildlife Management* 75:1103-1114.
- Baerwald, E. F., G. H. D'Amours, B. J. Klug, and R. M. R. Barclay. 2008. Barotrauma is a significant cause of bat fatalities at wind turbines. *Current Biology* 18:R695-R696.
- Barclay, R. M. R. 1986. The echolocation calls of hoary (*Lasiurus cinereus*) and silver-haired (*Lasionycteris noctivagans*) bats as adaptations for long-versus short-range foraging strategies and the consequences for prey selection. *Canadian Journal of Zoology* 64:2700-2705.
- Barclay, R. M. R. 1991. Population structure of temperate zone insectivorous bats in relation to foraging behaviour and energy demand. *Journal of Animal Ecology* 60:165–178.
- Barclay, R. M. R. 1999. Bats are not birds - A cautionary note on using echolocation calls to identify bats: A comment. *Journal of Mammalogy* 80:290–296.
- Barclay, R. M. R., and R. M. Brigham. 1994. Constraints on optimal foraging: a field test of prey discrimination by echolocating insectivorous bats. *Animal Behaviour* 48:1013-1021.

- Barclay, R. M. R., and R. M. Brigham. 2001. Year-to-year reuse of tree-roosts by California bats (*Myotis californicus*) in southern British Columbia. *The American Midland Naturalist* 146:80-85.
- Barclay, R. M. R. and L. Harder. 2003. Life histories of bats: life in the slow lane. In: Kunz, T. H., and M. B. Fenton (eds.). *Bat Ecology*. The University of Chicago Press. Chicago, Illinois, p. 209-253.
- Barclay R. M. R., and A. Kurta. 2007. Ecology and behavior of bats roosting in tree cavities and under bark. In: Lacki M. J., J. P. Hayes, and A. Kurta (eds.). *Bats in forests: Conservation and management*. John Hopkins University Press. Baltimore, Maryland, p. 17–60.
- Barclay, R. M. R., J. Ulmer, C. J. A. MacKenzie, M. S. Thompson, L. Olson, J. McCool, E. Cropley, and G. Poll. 2004. Variation in the reproductive rate of bats. *Canadian Journal of Zoology* 82:688–693.
- Barrett, S. 1996. Parks Canada. The historic role of fire in Waterton Lakes National Park, Alberta.
- Barton, K. 2020. MuMIn: Multi-Model Inference. R package version 1.43.17. <https://CRAN.R-project.org/package=MuMIn>.
- Bartos, D. L., and W. F. Meuggler. 1981. Early succession in aspen communities following fire in western Wyoming. *Journal of Range Management* 34:315-318.
- Bates, D., M. Maechler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using lme4. *Journal of Statistical Software* 67:1-48.
- Belwood, J. J., and M. B. Fenton. 1976. Variation in the diet of *Myotis lucifugus* (Chiroptera: Vespertilionidae). *Canadian Journal of Zoology* 54:1674-1678.
- Bender, M. J., S. Perea, S. B. Castleberry, D. A. Miller, and T. B. Wigley. 2021. Influence of insect abundance and vegetation structure on site-occupancy of bats in managed pine forests. *Forest Ecology and Management* 482:118839.
- Berg, A. 1997. Diversity and abundance of birds in relation to forest fragmentation, habitat quality and heterogeneity. *Bird Study* 44:355-366.
- Bergeson, S. M., T. C. Carter, and M. D. Whitby. 2015. Adaptive roosting gives little brown bats an advantage over endangered Indiana bats. *The American Midland Naturalist* 174:321-330.
- Betts, B. J. 1998. Effects of interindividual variation in echolocation calls on identification of big brown and silver-haired bats. *The Journal of Wildlife Management* 62:1003-1010.
- Black, H. L. 1974. A north temperate bat community: structure and prey populations. *Journal of Mammalogy* 55:138-157.
- Blakey, R. V., B. S. Law, R. T. Kingsford, and J. Stoklosa. 2017. Terrestrial laser scanning reveals below-canopy bat trait relationships with forest structure. *Remote Sensing of Environment* 198:40–51.

- Blakey, R. V., E. B. Webb, D. C. Kesler, R. B. Siegel, D. Corcoran, and M. Johnson. 2019. Bats in a changing landscape: Linking occupancy and traits of a diverse montane bat community to fire regime. *Ecology and Evolution* 9:5324-5337.
- Blakey, R. V., E. B. Webb, D. C. Kesler, R. B. Siegel, D. Corcoran, J. S. Cole, and M. Johnson. 2021. Extent, configuration and diversity of burned and forested areas predict bat richness in a fire-maintained forest. *Landscape Ecology* 36:1101-1115.
- Blanco, C. M., and J. Garrie. 2020. Species specific effects of prescribed burns on bat occupancy in northwest Arkansas. *Forest Ecology and Management* 460:117890.
- Bohmann, K., A. Monadjem, C. Lehmkuhl Noer, M. Rasmussen, M. R. Zeale, E. Clare, G. Jones, E. Willerslev, and M. T. P. Gilbert. 2011. Molecular diet analysis of two African free-tailed bats (Molossidae) using high throughput sequencing. *PLoS One* 6:e21441.
- Bolker, B. M. 2008. *Ecological models and data in R*. Princeton University Press. Princeton, New Jersey, USA.
- Bolnick, D. I., R. Svanback, J. A. Fordyce, L. H. Yang, J. M. Davis, C. D. Hulsey, and M. L. Forister. 2003. The ecology of individuals: incidence and implications of individual specialization. *The American Naturalist* 161:1-28.
- Bosso, L., L. Ancillotto, S. Smeraldo, S. D'Arco, A. Migliozi, P. Conti, and D. Russo. 2018. Loss of potential bat habitat following a severe wildfire: A model-based rapid assessment. *International Journal of Wildland Fire* 27:756–769.
- Boyles, J. G., and D. P. Aubrey. 2006. Managing forests with prescribed fire: Implications for a cavity-dwelling bat species. *Forest Ecology and Management* 222:108–115.
- Boyles, J. G., P. M. Cryan, G. F. McCracken, and T. H. Kunz. 2011. Economic importance of bats in agriculture. *Science* 332:41–42.
- Braun de Torrez, E. C., H. K. Ober, and R. A. McCleery. 2018a. Activity of an endangered bat increases immediately following prescribed fire. *Journal of Wildlife Management* 82:1115–1123.
- Braun de Torrez, E. C., H. K. Ober, and R. A. McCleery. 2018b. Critically imperiled forest fragment supports bat diversity and activity within a subtropical grassland. *Journal of Mammalogy* 99:273-282.
- Brigham, R. M., H. D. J. N. Aldridge, and R. L. Mackey. 1992. Variation in habitat use and prey selection by Yuma bats, *Myotis yumanensis*. *Journal of Mammalogy* 73:640-645.
- Brigham, R. M., S. D. Grindal, M. C. Firman, and J. L. Morissette. 1997a. The influence of structural clutter on activity patterns of insectivorous bats. *Canadian Journal of Mammalogy* 75:131-136.
- Brigham, R. M., M. J. Vonhof, R. M. R. Barclay, and J. C. Gwilliam. 1997b. Roosting behavior and roost-site preferences of forest-dwelling California bats (*Myotis californicus*). *Journal of Mammalogy* 78:1231-1239.

- Brigham, R. M., and M. B. Saunders. 1990. The diet of big brown bats (*Eptesicus fuscus*) in relation to insect availability in southern Alberta, Canada. *Northwest Science* 64:7-10.
- Brittingham, M. C., and L. M. Williams. 2000. Bat boxes as alternative roosts for displaced bat maternity colonies. *Wildlife Society Bulletin* 28:197-207.
- Brose, P., T. Schuler, D. Van Lear, and J. Berst. 2001. Bringing fire back. *Journal of Forestry* 99:30-35.
- Brunet-Rossinni, A. K., and S. N. Austad. 2004. Ageing studies on bats: A review. *Biogerontology* 5:211-222.
- Brunet-Rossinni, A. and G. Wilkinson. 2009. Methods for age estimation and the study of senescence in bats. In: Kunz, T. H., and S. Parsons (eds.). *Ecological and Behavioral Methods for the Study of Bats*. John Hopkins University Press, Baltimore, Maryland, p. 315-325.
- Buchalski, M. R., J. B. Fontaine, P. A. Heady, J. P. Hayes, and W. F. Frick. 2013. Bat response to differing fire severity in mixed-conifer forest California, USA. *PLoS One* 8:e57884.
- Buchler, E. R. 1980. The development of flight, foraging, and echolocation in the little brown bat (*Myotis lucifugus*). *Behavioral Ecology and Sociobiology* 6:211-218.
- Buddle, C. M., D. W. Langor, G. R. Pohl, and J. R. Spence. 2005. Arthropod responses to harvesting and wildfire: Implications for emulation of natural disturbance in forest management. *Biological Conservation* 128:346-357.
- Burles, D. W., R. M. Brigham, R. A. Ring, and T. E. Reimchen. 2008. Diet of two insectivorous bats, *Myotis lucifugus* and *Myotis keenii*, in relation to arthropod abundance in a temperate Pacific Northwest rainforest environment. *Canadian Journal of Zoology* 86:1367-1375.
- Burnett, C. D., and P. V. August. 1981. Time and energy budgets for day roosting in a maternity colony of *Myotis lucifugus*. *Journal of Mammalogy* 62:758-766.
- Burnham, K. P. and D. R. Anderson. 2002. *Model selection and multimodel inference: a practical information-theoretic approach*. Springer-Verlag. New York, USA.
- Burnham, K. P., D. R. Anderson, and K. P. Huyvaert. 2011. AIC model selection and multimodel inference in behavioral ecology: some background, observations, and comparisons. *Behavioral Ecology and Sociobiology* 65:23-35.
- Burns, L. E., T. R. Frasier, and H. G. Broders. 2014. Genetic connectivity among swarming sites in the wide ranging and recently declining little brown bat (*Myotis lucifugus*). *Ecology and Evolution* 4:4130-4149.
- Burns, L. K., S. C. Loeb, and W. C. Bridges Jr. 2019. Effects of fire and its severity on occupancy of bats in mixed pine-oak forests. *Forest Ecology and Management* 446:151-163.

- Canadian Wildlife Health Cooperative [CWHC]. 2021. Wildlife health and SARS-CoV-2 in Canada: Bats. Interim guidance for wildlife management agencies, version 1.2. Retrieved from [http://www.cwhc-rcsf.ca/docs/miscellaneous/CWHC%20Bat%20health%20and%20SARS-CoV-2%20\(Versions%201.2\).pdf](http://www.cwhc-rcsf.ca/docs/miscellaneous/CWHC%20Bat%20health%20and%20SARS-CoV-2%20(Versions%201.2).pdf). Accessed on December 7, 2021.
- Carbone, L. M., J. Tavella, J. G. Pausas, and R. Aguilar. 2019. A global synthesis of fire effects on pollinators. *Global Ecology and Biogeography* 28:1487-1498.
- Carter, T. C., M. A. Menzel, S. F. Owen, J. W. Edwards, J. M. Menzel, and W. M. Ford. 2003. Food habits of seven species of bats in the Allegheny Plateau and Ridge and Valley of West Virginia. *Northeastern Naturalist* 10:83-88.
- Chaverri, G., O. E. Quirós, M. Gamba-Rios, and T. H. Kunz. 2007. Ecological correlates of roost fidelity in the tent-making bat *Artibeus watsoni*. *Ethology* 113:598-605.
- Clare, E. L., B. R. Barber, B. W. Sweeney, P. D. N. Hebert, and M. B. Fenton. 2011. Eating local: influences of habitat on the diet of little brown bats (*Myotis lucifugus*). *Molecular Ecology* 20:1772-1780.
- Clare, E. L., E. E. Fraser, H. E. Braid, M. B. Fenton, and P. D. Hebert. 2009. Species on the menu of a generalist predator, the eastern red bat (*Lasiurus borealis*): using a molecular approach to detect arthropod prey. *Molecular Ecology* 18:2532-2542.
- Clare, E. L., W. O. Symondson, H. Broders, F. Fabianek, E. E. Fraser, A. MacKenzie, A. Boughen, R. Hamilton, C. K. R. Willis, F. Martinez-Nunez, A. K. Menzies, K. J. O. Norquay, M. Brigham, J. Poissant, J. Rintoul, R. M. R. Barclay, and J. P. Reimer. 2014a. The diet of *Myotis lucifugus* across Canada: assessing foraging quality and diet variability. *Molecular Ecology* 23:3618-3632.
- Clare, E. L., W. O. Symondson, and M. B. Fenton. 2014b. An inordinate fondness for beetles? Variation in seasonal dietary preferences of night-roosting big brown bats (*Eptesicus fuscus*). *Molecular Ecology* 23:3633-3647.
- Cleveland, C. J., M. Betke, P. Federico, J. D. Frank, T. G. Hallam, J. Horn, J. D. López, G. F. McCracken, R. A. Medellín, A. Moreno-Valdez, C. G. Sansone, J. K. Westbrook, and T. H. Kunz. 2006. Economic value of the pest control service provided by Brazilian free-tailed bats in south-central Texas. *Frontiers in Ecology and the Environment* 4:238-243.
- Cline, S. P., A. B. Berg, and H. M. Wight. 1980. Snag characteristics and dynamics in douglas-fir forests, western Oregon. *The Journal of Wildlife Management* 44:773-786.
- Cox, M. R., E. V. Willcox, P. D. Keyser, and A. L. Vander Yacht. 2016. Bat response to prescribed fire and overstory thinning in hardwood forest on the Cumberland Plateau, Tennessee. *Forest Ecology and Management* 359:221-231.
- Crampton, L. H., and R. M. R. Barclay. 1998. Selection of roosting and foraging habitat by bats in different-aged aspen mixedwood stands. *Conservation Biology* 12:1347-1358.
- Cryan, P. M. 2003. Seasonal Distribution of Migratory Tree Bats (*Lasiurus* and *Lasionycteris*) in North America. *Journal of Mammalogy* 84:579-593.

- Cryan, P. M., M. A. Bogan, and J. S. Altenbach 2000. Effect of elevation on distribution of female bats in the Black Hills, South Dakota. *Journal of Mammalogy* 81:719-725.
- Cryan, P. M., J. W. Jameson, E. F. Baerwald, C. K. R. Willis, R. M. R. Barclay, E. A. Snider, and E. G. Crichton. 2012. Evidence of late-summer mating readiness and early sexual maturation in migratory tree-roosting bats found dead at wind turbines. *PLoS One* 7:e47586-e47586.
- Czenze, Z. J., K. A. Jonasson, and C. K. R. Willis. 2017. Thrifty females, frisky males: Winter energetics of hibernating bats from a cold climate. *Physiological and Biochemical Zoology* 90:502–511.
- Dale, V. H., L. A. Joyce, S. McNulty, R. P. Neilson, M. P. Ayres, M. D. Flannigan, P. J. Hanson, L. C. Irland, A. E. Lugo, C. J. Peterson, D. Simberloff, F. J. Swanson, B. J. Stocks, and B. M. Wotton. 2001. Climate change and forest disturbances. *BioScience* 51:723-734.
- Davis, W. H., and H. B. Hitchcock. 1965. Biology and migration of the bat, *Myotis lucifugus*, in New England. *Journal of Mammalogy* 46:296-313.
- Dempster, W. R., and S. Meredith. 2021. Growth and yield of lodgepole pine stands disturbed by mountain pine beetle in the Lower Foothills of Alberta. *The Forestry Chronicle* 97:65-77.
- Denzinger, A., and H. Schnitzler. 2013. Bat guilds, a concept to classify the highly diverse foraging and echolocation behaviors of microchiropteran bats. *Frontiers in Physiology* 4:1–15.
- Dias-Silva, L., G. T. Duarte, R. Alves, M. J. R. Pereira, and A. Paglia. 2018. Feeding and social activity of insectivorous bats in a complex landscape: The importance of gallery forests and karst areas. *Mammalian Biology* 88:52–63.
- Dodd, L. E., M. J. Lacki, E. R. Britzke, D. A. Buehler, P. D. Keyser, J. L. Larkin, A. D. Rodewald, T. B. Wigley, P. B. Wood, and L. K. Rieske. 2012. Forest structure affects trophic linkages: How silvicultural disturbance impacts bats and their insect prey. *Forest Ecology and Management* 267:262–270.
- Doty, A. C., C. Stawski, B. S. Law, and F. Geiser. 2016. Post-wildfire physiological ecology of an Australian microbat. *Journal of Comparative Physiology B* 186:937-946.
- Doty, A. C., S. E. Currie, C. Stawski, and F. Geiser. 2018. Can bats sense smoke during deep torpor? *Physiology and Behavior* 185:31–38.
- Drapeau, P., A. Leduc, J. F. Giroux, J. P. L. Savard, Y. Bergeron, and W. L. Vickery. 2000. Landscape-scale disturbances and changes in bird communities of boreal mixed-wood forests. *Ecological Monographs* 70:423-444.
- Eberhard, M. J. W. 1975. The evolution of social behavior by kin selection. *The Quarterly Review of Biology* 50:1-33.

- Elemans, C. P., A. F. Mead, L. Jakobsen, and J. M. Ratcliffe. 2011. Superfast muscles set maximum call rate in echolocating bats. *Science* 333:1885-1888.
- Elia, M., R. Laforzezza, E. Tarasco, G. Colangelo, and G. Sanesi. 2012. The spatial and temporal effects of fire on insect abundance in Mediterranean forest ecosystems. *Forest Ecology and Management* 263:262-267.
- Erickson, J. L., and S. D. West. 2003. Associations of bats with local structure and landscape features of forested stands in western Oregon and Washington. *Biological Conservation* 109:95-102.
- Erickson, J. L., and S. D. West. 2002. The influence of regional climate and nightly weather conditions on activity patterns of insectivorous bats. *Acta Chiropterologica* 4:17-24.
- Evelyn, M. J., D. A. Stiles, and R. A. Young. 2004. Conservation of bats in suburban landscapes: roost selection by *Myotis yumanensis* in a residential area in California. *Biological Conservation* 115:463-473.
- Faure, P. A., and R. M. R. Barclay. 1994. Substrate-gleaning versus aerial-hawking: plasticity in the foraging and echolocation behaviour of the long-eared bat, *Myotis evotis*. *Journal of Comparative Physiology A* 174:651-660.
- Fenton, M. B. 1970. A technique for monitoring bat activity with results obtained from different environments in southern Ontario. *Canadian Journal of Zoology* 48:847-851.
- Fenton, M. B. 1997. Science and the conservation of bats. *Journal of Mammalogy* 78:1-14.
- Fenton, M. B. 2013. Questions, ideas and tools: Lessons from bat echolocation. *Animal Behaviour* 85:869-879.
- Fenton, M. B., and R. M. R. Barclay. 1980. *Myotis lucifugus*. *American Society of Mammalogists* 142:1-8.
- Findlay, S. V., and R. M. R. Barclay. 2020. Acoustic surveys for bats are improved by taking habitat type into account. *Wildlife Society Bulletin* 44:86-93.
- Flannigan, M. D., and J. B. Harrington. 1988. A study of the relation of meteorological variables to monthly provincial area burned by wildfire in Canada (1953-80). *Journal of Applied Meteorology and Climatology* 27:441-452.
- Flannigan, M. D., B. J. Stocks, and B. M. Wotton. 2000. Climate change and forest fires. *Science of the Total Environment* 262:221-229.
- Fontaine, J. B., and P. L. Kennedy. 2012. Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in U.S. Fire-prone forests. *Ecological Applications* 22:1547-1561.
- Ford, W. M., J. B. Johnson, and T. V. Gundy. 2021. Northern long-eared bat (*Myotis septentrionalis*) day-roost loss in the central Appalachian Mountains following prescribed burning. *International Journal of Forestry Research* 2021:1-6.

- Ford, W. M., J. M. Menzel, M. A. Menzel, J. W. Edwards, and J. C. Kilgo. 2006. Presence and absence of bats across habitat scales in the upper coastal plain of South Carolina. *The Journal of Wildlife Management* 70:1200-1209.
- Fox, J., and S. Weisberg. 2019. *An {R} Companion to Applied Regression*, Third Edition. Thousand Oaks CA: Sage. <https://socialsciences.mcmaster.ca/jfox/Books/Companion/>.
- Frick, W. F. 2013. Acoustic monitoring of bats, considerations of options for long-term monitoring. *Therya* 4:69-70.
- Frick, W. F., E. F. Baerwald, J. F. Pollock, R. M. R. Barclay, J. A. Szymanski, T. J. Weller, A. L. Russell, S. C. Loeb, R. A. Medellin, and L. P. McGuire. 2017. Fatalities at wind turbines may threaten population viability of a migratory bat. *Biological Conservation* 209:172–177.
- Frick W.F., S. J. Puechmaille, and C. K. R. Willis. 2016. White-nose syndrome in bats. In: Voigt C., and T. Kingston (eds.). *Bats in the Anthropocene: Conservation of Bats in a Changing World*. Springer Open, New York, p. 245-262.
- Frick, W. F., D. S. Reynolds, and T. H. Kunz. 2010. Influence of climate and reproductive timing on demography of little brown myotis *Myotis lucifugus*. *Journal of Animal Ecology* 79:128–136.
- Friedenberg, N. A., and W. F. Frick. 2021. Assessing fatality minimization for hoary bats amid continued wind energy development. *Biological Conservation* 262:109309.
- Fuentes-Montemayor, E., D. Goulson, L. Cavin, J. M. Wallace, and K. J. Park. 2013. Fragmented woodlands in agricultural landscapes: The influence of woodland character and landscape context on bats and their insect prey. *Agriculture, Ecosystems and Environment* 172:6–15.
- Fukui, D., K. Okazaki, M. Miyazaki, and K. Maeda. 2010. The effect of roost environment on roost selection by non-reproductive and dispersing Asian parti-coloured bats *Vespertilio sinensis*. *Mammal Study* 35:99-109.
- Fuller, R. J., T. H. Oliver, and S. R. Leather. 2008. Forest management effects on carabid beetle communities in coniferous and broadleaved forests: implications for conservation. *Insect Conservation and Diversity* 1:242-252.
- Gallant, A. J., and H. G. Broders. 2015. Body condition explains little of the interindividual variation in the swarming behaviour of adult male little brown myotis (*Myotis lucifugus*) in Nova Scotia, Canada. *Canadian Journal of Zoology* 93:469-476.
- Geiser, F., C. Stawski, A. C. Doty, C. E. Cooper, and J. Nowack. 2018. A burning question: What are the risks and benefits of mammalian torpor during and after fires? *Conservation Physiology* 6:1–12.
- Gibbons, P., and D. Lindenmayer. 2002. *Tree hollows and wildlife conservation in Australia*. CSIRO Publishing. Collingwood, Australia.

- Gillett, N. P., A. J. Weaver, F. W. Zwiers, and M. D. Flannigan. 2004. Detecting the effect of climate change on Canadian forest fires. *Geophysical Research Letters* 31:1-4.
- GOA [Government of Alberta]. 2021. Alberta river basins. Retrieved from <https://rivers.alberta.ca/>. Accessed December 7, 2021.
- GOA [Government of Alberta]. 2020. Alberta regional dashboard, Improvement district No. 4 Waterton. Retrieved from <https://regionaldashboard.alberta.ca/region/improvement-district-no-4-waterton/#/>. Accessed December 7, 2021.
- GOC [Government of Canada]. 2019. Canada Open Data License. Information licensed from Parks Canada under the Open Government Licence – Canada. Retrieved from <https://open.canada.ca/en/open-government-licence-canada>. Accessed on December 7, 2021.
- GOC [Government of Canada]. 2021. Species at risk public registry. Retrieved from <https://www.canada.ca/en/environment-climate-change/services/species-risk-public-registry.html>. Accessed on December 7, 2021.
- Goerlitz, H. R. 2018. Weather conditions determine attenuation and speed of sound: Environmental limitations for monitoring and analyzing bat echolocation. *Ecology and Evolution* 8:5090-5100.
- Gonsalves, L., B. Law, C. Webb, and V. Monamy. 2013. Foraging ranges of insectivorous bats shift relative to changes in mosquito abundance. *PloS One* 8:e64081.
- Griffin, D. R., F. A. Webster, and C. R. Michael. 1960. The echolocation of flying insects by bats. *Animal Behaviour* 8:141–154.
- Griffiths, S. R. 2013. Echolocating bats emit terminal phase buzz calls while drinking on the wing. *Behavioural Processes* 98:58–60.
- Griffiths, A. D., S. T. Garnett, and B. W. Brook. 2015. Fire frequency matters more than fire size: Testing the pyrodiversity-biodiversity paradigm for at-risk small mammals in an Australian tropical savanna. *Biological Conservation* 186:337–346.
- Grindal, S. D., T. S. Collard, R. M. Bringham, and R. M. R. Barclay. 1992. The influence of precipitation on reproduction by *Myotis* bats in British Columbia. *American Midland Naturalist* 128:339-344.
- Grindal, S. D., J. L. Morissette, and R. M. Bringham. 1999. Concentration of bat activity in riparian habitats over an elevational gradient. *Canadian Journal of Zoology* 77:972-977.
- Haarsma, A. J. 2008. Manual for assessment of reproductive status, age and health in European Vespertilionid bats. Retrieved from <https://arnhemspeil.nl/docs/2008-09-12-zoogdiervereniging-dds-manual-for-assessment-of-reproductive-status-age-and-health-in-european-vespertilionid-bats.pdf>. Accessed on December 7, 2021.

- Hancock, M. H., A. Amphlett, R. Proctor, D. Dugan, J. Willi, P. Harvey, and R. W. Summers. 2011. Burning and mowing as habitat management for capercaillie *Tetrao urogallus*: An experimental test. *Forest Ecology and Management* 262:509–521.
- Hanley, T. A. 1996. Potential role of deer (Cervidae) as ecological indicators of forest management. *Forest Ecology and Management* 88:199–204.
- Hamilton, I. M., and R. M. R. Barclay. 1998. Diets of juvenile, yearling, and adult big brown bats (*Eptesicus fuscus*) in southeastern Alberta. *Journal of Mammalogy* 79:764–771.
- Heinselman, M. L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. *Quaternary Research* 3:329–382.
- Henry, M., J. F. Cosson, and J. M. Pons. 2007. Abundance may be a misleading indicator of fragmentation-sensitivity: The case of fig-eating bats. *Biological Conservation* 139:462–467.
- Henry, M., D. W. Thomas, R. Vaudry, and M. Carrier. 2002. Foraging distances and home range of pregnant and lactating little brown bats (*Myotis lucifugus*). *Journal of Mammalogy* 83:767–774.
- Hill, A. P., P. Prince, J. L. Snaddon, C. P. Doncaster, and A. Rogers. 2019. AudioMoth: A low-cost acoustic device for monitoring biodiversity and the environment. *HardwareX* 6:e00073.
- Huebschman, J.J., and T. B. Bragg. 2000. Response of regal fritillary (*Speyeria idalia* Drury) to spring burning in an eastern Nebraska tallgrass prairie, USA. *Natural Areas Journal* 20:386–388.
- Humes, M. L., J. P. Hayes, and M. W. Collopy. 1999. Bat activity in thinned, unthinned, and old-growth forests in western Oregon. *The Journal of Wildlife Management* 63:553–561.
- Humphrey, S. R. 1975. Nursery roosts and community diversity of Nearctic bats. *Journal of Mammalogy* 56:321–346.
- Inkster-Draper, T. E., M. Sheaves, C. N. Johnson, and S. K. A. Robson. 2013. Prescribed fire in eucalypt woodlands: immediate effects on a microbat community of northern Australia. *Wildlife Research* 40:70–76.
- IPCC [Intergovernmental Panel on Climate Change]. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. Geneva, Switzerland.
- Jaffe, K. E., and L. A. Isbell. 2009. After the fire: Benefits of reduced ground cover for vervet monkeys (*Cercopithecus aethiops*). *American Journal of Primatology* 71:252–260.
- Jakobsen, L., and A. Surlykke. 2010. Vespertilionid bats control the width of their biosonar sound beam dynamically during prey pursuit. *Proceedings of the National Academy of Sciences* 107:13930–13935.

- Jantzen, M. K., and M. B. Fenton. 2013. The depth of edge influence among insectivorous bats at forest–field interfaces. *Canadian Journal of Zoology* 91:287-292.
- Johansson, M., C. R. Primmer, J. Sahlsten, and J. Merila. 2005. The influence of landscape structure on occurrence, abundance and genetic diversity of the common frog, *Rana temporaria*. *Global Change Biology* 11:1664-1679.
- Johnson, J. B., J. W. Edwards, W. M. Ford, and J. E. Gates. 2009. Roost tree selection by northern myotis (*Myotis septentrionalis*) maternity colonies following prescribed fire in a central Appalachian Mountains hardwood forest. *Forest Ecology and Management* 258:233–242.
- Johnson, J. B., W. M. Ford, J. L. Rodrigue, J. W. Edwards, and C. M. Johnson. 2010. Roost selection by male Indiana *Myotis* following forest fires in central Appalachian hardwoods forests. *Journal of Fish and Wildlife Management* 1:111-121.
- Johnson, J. S., M. J. Lacki, and M. D. Baker. 2007. Foraging ecology of long-legged *Myotis* (*Myotis volans*) in north-central Idaho. *Journal of Mammalogy* 88:1261-1270.
- Johnson, J. S., J. J. Treanor, M. J. Lacki, M. D. Baker, G. A. Falxa, L. E. Dodd, A. G. Waag, and E. H. Lee. 2017. Migratory and winter activity of bats in Yellowstone National Park. *Journal of Mammalogy* 98:211–221.
- Johnson, J. S., J. J. Treanor, A. C. Slusher, and M. J. Lacki. 2019. Buildings provide vital habitat for little brown *Myotis* (*Myotis lucifugus*) in a high-elevation landscape. *Ecosphere* 10:e02925
- Johnstone, J. F., C. D. Allen, J. F. Franklin, L. E. Frelich, B. J. Harvey, P. E. Higuera, M. C. Mack, R. K. Meentemeyer, M. R. Metz, G. L. W. Perry, T. Schoennagel, and M. G. Turner. 2016. Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment* 14:369–378.
- Jones, G., D. S. Jacobs, T. H. Kunz, M. R. Wilig, and P. A. Racey. 2009. Carpe noctem: The importance of bats as bioindicators. *Endangered Species Research* 8:93–115.
- Jonsen, I. D., and L. Fahrig. 1997. Response of generalist and specialist insect herbivores to landscape spatial structure. *Landscape Ecology* 12:185-197.
- Jung, T. S., I. D. Thompson, and R. D. Titman. 2004. Roost site selection by forest-dwelling male *Myotis* in central Ontario, Canada. *Forest Ecology and Management* 202:325-335.
- Jung, T. S. 2020. Bats in the changing boreal forest: response to a megafire by endangered little brown bats (*Myotis lucifugus*). *Ecoscience* 27:59-70.
- Kalda, O., R. Kalda, and J. Liira. 2015. Multi-scale ecology of insectivorous bats in agricultural landscapes. *Agriculture, Ecosystems and Environment* 199:105–113.
- Kalko, E. K. 1995. Insect pursuit, prey capture and echolocation in pipistrelle bats (*Microchiroptera*). *Animal Behaviour* 50:861-880.

- Kassambara, A. 2019. ggcorrplot: Visualization of a Correlation Matrix using 'ggplot2'. R package version 0.1.3. <https://CRAN.R-project.org/package=ggcorrplot>.
- Kaupas, L. 2016. Roosting behaviour and thermoregulation of the northern long-eared bat (*Myotis septentrionalis*) near the northern extent of its range. MSc thesis, University of Calgary, Calgary, Alberta.
- Kelly, L. T., A. Haslem, G. J. Holland, S. W. J. Leonard, J. Machunter, M. Bassett, A. F. Bennett, M. J. Bruce, E. K. Chia, F. J. Christie, M. F. Clarke, J. Di Stefano, R. Loyn, M. A. McCarthy, A. Pung, N. Robinson, H. Sitters, M. Swan, and A. York. 2017. Fire regimes and environmental gradients shape vertebrate and plant distributions in temperate eucalypt forests. *Ecosphere* 8:e01781.
- Kerth, G., and B. König. 1999. Fission, fusion and nonrandom associations in female Bechstein's bats (*Myotis bechsteinii*). *Behaviour* 136:1187-1202.
- Kerth, G., K. Safi, and B. König. 2002. Mean colony relatedness is a poor predictor of colony structure and female philopatry in the communally breeding Bechstein's bat (*Myotis bechsteinii*). *Behavioral Ecology and Sociobiology* 52:203-210.
- Kinsella, E. K. 2003. A comparison of terrestrial arthropods on post-fire forested landscapes under different management regimes in southwestern Alberta. MSc thesis, University of Calgary, Calgary, Alberta.
- Klimesova, J., A. Pokorna, and L. Klimes. 2009. Establishment growth and bud-bank formation in *Epilobium angustifolium*: the effects of nutrient availability, plant injury, and environmental heterogeneity. *Botany* 87:195-201.
- Koch, D., M. A. Lewis, and S. Lele. 2021. The signature of endemic populations in the spread of mountain pine beetle outbreaks. *Bulletin of Mathematical Biology* 83:1-24.
- Koehler, C. E., and R. M. R. Barclay. 2000. Post-natal growth and breeding biology of the hoary bat (*Lasiurus cinereus*). *Journal of Mammalogy* 81:234-244.
- Kral, K. C., R. F. Limb, J. P. Harmon, and T. J. Hovick. 2017. Arthropods and fire: previous research shaping future conservation. *Rangeland Ecology and Management* 70:589-598.
- Krawchuk, M. A., G. W. Meigs, J. M. Cartwright, J. D. Coop, R. Davis, A. Holz, C. Kolden, and A. J. H. Meddens. 2020. Disturbance refugia within mosaics of forest fire, drought, and insect outbreaks. *Frontiers in Ecology and the Environment* 18:235-244.
- Krebs, C. J. 1989. *Ecological Methodology*. Harper and Row. New York, New York, USA.
- Kunz, T. H. 1971. Reproduction of some Vespertilionid bats in central Iowa. *American Midland Naturalist* 86:477-486.
- Kunz, T. H. 1982. Roosting ecology of bats. In: Kunz, T. H. (ed.). *Ecology of Bats*. Springer. Boston, Massachusetts, p. 1-55.
- Kunz, T. H., E. Braun de Torrez, D. Bauer, T. Lobova, and T. H. Fleming. 2011. Ecosystem services provided by bats. *Annals of the New York Academy of Sciences* 1223:1-38.

- Kunz, T. H., and L. F. Lumsden. 2003. Ecology of cavity and foliage roosting bats. In: Kunz, T. H., and M. B. Fenton (eds.). *Bat Ecology*. University of Chicago Press. Chicago, Illinois, p. 3-89.
- Kunz, T. H., O. T. Oftedal, S. K. Robson, M. B. Kretzmann, and C. Kirk. 1995. Changes in milk composition during lactation in three species of insectivorous bats. *Journal of Comparative Physiology B* 164:543–551.
- Kurta, A., G. P. Bell, K. A. Nagy, and T. H. Kunz. 1989. Energetics of pregnancy and lactation in freeranging little brown bats (*Myotis lucifugus*). *Physiological Zoology* 62:804-818.
- Lacki, M.J., S.K. Amelon, and M.D. Baker. 2007. Foraging ecology of bats in forests. In: Lacki, M. J., J. P. Hayes, and A. Kurta (eds.). *Bats in forests: conservation and management*. Johns Hopkins University Press. Baltimore, Maryland, p. 83–127.
- Lacki, M. J., M. D. Baker, and J. S. Johnson. 2010. Geographic variation in roost-site selection of long-legged *Myotis* in the Pacific Northwest. *Journal of Wildlife Management* 74:1218–1228.
- Lacki, M. J., D. R. Cox, L. E. Dodd, and M. B. Dickinson. 2009. Response of northern bats (*Myotis septentrionalis*) to prescribed fires in eastern Kentucky forests. *Journal of Mammalogy* 90:1165–1175.
- Lacki, M. J., and L. E. Dodd. 2011. Diet and foraging behavior of *Corynorhinus* in eastern North America. In: Loeb, S. C., M. J. Lacki, and D. A. Miller (eds.). *Conservation and management of eastern big-eared bats: a symposium*. General Technical Report SRS-145. Asheville, NC: US Department of Agriculture Forest Service, Southern Research Station, p. 39-52.
- Lacki, M. J., L. E. Dodd, N. S. Skowronski, M. B. Dickinson, and L. K. Rieske. 2017. Relationships among burn severity, forest canopy structure and bat activity from spring burns in oak–hickory forests. *International Journal of Wildland Fire* 26:963-972.
- Lacki, M. J., and J. H. Schwierjohann. 2001. Day-roost characteristics of northern bats in mixed mesophytic forest. *The Journal of Wildlife Management* 65:482-488.
- Langor, D. 2019. Responses of insects to the Kenow Fire in Waterton Lakes National Park. Research Report. Canadian Forest Service, Edmonton, Alberta.
- Lausen, C. L. 2007. Roosting ecology and landscape genetics of prairie bats. PhD thesis, University of Calgary, Calgary, Alberta.
- Lausen, C. 2012. Waterton Lakes National Park Bat Survey. Report for Parks Canada. Birchdale Ecological Ltd. Kaslo, British Columbia.
- Lausen, C. 2016. Bat sonograms from Acoustic Techniques Course [class handouts]. Wildlife Conservation Society Canada. Creston, British Columbia.
- Lausen, C. L., and R. M. R. Barclay. 2006. Benefits of living in a building: big brown bats (*Eptesicus fuscus*) in rocks versus buildings. *Journal of Mammalogy* 87:362–370.

- Law B. S., and R. V. Blakey. 2021. Bats in temperate forests: Where are the trends in bat populations? In: Lim, B. K., M. B. Fenton, R. M. Brigham, S. Mistry, A. Kurta, E. H. Gillam, A. Russell, and J. Ortega (eds.). *50 Years of Bat Research, Foundations and New Frontiers*. Springer International Publishing, Switzerland, p. 93-104.
- Law, B., A. Doty, M. Chidel, and T. Brassil. 2018. Bat activity before and after a severe wildfire in Pilliga forests: Resilience influenced by fire extent and landscape mobility? *Austral Ecology* 43:706–718.
- Lawes, M. J., B. P. Murphy, A. Fisher, J. C. Woinarski, A. C. Edwards, and J. Russell-Smith. 2015. Small mammals decline with increasing fire extent in northern Australia: evidence from long-term monitoring in Kakadu National Park. *International Journal of Wildland Fire* 24:712-722.
- Lemen, C., P. W. Freeman, J. A. White, and B. R. Andersen. 2015. The problem of low agreement among automated identification programs for acoustical surveys of bats. *Western North American Naturalist* 75:218-225.
- Lenth, R. V. 2021. emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.6.1. <https://CRAN.R-project.org/package=emmeans>.
- Lewis, S. E. 1995. Roost fidelity of bats: a review. *Journal of Mammalogy* 76:481–496.
- Li, H., R. Petric, Z. Alazzawi, J. Kauzlarich, R. H. Mahmoud, R. McFadden, N. Perslow, A. Rodriguez Flores, H. Soufi, K. Morales, M. C. Kalcounis-Rueppell, M. D. Schug, and L. A. Zarecky. 2021. Four years continuous monitoring reveals different effects of urban constructed wetlands on bats. *Land* 10:1087.
- Limpens, H. J. G. A. 2004. Field identification: using bat detectors to identify species. In: Brigham, R. M., E. K. V. Kalko, G. Jones, S. Parsons, and H. J. G. A. Limpens (eds.). *Bat Echolocation Research: Tools, Techniques and Analysis*. Bat Conservation International, Austin, Texas, p. 46–57.
- Loeb, S.C., T. J. Rodhouse, L. E. Ellison, C. L. Lausen, J. D. Reichard, K. M. Irvine, T. E. Ingersoll, J. T. Coleman, W. E. Thogmartin, J. R. Sauer, C. M. Francis, M. L. Bayless, T. R. Stanley, and D. H. Johnson. 2015. A plan for the North American bat monitoring program (NABat). General Technical Report SRS-208. Asheville, NC: US Department of Agriculture Forest Service, Southern Research Station.
- Loeb, S. C., and T. A. Waldrop. 2008. Bat activity in relation to fire and fire surrogate treatments in southern pine stands. *Forest Ecology and Management* 255:3185–3192.
- Loehle, C., T. B. Wigley, P. A. Shipman, S. F. Fox, S. Rutzmoser, R. E. Thill, and M. A. Melchior. 2005. Herpetofaunal species richness responses to forest landscape structure in Arkansas. *Forest Ecology and Management* 209:293-308.
- Ludecke, D. 2018. ggeffects: Tidy Data Frames of Marginal Effects from Regression Models. *Journal of Open Source Software* 3:1-5.

- Ludecke, D., D. Makowski, P. Waggoner, and I. Patil. 2020. Performance: Assessment of regression models performance. CRAN. <https://easystats.github.io/performance/>.
- Mackey, R. L., and R. M. R. Barclay. 1988. The influence of physical clutter and noise on the activity of bats over water. *Canadian Journal of Zoology* 67:1167-1170.
- Malison, R. L., and C. V. Baxter. 2010a. The fire pulse: Wildfire stimulates flux of aquatic prey to terrestrial habitats driving increases in riparian consumers. *Canadian Journal of Fisheries and Aquatic Sciences* 67:570–579.
- Malison, R. L., and C. V. Baxter. 2010b. Effects of wildfire of varying severity on benthic stream insect assemblages and emergence. *Journal of the North American Benthological Society* 29:1324-1338.
- Maplogs. 2021. Worldwide elevation map finder sunset sunrise times lookup. Sunrise sunset times of Waterton, AB T0K, Canada. Retrieved from https://sunrise.maplogs.com/waterton_ab_t0k_canada.9556.html?. Accessed on December 7, 2021.
- Matsuta, N., S. Hiryu, E. Fujioka, Y. Yamada, H. Riquimaroux, and Y. Watanabe. 2013. Adaptive beam-width control of echolocation sounds by CF–FM bats, *Rhinolophus ferrumequinum nippon*, during prey-capture flight. *Journal of Experimental Biology* 216:1210-1218.
- Maucieri, D. G., and R. M. R. Barclay. 2021. Consumption of spiders by the little brown bat (*Myotis lucifugus*) and the long-eared myotis (*Myotis evotis*) in the Rocky Mountains of Alberta, Canada. *Canadian Journal of Zoology* 99:221-226.
- Mayberry, H. W., L. P. McGuire, and C. K. R. Willis. 2018. Body temperatures of hibernating little brown bats reveal pronounced behavioural activity during deep torpor and suggest a fever response during white-nose syndrome. *Journal of Comparative Physiology B* 188:333–343.
- McCracken, G. F., J. K. Westbrook, V. A. Brown, M. Eldridge, P. Federico, and T. H. Kunz. 2012. Bats track and exploit changes in insect pest populations. *PLoS ONE* 7:e43839.
- McGuire, L. P., M. B. Fenton, and C. G. Guglielmo. 2009. Effect of age on energy storage during prehibernation swarming in little brown bats (*Myotis lucifugus*). *Canadian Journal of Zoology* 87:515-519.
- Menzel, M. A., T. C. Carter, J. M. Menzel, W. M. Ford, and B. R. Chapman. 2002. Effects of group selection silviculture in bottomland hardwoods on the spatial activity patterns of bats. *Forest Ecology and Management* 162:209–218.
- Meyer, D., A. Zeileis, and K. Hornik. 2020. vcd: Visualizing Categorical Data. R package version 1.4-8.
- Mickleburgh, S. P., A. M. Hutson, and P. A. Racey. 2002. A review of the global conservation status of bats. *Oryx* 36:18-34.

- Miller, L. A., and H. J. Degn. 1981. The acoustic behavior of four species of Vespertilionid bats studied in the field. *Journal of Comparative Physiology* 142:67–74.
- Minshall, G. W. 2003. Responses of stream benthic macroinvertebrates to fire. *Forest Ecology and Management* 178:155-161.
- Minshall, G. W., C. T. Robinson, and D. E. Lawrence. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 54:2509-2525.
- Moretti, M., P. Duelli, and M. K. Obrist. 2006. Biodiversity and resilience of arthropod communities after fire disturbance in temperate forests. *Oecologia* 149:312-327.
- Morningstar, D., and A. Sandilands. 2019. Summer movements of a radio-tagged hoary bat (*Lasiurus cinereus*) captured in southwestern Ontario. *Canadian Field-Naturalist* 133:125–129.
- Morrison, M. L., and M. G. Raphael. 1993. Modeling the dynamics of snags. *Ecological Applications* 3:322-330.
- Mukaka, M. M. 2012. A guide to appropriate use of correlation coefficient in medical research. *Malawi Medical Journal* 24:69-71.
- Muller, J., M. Mehr, C. Bassler, M. B. Fenton, T. Hothorn, H. Pretzsch, H. Klemmt, and R. Brandl. 2012. Aggregative response in bats: prey abundance versus habitat. *Oecologia* 169:673-684.
- Munshi-South, J., and G. S. Wilkinson. 2010. Bats and birds: Exceptional longevity despite high metabolic rates. *Ageing Research Reviews* 9:12–19.
- Muthersbaugh, M. S., W. M. Ford, K. E. Powers, and A. Silvis. 2019. Activity patterns of bats during the fall and spring along ridgelines in the central Appalachians. *Journal of Fish and Wildlife Management* 10:180-195.
- Nagorsen, D. W., and R. M. Brigham. 1993. Bats of British Columbia. *Royal British Columbia Museum Handbook, Vol. 1. The Mammals of British Columbia*. UBC Press, Vancouver, British Columbia.
- Nakagawa, S., P. C. D. Johnson, and H. Schielzeth. 2017. The coefficient of determination R^2 and intra-class correlation coefficient from generalized linear mixed-effects models revisited and expanded. *Journal of the Royal Society Interface* 14:20170213.
- Naughton, D. 2012. *The natural history of Canadian mammals*. University of Toronto Press, Toronto, Ontario.
- Neilson, A. L., and M. B. Fenton. 1994. Responses of little brown *Myotis* to exclusion and to bat houses. *Wildlife Society Bulletin* 22:8-14.
- Nelson, J. J., and E. H. Gillam. 2017. Selection of foraging habitat by female little brown bats (*Myotis lucifugus*). *Journal of Mammalogy* 98:222-231.

- Nilsson, J. A., C. Bronmark, L. A. Hansson, and B. B. Chapman. 2014. Individuality in movement: the role of animal personality. In: Hansson, L. A., and S. Akesson (eds.). *Animal movement across scales*. Oxford University Press, United Kingdom, p. 90-109.
- Nocera, T., W. M. Ford, A. Silvis, and C. A. Dobony. 2019. Let's Agree to Disagree: Comparing auto-acoustic identification programs for northeastern bats. *Journal of Fish and Wildlife Management* 10:346-361.
- Norberg, U. M. 1981. Allometry of bat wings and legs and comparison with birds wings. *Philosophical Transactions of the Royal Society of London B* 292:359-398.
- Norberg, U. M. 1990. Vertebrate flight, mechanics, physiology, morphology, ecology and evolution. *Zoophysiology* 27:1-291.
- Obrist, M. K. 1995. Flexible bat echolocation: the influence of individual, habitat and conspecifics on sonar signal design. *Behavioral Ecology and Sociobiology* 36:207–219.
- Obrist, M. K., E. Rathey, F. Bontadina, A. Martinoli, M. Conedera, P. Christe, and M. Moretti. 2011. Response of bat species to sylvo-pastoral abandonment. *Forest Ecology and Management* 261:789-798.
- Ogle, D.H., P. Wheeler, and A. Dinno. 2021. FSA: Fisheries Stock Analysis. R package version 0.9.0. <https://github.com/droglenc/FSA>.
- O'Keefe, J. M., and S. C. Loeb. 2017. Indiana bats roost in ephemeral, fire-dependent pine snags in the southern Appalachian Mountains, USA. *Forest Ecology and Management* 391:264–274.
- Oliveira, J. M., A. L. F. Destro, M. B. Freitas, and L. L. Oliveira. 2021. How do pesticides affect bats? – a brief review of recent publications. *Brazilian Journal of Biology* 81:499–507.
- Olson, C. R., and T. L. Flach. 2016. Beneficial management practices for bats for the Milk River and the South Saskatchewan Watershed in Alberta. Alberta Conservation Association, MULTISAR Project. Lethbridge, Alberta. Retrieved from <http://multisar.ca/wp-content/uploads/2015/10/Multisar-Bat-BMP-Report-Final.pdf>. Accessed on December 7, 2021.
- Parks Canada. 2010. Waterton Lakes National Park of Canada: Management Plan. Retrieved from <https://www.pc.gc.ca/en/pn-np/ab/waterton/info/index/directeur-management>. Accessed on December 7, 2021.
- Parks Canada. 2017a. Waterton Lakes National Park EI monitoring program. 2015-2017 species inventory and activity levels monitoring protocol for bat populations in Waterton Lakes National Park v1.0.
- Parks Canada. 2017b. Waterton Lakes National Park, Impact of mountain pine beetle. Retrieved from <https://www.pc.gc.ca/en/docs/v-g/dpp-mpb/sec4/dpp-mpb4b>. Accessed on December 7, 2021.

- Parks Canada. 2018. Waterton Lakes National Park, Climate. Retrieved from <https://www.pc.gc.ca/en/pn-np/ab/waterton/nature/environment/climat-climate>. Accessed on December 7, 2021.
- Parks Canada 2021a. Waterton Lakes National Park, Kenow Wildfire. Retrieved from <https://www.pc.gc.ca/en/pn-np/ab/waterton/nature/environment/feu-fire/feu-fire-kenow>. Accessed on December 7, 2021.
- Parks Canada 2021b. Waterton Lakes National Park, Kenow fire: Burn severity map. Retrieved from <https://www.pc.gc.ca/en/pn-np/ab/waterton/nature/environment/feu-fire/feu-fire-kenow/brulage-burn>. Accessed on December 7, 2021.
- Parks Canada 2021c. Waterton Lakes National Park, 125 of our favourite Waterton facts. Retrieved from <https://www.pc.gc.ca/en/pn-np/ab/waterton/culture/histoire-history/faits-facts>. Accessed on January 19, 2022.
- Patriquin, K. J., and R. M. R. Barclay. 2003. Foraging by bats in cleared, thinned and unharvested boreal forest. *Journal of Applied Ecology* 40:646–657.
- Patriquin, K. J., L. K. Hogberg, B. J. Chruszcz, and R. M. R. Barclay. 2003. The influence of habitat structure on the ability to detect ultrasound using bat detectors. *Wildlife Society Bulletin* 31:475-481.
- Pausas, J. G., and C. L. Parr. 2018. Towards an understanding of the evolutionary role of fire in animals. *Evolutionary Ecology* 32:113–125.
- Perea, S., and E. Tena. 2020. Different bat detectors and processing software... Same results? *Journal of Bat Research and Conservation* 13:1-5.
- Perry, R. W. 2012. A review of fire effects on bats and bat habitat in the eastern oaks region. In: Dey, D. C., M. C. Stambaugh, S. L. Clark, and C. J. Schweitzer (eds.). *Proceedings of the 4th Fire in Eastern Oak Forests Conference*. Department of Agriculture, Forest Service, Northern Research Station. Newtown Square, Pennsylvania, p. 170–191.
- Perry, R. W., and V. L. McDaniel. 2015. Temperatures below leaf litter during winter prescribed burns: implications for litter-roosting bats. *International Journal of Wildland Fire* 24:544-549.
- Perry, R. W., and R. E. Thill. 2008. Diurnal roosts of male evening bats (*Nycticeius humeralis*) in diversely managed pine-hardwood forests. *The American Midland Naturalist* 160:374-386.
- Powell, A. F. L. A., W. H. Busby, and K. Kindscher. 2007. Status of the regal fritillary (*Speyeria idalia*) and effects of fire management on its abundance in northeastern Kansas, USA. *Journal of Insect Conservation* 11:299–308.
- Psyllakis, J. M., and R. M. Brigham. 2006. Characteristics of diurnal roosts used by female *Myotis* bats in sub-boreal forests. *Forest Ecology and Management* 223:93-102.

- Racey, P. A. 1973. Environmental factors affecting the length of gestation in heterothermic bats. *Journal of Reproduction and Fertility* 19:175-189.
- Racey, P. A. 1998. Ecology of European bats in relation to their conservation. In: Kunz, T.H., and P.A. Racey (eds.). *Bat Biology and Conservation*. Smithsonian Institution Press, Washington, DC, p. 249–260.
- Racey, P. A. 2009. Reproductive assessment of bats. In: Kunz, T. H., and S. Parsons (eds.). *Ecological and Behavioral Methods for the Study of Bats*. John Hopkins University Press, Baltimore, Maryland, p. 249-264.
- Racey, P. A., and S. M. Swift. 1981. Variations in gestation length in a colony of pipistrelle bats (*Pipistrellus pipistrellus*) from year to year. *Journal of Reproduction and Fertility* 61:123-129.
- Rainho, A., A. M. Augusto, and J. M. Palmeirim. 2010. Influence of vegetation clutter on the capacity of ground foraging bats to capture prey. *Journal of Applied Ecology* 47:850–858.
- Rainho, A., and J. M. Palmeirim. 2011. The importance of distance to resources in the spatial modelling of bat foraging habitat. *PLoS ONE* 6:e19227.
- Randall, L. A., T. S. Jung, and R. M. R. Barclay. 2014. Roost-site selection and movements of little brown myotis (*Myotis lucifugus*) in southwestern Yukon. *Northwestern Naturalist* 95:312-317.
- Randall-Parker, T., and R. Miller. 2002. Effects of prescribed fire in ponderosa pine on key wildlife habitat components: preliminary results and a method for monitoring. Proceedings of the symposium on the ecology and management of dead wood in western forests. Pacific southwest research station, USDA forest service general technical report PSW-GTR-181:823-834.
- Ransome, R.D. 1973. Factors affecting the timing of births of the Greater Horseshoe bat, (*Rhinolophus ferrumequinum*). *Periodical Biology* 75:169-175.
- Ratcliffe, J. M., and J. W. Dawson. 2003. Behavioural flexibility: the little brown bat, *Myotis lucifugus*, and the northern long-eared bat, *M. septentrionalis*, both glean and hawk prey. *Animal Behaviour* 66:847-856.
- Ratcliffe, J. M., C. P. Elemans, L. Jakobsen, and A. Surlykke. 2013. How the bat got its buzz. *Biology Letters* 9:20121031.
- R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria. <https://www.R-project.org/>.
- Reed, C. C. 1997. Responses of prairie insects and other arthropods to prescription burns. *Natural Areas Journal* 17:380–385.

- Reichert, B., C. Lausen, S. Loeb, T. Weller, R. Allen, E. Britzke, T. Hohoff, J. Siemers, B. Burkholder, C. Herzog, and M. Verant. 2018. A guide to processing bat acoustic data for the North American Bat Monitoring Program (NABat). U.S. Geological Survey Open-File Report 2018–1068. Retrieved from <https://pubs.er.usgs.gov/publication/ofr20181068>. Accessed on December 7, 2021.
- Reimer, J. P. 2013. Nocturnality under the midnight sun: Behavioural adaptations of the little brown bat (*Myotis lucifugus*) at 60N degrees latitude. MSc thesis. University of Calgary. Calgary, Alberta.
- Reimer, J. P., E. F. Baerwald, and R. M. R. Barclay. 2010. Diet of hoary (*Lasiurus cinereus*) and silver-haired (*Lasionycteris noctivagans*) bats while migrating through southwestern Alberta in late summer and autumn. *The American Midland Naturalist* 164:230–237.
- Rocca, M. E., P. M. Brown, L. H. MacDonald, and C. M. Carrico. 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. *Forest Ecology and Management* 327:290–305.
- Romme, W. H., M. S. Boyce, R. Gresswell, E. H. Merrill, G. W. Minshall, C. Whitlock, and M. G. Turner. 2011. Twenty years after the 1988 Yellowstone fires: Lessons about disturbance and ecosystems. *Ecosystems* 14:1196–1215.
- Runge, M. C., E. H. C. Grant, J. T. H. Coleman, J. D. Reichard, S. E. J. Gibbs, P. M. Cryan, K. J. Olival, D. P. Walsh, D. S. Blehert, M. C. Hopkins, and J. M. Sleeman. 2020. Assessing the risks posed by SARS-CoV-2 in and via North American bats—Decision framing and rapid risk assessment. U.S. Geological Survey Open-File Report 2020–1060. Retrieved from <https://pubs.usgs.gov/of/2020/1060/ofr20201060.pdf>. Accessed on December 7, 2021.
- Russo, D., and L. Ancillotto. 2015. Sensitivity of bats to urbanization: A review. *Mammalian Biology* 80:205–212.
- Russo, D., L. Ancillotto, L. Cistrone, and C. Korine. 2016. The buzz of drinking on the wing in echolocating bats. *Ethology* 122:226–235.
- Russo, D., L. Ancillotto, and G. Jones. 2018a. Bats are still not birds in the digital era: echolocation call variation and why it matters for bat species identification. *Canadian Journal of Zoology* 96:63–78.
- Russo, D., L. Bosso, and L. Ancillotto. 2018b. Novel perspectives on bat insectivory highlight the value of this ecosystem service in farmland: Research frontiers and management implications. *Agriculture, Ecosystems and Environment* 266:31–38.
- Russo, D., L. Cistrone, and G. Jones. 2005. Spatial and temporal patterns of roost use by tree-dwelling barbastelle bats *Barbastella barbastellus*. *Ecography* 28:769–776.
- Russo, D., G. Jones, and R. Arlettaz. 2007. Echolocation and passive listening by foraging mouse-eared bats *Myotis myotis* and *M. blythii*. *Journal of Experimental Biology* 210:166–176.

- Russo, D., and C. C. Voigt. 2016. The use of automated identification of bat echolocation calls in acoustic monitoring: A cautionary note for a sound analysis. *Ecological Indicators* 66:598–602.
- Rydell, J., S. Nyman, J. Eklof, G. Jones, and D. Russo. 2017. Testing the performances of automated identification of bat echolocation calls: A request for prudence. *Ecological Indicators* 78:416–420.
- Salcedo, H. D. L. C., M. B. Fenton, M. Hickey, and R. Blake. 1995. Energetic consequences of flight speeds of foraging red and hoary bats (*Lasiurus borealis* and *Lasiurus cinereus*; Chiroptera: Vespertilionidae). *The Journal of Experimental Biology* 198:2245–2251.
- Salinas-Ramos, V. B., L. Ancillotto, L. Bosso, V. Sánchez-Cordero, and D. Russo. 2020. Interspecific competition in bats: state of knowledge and research challenges. *Mammal Review* 50:68–81.
- Salvarina, I. 2016. Bats and aquatic habitats: a review of habitat use and anthropogenic impacts. *Mammal Review* 46:131–143.
- Saunders, E. H. 2015. Bat assemblage and selection of maternity roosts in a post-wildfire landscape. MSc thesis, Northern Arizona University, Flagstaff, Arizona.
- Saunders, M. B., and R. M. R. Barclay. 1992. Ecomorphology of insectivorous bats: a test of predictions using two morphologically similar species. *Ecology* 73:1335–1345.
- Schirmer, A., A. Herde, J. A. Eccard, and M. Dammhahn. 2019. Individuals in space: personality-dependent space use, movement and microhabitat use facilitate individual spatial niche specialization. *Oecologia* 189:647–660.
- Schmieder, D. A., T. Kingston, R. Hashim, and B. M. Siemers. 2010. Breaking the trade-off: rainforest bats maximize bandwidth and repetition rate of echolocation calls as they approach prey. *Biology Letters* 6:604–609.
- Schnitzler, H. U., and E. K. Kalko. 2001. Echolocation by insect-eating bats: we define four distinct functional groups of bats and find differences in signal structure that correlate with the typical echolocation tasks faced by each group. *Bioscience* 51:557–569.
- Schnitzler, H., C. F. Moss, and A. Denzinger. 2003. From spatial orientation to food acquisition in echolocating bats. *Trends in Ecology and Evolution* 18:386–394.
- Schowalter, D. B. 1980. Swarming, reproduction, and early hibernation of *Myotis lucifugus* and *M. volans* in Alberta, Canada. *Journal of Mammalogy* 61:350–354.
- Schwab, N. A. 2006. Roost-site selection and potential prey sources after wildland fire for two insectivorous bat species (*Myotis evotis* and *Myotis lucifugus*) in mid-elevation forests of western Montana. MSc thesis, University of Montana, Missoula, Montana.
- Sedgeley, J. A., and C. F. O'Donnell. 1999. Factors influencing the selection of roost cavities by a temperate rainforest bat (Vespertilionidae: *Chalinolobus tuberculatus*) in New Zealand. *Journal of Zoology* 249:437–446.

- Shalaway, S. 2004. Butterflies in the backyard. Stackpole Books. Mechanicsburg, Pennsylvania, USA.
- Sherwin, H. A., W. I. Montgomery, and M. G. Lundy. 2013. The impact and implications of climate change for bats. *Mammal Review* 43:171–182.
- Shively, R., and P. Barboza. 2017. Range and roosting ecology of the little brown bat, *Myotis lucifugus*, in interior and northern Alaska. *Northwestern Naturalist* 98:122–131.
- Shump, K. A., and A. U. Shump. 1982. *Lasiurus borealis*. *Mammalian Species* 183:1–6.
- Silvis, A., S. D. Gehrt, and R. A. Williams. 2016. Effects of shelterwood harvest and prescribed fire in upland Appalachian hardwood forests on bat activity. *Forest Ecology and Management* 360:205–212.
- Singleton, M. P., A. E. Thode, A. J. Sanchez Meador, and J. M. Iniguez. 2019. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. *Forest Ecology and Management* 433:709–719.
- Sleep, D. J., and R. M. Brigham. 2003. An experimental test of clutter tolerance in bats. *Journal of Mammalogy* 84:216–224.
- Smith, T. N. 2019. Elucidating patterns of bat species occupancy across a disturbed landscape in California's central valley. MSc thesis, Humboldt State University, Arcata, California.
- Snider, E. A. 2009. Post-fire insect communities and roost selection by western long-eared *Myotis* (*Myotis evotis*) in Mesa Verde National Park. MSc thesis, Colorado State University, Fort Collins, Colorado.
- Snider, E. A., P. M. Cryan, and K. R. Wilson. 2013. Roost selection by western long-eared *Myotis* (*Myotis evotis*) in burned and unburned pinon–juniper woodlands of southwestern Colorado. *Journal of Mammalogy* 94:640–649.
- Solick, D. I., and R. M. R. Barclay. 2007. Geographic variation in the use of torpor and roosting behaviour of female western long-eared bats. *Journal of Zoology* 272:358–366.
- Spencer, C. N., and F. R. Hauer. 1991. Phosphorus and nitrogen dynamics in streams during a wildfire. *Journal of the North American Benthological Society* 10:24–30.
- Starbuck, C. A., E. S. Considine, and C. L. Chambers. 2020. Water and elevation are more important than burn severity in predicting bat activity at multiple scales in a post-wildfire landscape. *PLoS One* 15:e0231170.
- Steel, Z. L., B. Campos, W. F. Frick, R. Burnett, and H. D. Safford. 2019. The effects of wildfire severity and pyrodiversity on bat occupancy and diversity in fire-suppressed forests. *Scientific Reports* 9:1–11.
- Stephens, S. L., N. Burrows, A. Buyantuyev, R. W. Gray, R. E. Keane, R. Kubian, S. Liu, F. Seijo, L. Shu, K. G. Tolhurst, and J. W. van Wagtendonk. 2014. Temperate and boreal forest mega-fires: characteristics and challenges. *Ecological Society of America* 12:115–122.

- Sullivan, T. P., D. S. Sullivan, and P. M. Lindgren. 2000. Small mammals and stand structure in young pine, seed-tree, and old-growth forest, southwest Canada. *Ecological Applications* 10:1367-1383.
- Swengel, A. B. 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitat. *Biodiversity and Conservation* 10:1141-1169.
- Swengel, A.B., and S. R. Swengel. 2007. Benefit of permanent non-fire refugia for Lepidoptera conservation in fire-managed sites. *Journal of Insect Conservation* 11:263–279.
- Swetnam, T. W. 1993. Fire history and climate change in giant sequoia groves. *Science* 262:885-889.
- Szewczak, J. M. 2010. The benefits of full-spectrum data for analyzing bat echolocation calls. SonoBat, bat call analysis software. Retrieved from <https://sonobat.com/wp-content/uploads/2014/02/presentation.pdf>. Accessed on December 7, 2021.
- Szewczak, J. 2018. Echolocation call characteristics of western US bats [class handouts]. Humboldt State University Bat Lab. Arcata, California.
- Taillie, P. J., E. C. Braun de Torrez, A. D. Potash, W. W. Boone IV, M. Jones, M. A. Wallrichs, F. Schellenberg, K. Hooker, H. K. Ober, and R. A. McCleery. 2021. Bat activity response to fire regime depends on species, vegetation conditions, and behavior. *Forest Ecology and Management* 502 :119722.
- Tena, E., G. Fandos, O. de Paz, R. de la Pena, and J. L. Telleria. 2020. Size does matter: Passive sampling in urban parks of a regional bat assemblage. *Urban Ecosystems* 23:227-234.
- Thomas, J. P., M. L. Reid, T. S. Jung, and R. M. R. Barclay. 2019. Site occupancy of little brown bats (*Myotis lucifugus*) in response to salvage logging in the boreal forest. *Forest Ecology and Management* 451:117501.
- Titchenell, M. A., R. A. Williams, and S. D. Gehrt. 2011. Bat response to shelterwood harvests and forest structure in oak-hickory forests. *Forest Ecology and Management* 262:980–988.
- Tormanen, A. P., and J. Garrie. 2021. Responses of bat-prey to prescribed burns and mechanical thinning. *Forest Ecology and Management* 491:119162.
- Torrey, K. E. 2018. Interactions between imperiled bat species and a fire-maintained ecosystem in the southern Appalachian Mountains. MSc thesis, University of West Georgia, Carrollton, Georgia.
- Toscano, B. J., N. J. Gownaris, S. M. Heerhartz, and C. J. Monaco. 2016. Personality, foraging behavior and specialization: Integrating behavioral and food web ecology at the individual level. *Oecologia* 182:55-69.
- U.S. Fish and Wildlife Service. 2021. White-nose syndrome response team. Spread map. Retrieved from <https://www.whitenosesyndrome.org/>. Accessed December 7, 2021.

- Valdez, E. W., and P. M. Cryan. 2013. Insect prey eaten by hoary bats (*Lasiurus cinereus*) prior to fatal collisions with wind turbines. *Western North American Naturalist* 73:516-524.
- van Kooten, G. C., J. Duan, and R. Lynch. 2016. Is there a future for nuclear power? Wind and emission reduction targets in fossil-fuel Alberta. *PloS One* 11:e0165822.
- Verboom, B., and H. Huitema. 1997. The importance of linear landscape elements for the pipistrelle *Pipistrellus pipistrellus* and the serotine bat *Eptesicus serotinus*. *Landscape Ecology* 12:117–125.
- Verboom, B., and K. Spoelstra. 1999. Effects of food abundance and wind on the use of tree lines by an insectivorous bat, *Pipistrellus pipistrellus*. *Canadian Journal of Zoology* 77:1393-1401.
- Vieira, N. K., W. H. Clements, L. S. Guevara, and B. F. Jacobs. 2004. Resistance and resilience of stream insect communities to repeated hydrologic disturbances after a wildfire. *Freshwater Biology* 49:1243-1259.
- Vogel, J.A., R. R. Koford, and D. M. Debinski. 2010. Direct and indirect responses of tallgrass prairie butterflies to prescribed burning. *Journal of Insect Conservation* 14:663–677.
- Voigt C. C., and T. Kingston. 2016. Bats in the Anthropocene. In: Voigt C., and T. Kingston (eds.). *Bats in the Anthropocene: Conservation of Bats in a Changing World*. Springer Open, New York, p. 1-9.
- Voigt, C. C., and D. Lewanzik. 2011. Trapped in the darkness of the night: Thermal and energetic constraints of daylight flight in bats. *Proceedings of the Royal Society B: Biological Sciences* 278:2311–2317.
- Vonhof, M. J. 2006. Handbook of inventory methods and standard protocols for surveying bats in Alberta. Alberta Sustainable Resource Development, Fish and Wildlife Division, Edmonton, Alberta.
- Vonhof, M. J., and R. M. R. Barclay. 1996. Roost-site selection and roosting ecology of forest-dwelling bats in southern British Columbia. *Canadian Journal of Zoology* 74:1797–1805.
- Waldrop, T. A., D. L. White, and M. Jones. 1992. Fire regimes for pine-grassland communities in the southeastern United States. *Forest Ecology and Management* 47:195–210.
- Walters, C. L., A. Collen, T. Lucas, K. Mroz, C. A. Sayer, and K. E. Jones. 2013. Challenges of using bioacoustics to globally monitor bats. In: Adams, R. A., and S. C. Pedersen (eds.). *Bat Evolution, Ecology, and Conservation*. Springer, New York, p. 479–499.
- Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. *Science* 313:940–943.
- Whelan, R. J. 1995. *The ecology of fire*. Cambridge University Press. Cambridge, United Kingdom.
- Wickham, H., R. Francois, L. Henry, and K. Muller. 2021. dplyr: A Grammar of Data Manipulation. R package version 1.0.5. <https://CRAN.R-project.org/package=dplyr>.

- Wilde, C. J., C. H. Knight, and P. A. Racey. 1999. Influence of torpor on milk protein composition and secretion in lactating bats. *Journal of Experimental Zoology* 284:35-41.
- Willis, C. K. R., and R. M. Brigham. 2007. Social thermoregulation exerts more influence than microclimate on forest roost preferences by a cavity dwelling bat. *Behavioral Ecology and Sociobiology* 62:97–108.
- Wimsatt, W. A. 1945. Notes on breeding behavior, pregnancy, and parturition in some vesperilionid bats of the eastern United States. *Journal of Mammalogy* 26:23-33.
- Wright, H. A., and A. W. Bailey. 1982. *Fire ecology: United states and southern Canada*. John Wiley and Sons. New York, USA.
- Yates, M. D., and R. M. Muzika. 2006. Effect of forest structure and fragmentation on site occupancy of bat species in Missouri Ozark forests. *The Journal of Wildlife Management* 70:1238-1248.
- Zamora-Gutierrez, V., G. M. C. MacSwiney, S. Martínez Balvanera, and E. Robredo Esquivelzeta. 2021. The Evolution of Acoustic Methods for the Study of Bats. In: Lim, B. K., M. B. Fenton, R. M. Brigham, S. Mistry, A. Kurta, E. H. Gillam, A. Russell, and J. Ortega (eds.). *50 Years of Bat Research, Foundations and New Frontiers*. Springer International Publishing, Switzerland, p. 43-59.
- Zuur, A. F., A. Mira, F. Carvalho, E. N. Ieno, A. A. Saveliev, G. M. Smith, and N. J. Walker. 2009. Negative binomial GAM and GAMM to analyse amphibian roadkills. In: Zuur, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, G. M. Smith. *Mixed Effects Models and Extensions in Ecology with R*. Springer, New York, p. 383-397.

Appendix A: Fire History of Waterton Lakes National Park, Alberta

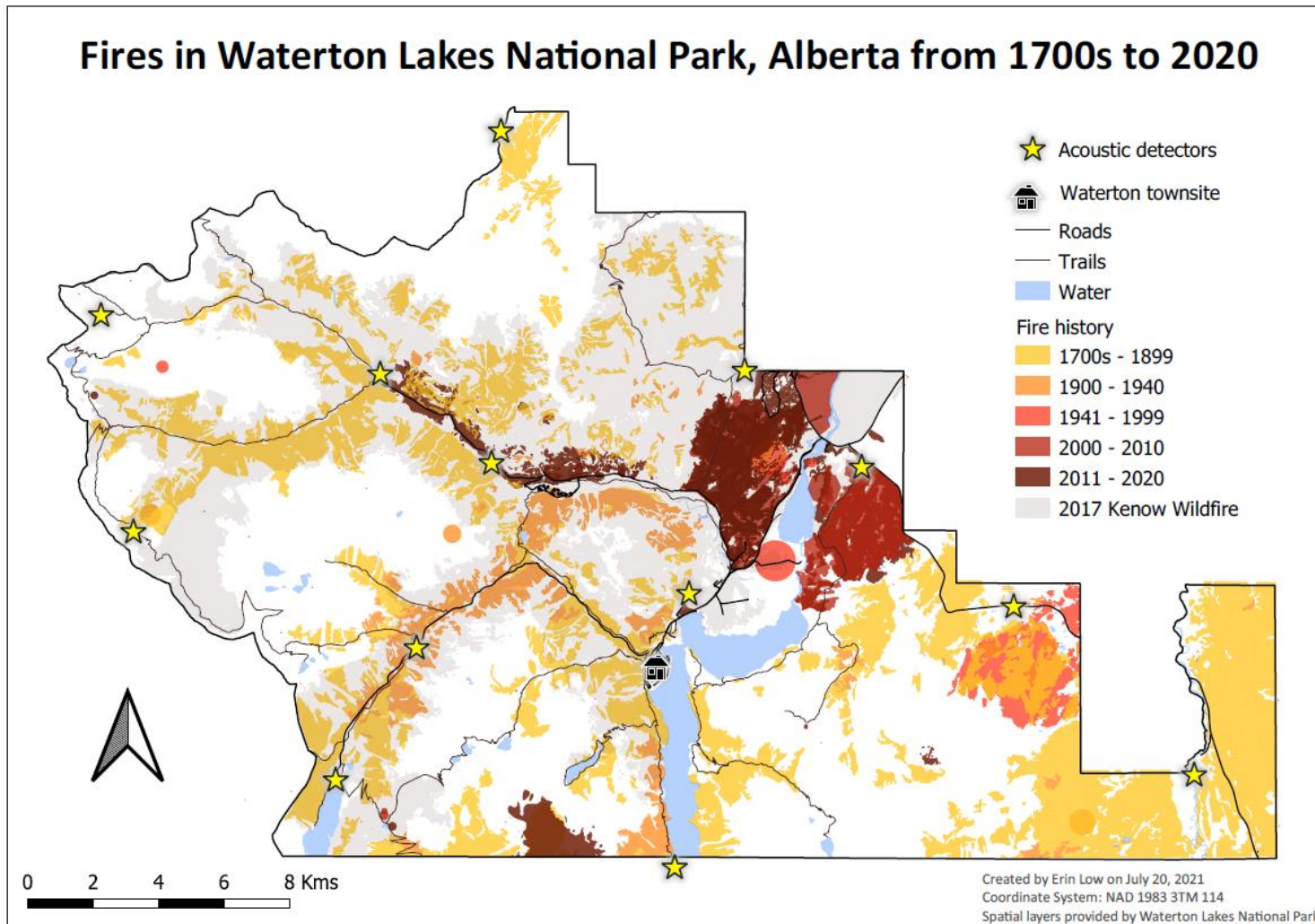


Fig. A.1. Fires (both wild and prescribed) that occurred in Waterton Lakes National Park, Alberta from the 1700s to 2020. Note that the fire distribution from the 1700s to approximately 1950 is based on empirical observation of fire scars and fire-induced regeneration (Barrett 1996). It should therefore be taken as an approximation of the fire extent during this time period. The map contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

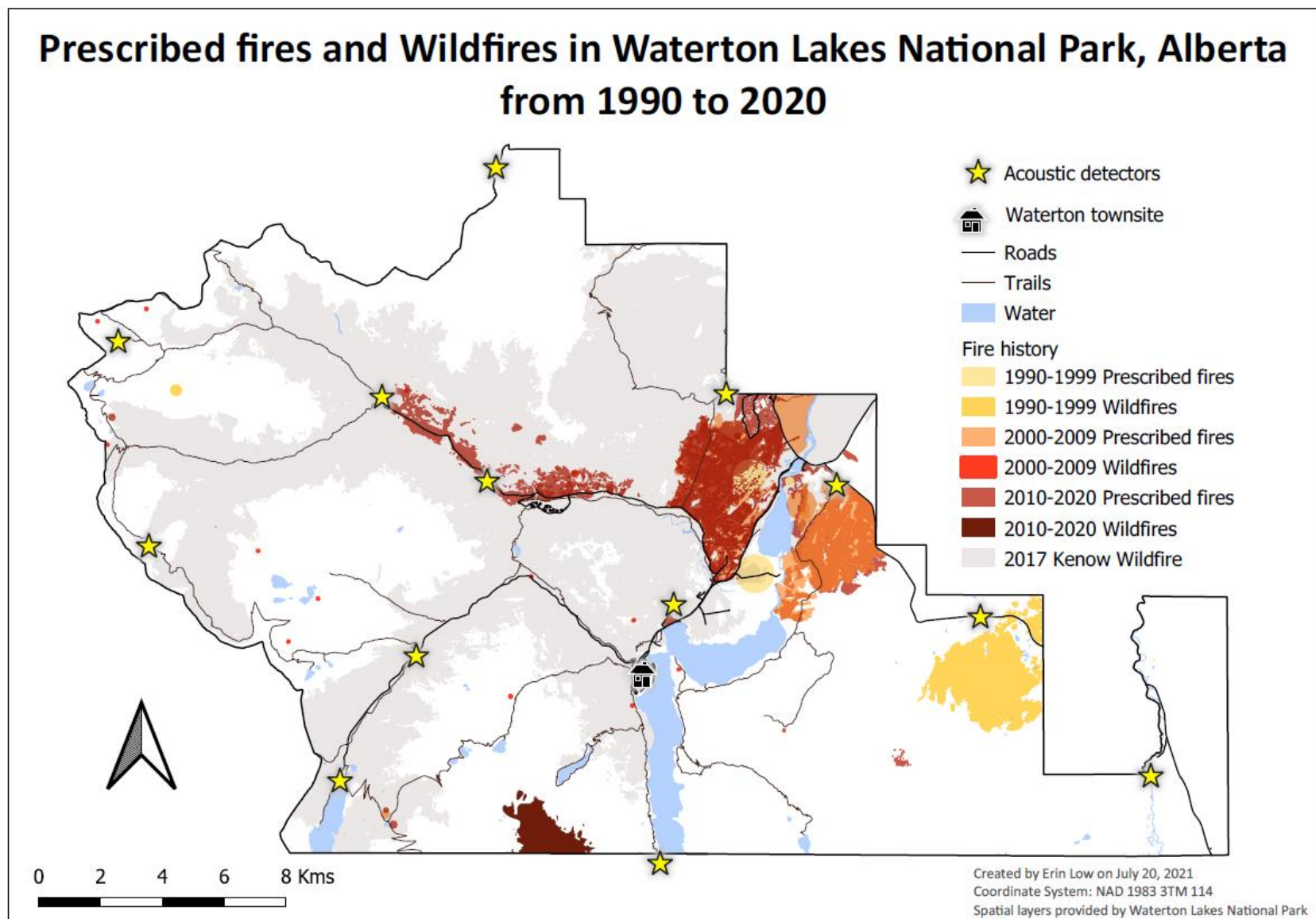


Fig. A.2. Prescribed fires and wildfires that occurred in Waterton Lakes National Park, Alberta from 1990 to 2020. The name of each acoustic detector site is displayed. The map contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

Prescribed fires and Wildfires in Waterton Lakes National Park, Alberta from 2015 to 2020

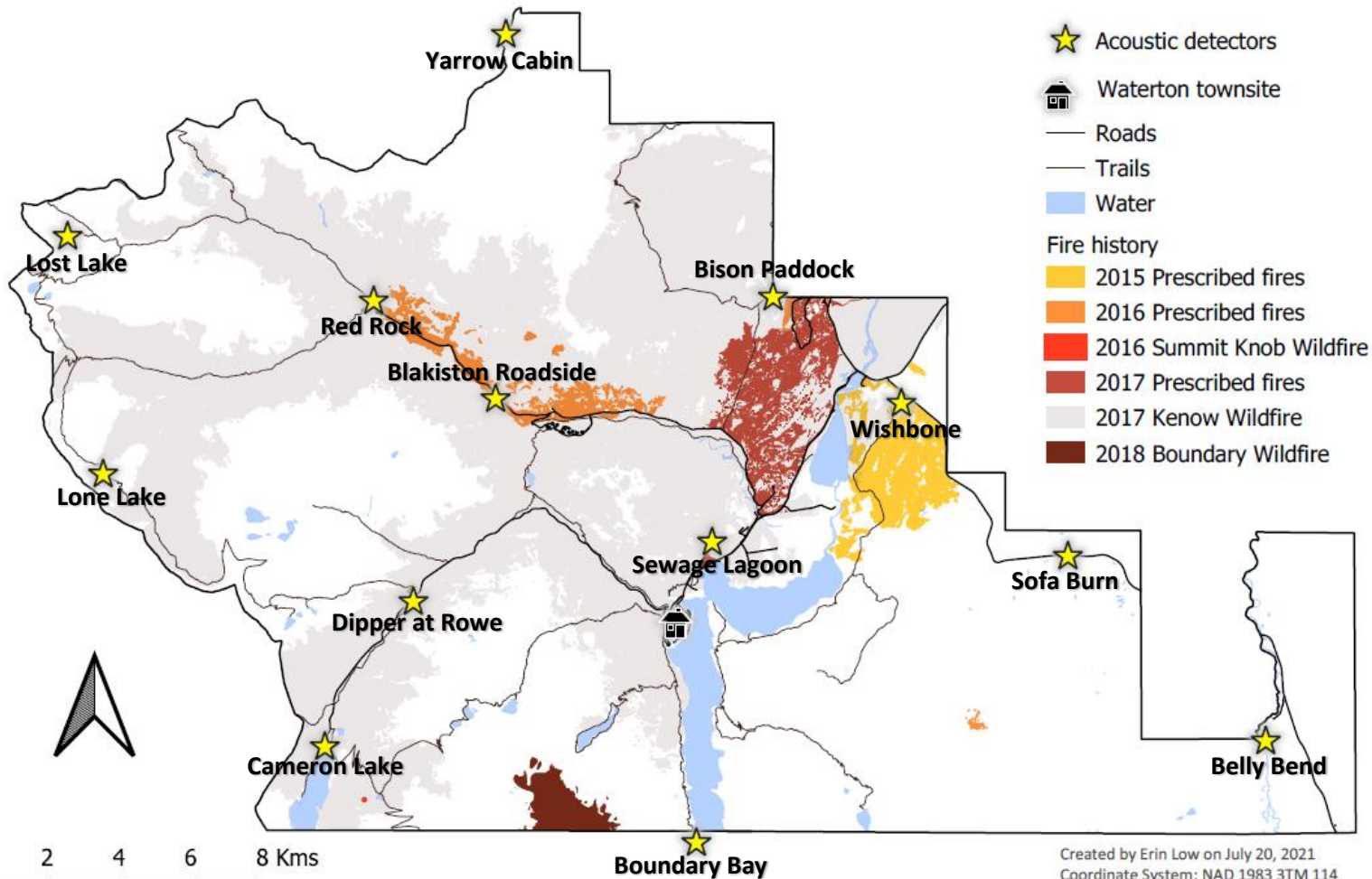


Fig. A.3. Prescribed fires and wildfires that occurred in Waterton Lakes National Park, Alberta from 2015 to 2020. The map contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

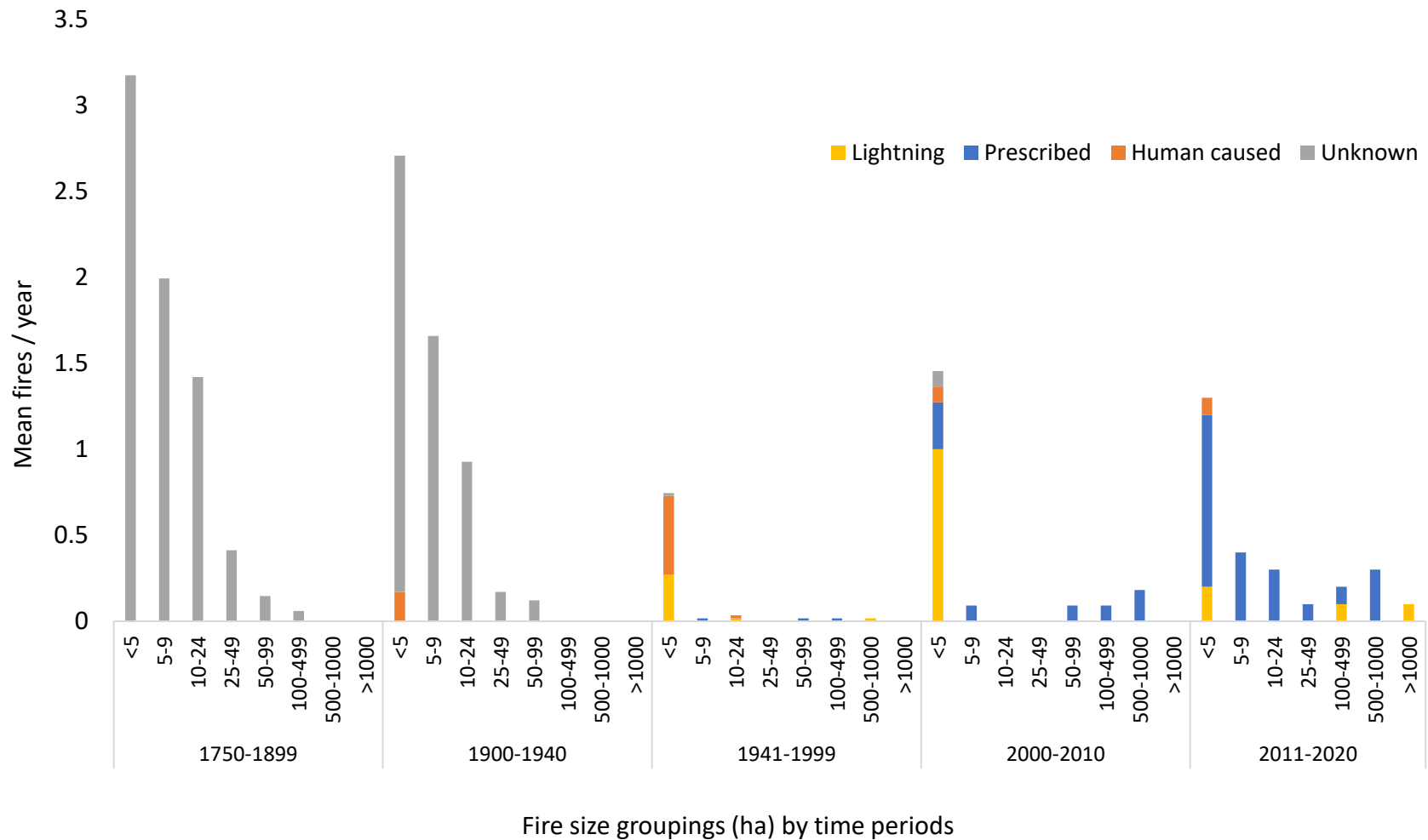


Fig. A.4. The mean number of fires (lightning caused, prescribed, human caused, and of unknown cause) in Waterton Lakes National Park, Alberta, per year for each group size (ha) and time period (1750 to 2020). The figure contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

Table A.1. The prescribed fires and wildfires that occurred during the years of the acoustic surveys (2015 to 2020) in Waterton Lakes National Park, Alberta. The table contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

Year and date of fires	Sites within 50 m of burn	Bat activity levels affected?	Years affected by fire*	
2015				
Prescribed fires				
March 13	Wishbone	Possible	2015 – 2020	
May 5	None	No		
2016				
Summit Knob Wildfire				
July 28	None		2016 – 2020	
Prescribed fires				
March 16	None			
April 2	Blakiston Roadside, Red Rock	Possible		
June 17	None	No		
September 1	None	No		
November 8	None	No		
November 10	None	No		
2017				
Kenow Wildfire				
August 30	Bison Paddock, Blakiston Roadside, Cameron Lake, Dipper at Rowe, Red Rock, Sewage Lagoon	Yes	2018 – 2020	
Prescribed fires				
April 11	None	No		
Mid April	None	No		
April 20	None	No		
2018				
Boundary Wildfire				
August 23	None	No	2019 – 2020	
2019				
No fires	NA	NA		
2020				
No fires	NA	NA		

*Acoustic detectors were deployed from approximately late June to early August, therefore if a fire occurred after this date, the effects would be seen in the following year.

Appendix B: Historic Mountain Pine Beetle Distribution in Waterton Lakes National Park, Alberta

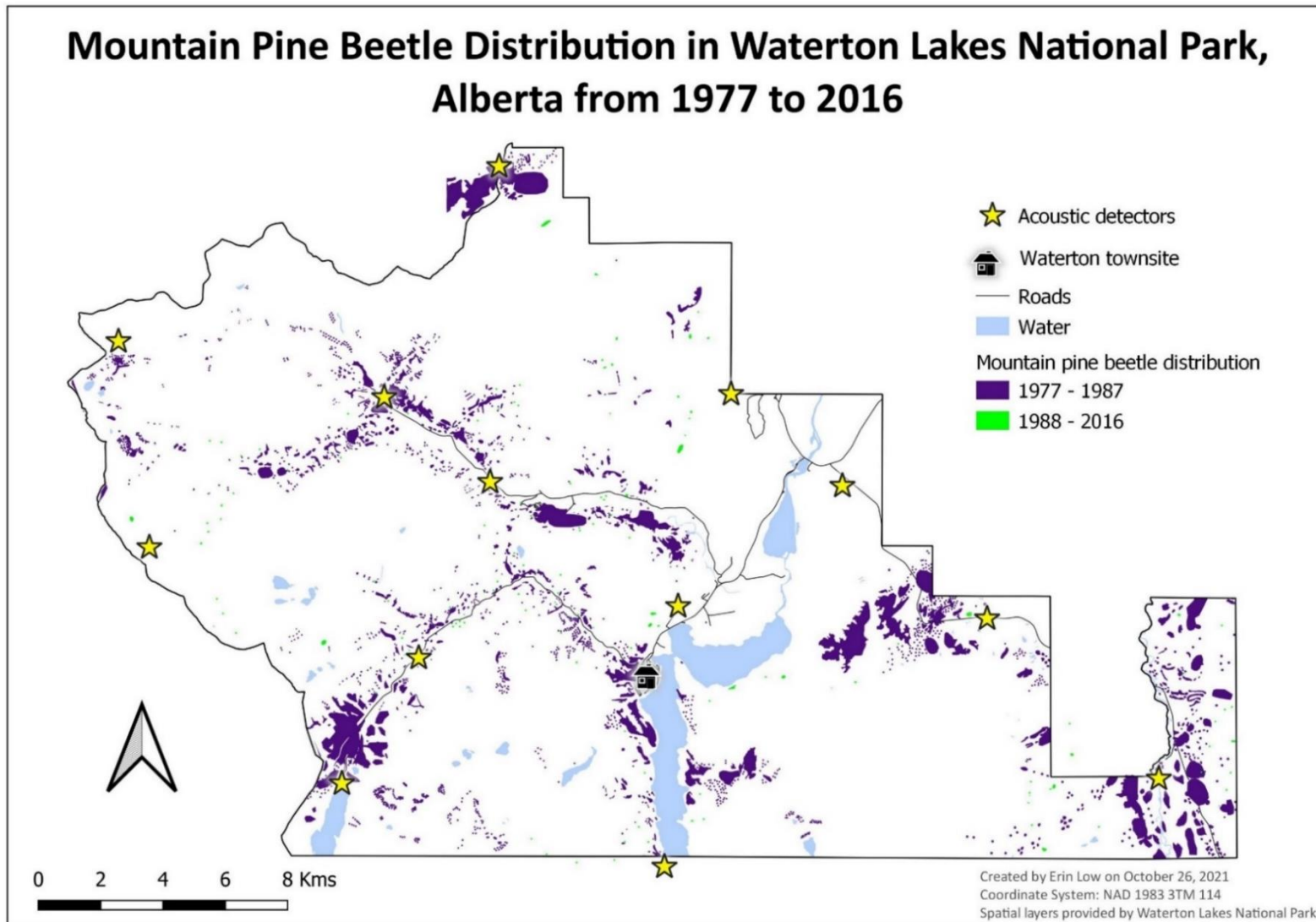


Fig. B.1. The mountain pine beetle (*Dendroctonus ponderosae*) distribution in Waterton Lakes National Park, Alberta from 1977 to 2016. The map contains information licensed from Parks Canada under the Open Government Licence – Canada (GOC 2019).

Appendix C: Metadata for the little brown *Myotis* radio-tracked in Waterton Lakes National Park, Alberta in 2019 and 2020

Table C.1. Metadata for the female little brown *Myotis* radio-tracked in Waterton Lakes National Park, Alberta in 2019 and 2020.

Bat ID	Capture date	Capture site name	Capture site UTM	Repro. condition*	Roost area	Confirmed roost or approximate area†	Roost type	Roost site UTM	Roost to capture site distance (km)‡	Emergence count
Bat 1	09-Jul-2019	Cameron Lake	11 U 715971 5433837	NOP	Cameron Lake	Approximate area	Likely tree	11 U 716636 5433529, 11 U 716594 5433452	0.7	NA
Bat 2	16-Jul-2020	Hay Barn	12 U 291820 5440280	NOP	Hay Barn	Confirmed roost	Tree	12 U 291792 5440111	0.2	NA
					Townsite	Confirmed roost	Building	Waterton townsite	approx. 4.5	100
Bat 3	24-Jun-2019	Knight's Lake	12 U 291823 5442783	NOP	Wishbone	Approximate area	Likely tree	12 U 294190 5442390, 12 U 294793 5441209, 12 U 294994 5441147	2.4 - 3.6	NA
					Townsite	Confirmed roost	Building	Waterton townsite	approx. 7	60
Bat 4	03-Jul-2019	Knight's Lake	12 U 291823 5442783	P	Townsite	Confirmed roost	Building	Waterton townsite	approx. 7	62
Bat 5	14-Jul-2019	Knight's Lake	12 U 291823 5442783	L	Townsite	Confirmed roost	Building	Waterton townsite	approx. 7	52
Bat 6	28-Jul-2019	Crandell Lake	12 U 283274 5441558	L	Townsite	Confirmed roost	Building	Waterton townsite	approx. 6	135
					Blakiston valley	Bearing area	Night roost/ foraging area	12 U 286722 5442566	3.6	NA
Bat 7	03-Aug-2019	Crandell Lake	12 U 283274 5441558	L	Townsite	Confirmed roost	Building	Waterton townsite	approx. 6	200
Bat 8	03-Aug-2019	Crandell Lake	12 U 283274 5441558	L	Townsite	Confirmed roost	Building	Waterton townsite	approx. 6	82

Bat 9	28-Jul-2019	Crandell Lake	12 U 283274 5441558	L	Blakiston valley	Approximate area	Night roost/ foraging area	12 U 288869 5439470	6	NA
Bat 10	09-Aug-2019	Highway bridge	12 U 292624 5443975	L	Highway bridge Blakiston valley	Confirmed roost	Tree	12 U 292532 5443947	0.1	NA
						Bearing area	Likely tree or rock	12 U 288334 5440175	Unknown	NA
Bat 11	23-Aug-2020	Cameron Lake	11 U 715971 5433837	NR	Cameron Lake	Confirmed roost	Tree	11U 715597 5434080, 11U 715649 5433630	0.4	1
						Bearing area	Likely tree or rock	11 U 714546 5431648, 11 U 714586 5432858	Unknown	NA
Bat 12	06-Jul-2019	Cameron Lake	11 U 715971 5433837	NOP	Cameron Lake	Unknown, only heard foraging late at night at Cameron Lake	Likely tree or rock	Unknown	Unknown	NA

* “NOP” refers to not obviously pregnant (i.e. caught early in the season and could not determine if pregnant), “P” refers to pregnant, “L” refers to lactating, and “NR” refers to females that were non-reproductive (i.e. not pregnant, lactating or post-lactating).

† Confirmed roosts are the exact roosts. Approximate areas are general areas where bats were tracked to but the exact roost could not be confirmed. Often the roost type could be determined from the approximate area based on what was available in the surrounding habitat. Bearing areas are broad areas where the signal was detected from but the roosting location was estimated using bearings only.

‡ Distances were calculated using the vector analysis Distance Matrix tool in QGIS (version 3.16.8-Hannover).

Table C.2. Metadata for the male little brown *Myotis* radio-tracked in Waterton Lakes National Park, Alberta in 2020.

Bat ID	Capture date	Capture site name	Capture site UTM	Roost date	Roost area	Burned/ unburned roost	Roost type*	Roost site UTM	Roost to capture site distance (km) [†]
Bat 13	09-Aug-2020	Lost Lake	11 U 708237 5447757	10-Aug-2020	Lost Lake	Unburned	Likely rock crevice	11U 707155 5448361	Unknown
Bat 13	09-Aug-2020	Lost Lake	11 U 708237 5447757	11-Aug-2020	Lost Lake	Unburned	Confirmed rock crevice	11 U 707156 5448402	1.3
Bat 13	09-Aug-2020	Lost Lake	11 U 708237 5447757	12-Aug-2020	Lost Lake	Unburned	Likely rock crevice	11 U 707147 5448225	Unknown
Bat 14	09-Aug-2020	Lost Lake	11 U 708237 5447757	10-Aug-2020	Lost Lake	Unburned	Confirmed rock crevice	11 U 707650 5448541	1.0
Bat 14	09-Aug-2020	Lost Lake	11 U 708237 5447757	11-Aug-2020	Lost Lake	Unburned	Confirmed rock crevice	11 U 707649 5448542	1.0
Bat 15	09-Aug-2020	Lost Lake	11 U 708237 5447757	10-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709327 5447597	1.1
Bat 15	09-Aug-2020	Lost Lake	11 U 708237 5447757	11-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709327 5447597	1.1
Bat 15	09-Aug-2020	Lost Lake	11 U 708237 5447757	12-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709327 5447597	1.1
Bat 15	09-Aug-2020	Lost Lake	11 U 708237 5447757	13-Aug-2020	Lost Lake	Unburned	Confirmed rock crevice	11 U 709334 5447316	1.2
Bat 15	09-Aug-2020	Lost Lake	11 U 708237 5447757	14-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709327 5447597	1.1
Bat 16	09-Aug-2020	Lost Lake	11 U 708237 5447757	10-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709541 5447522	1.3
Bat 16	09-Aug-2020	Lost Lake	11 U 708237 5447757	11-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709595 5447569	1.4
Bat 16	09-Aug-2020	Lost Lake	11 U 708237 5447757	12-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709496 5447657	1.3
Bat 16	09-Aug-2020	Lost Lake	11 U 708237 5447757	13-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709541 5447522	1.3
Bat 16	09-Aug-2020	Lost Lake	11 U 708237 5447757	14-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709627 5447610	1.4
Bat 17	09-Aug-2020	Lost Lake	11 U 708237 5447757	10-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709687 5447406	1.5
Bat 17	09-Aug-2020	Lost Lake	11 U 708237 5447757	11-Aug-2020	Lost Lake	Burned	Confirmed rock crevice	11 U 711045 5448474	2.9
Bat 17	09-Aug-2020	Lost Lake	11 U 708237 5447757	12-Aug-2020	Lost Lake	Burned	Confirmed rock crevice	11 U 710928 5448428	2.8

Bat 17	09-Aug-2020	Lost Lake	11 U 708237 5447757	13-Aug-2020	Lost Lake	Burned	Likely rock crevice	11 U 711045 5448474	Unknown
Bat 17	09-Aug-2020	Lost Lake	11 U 708237 5447757	14-Aug-2020	Lost Lake	Unburned	Confirmed tree	11 U 709687 5447406	1.5
Bat 18	09-Aug-2020	Lost Lake	11 U 708237 5447757	12-Aug-2020	Lost Lake	Burned	Confirmed boulder	11 U 712327 5448659	4.2
Bat 18	09-Aug-2020	Lost Lake	11 U 708237 5447757	13-Aug-2020	Lost Lake	Burned	Confirmed boulder	11 U 712327 5448659	4.2
Bat 18	09-Aug-2020	Lost Lake	11 U 708237 5447757	14-Aug-2020	Lost Lake	Burned	Confirmed boulder	11 U 712327 5448659	4.2
Bat 19	23-Aug-2020	Cameron Lake	11 U 715971 5433837	28-Aug-2020	Cameron Lake	Burned	Bearing area	11 U 715601 5434780	Unknown
Bat 19	23-Aug-2020	Cameron Lake	11 U 715971 5433837	29-Aug-2020	Cameron Lake	Burned	Bearing area	11 U 715407 5434398	Unknown
Bat 20	23-Aug-2020	Cameron Lake	11 U 715971 5433837	24-Aug-2020	Cameron Lake	Burned	Confirmed tree	11 U 715895 5434307	0.5
Bat 20	23-Aug-2020	Cameron Lake	11 U 715971 5433837	26-Aug-2020	Cameron Lake	Burned	Confirmed tree	11 U 715784 5434149	0.4
Bat 20	23-Aug-2020	Cameron Lake	11 U 715971 5433837	27-Aug-2020	Cameron Lake	Burned	Confirmed tree	11 U 715784 5434149	0.4
Bat 20	23-Aug-2020	Cameron Lake	11 U 715971 5433837	28-Aug-2020	Cameron Lake	Burned	Confirmed tree	11 U 715690 5434251	0.5
Bat 20	23-Aug-2020	Cameron Lake	11 U 715971 5433837	29-Aug-2020	Cameron Lake	Burned	Likely tree	11 U 715611 5434006	Unknown
Bat 21	23-Aug-2020	Cameron Lake	11 U 715971 5433837	26-Aug-2020	Lineham trail	Unknown	Bearing area	11 U 718108 5438858	Unknown
Bat 22	20-Jul-2020	Crandell Lake	12 U 283274 5441558	21-Jul-2020	Crandell Lake	Burned	Bearing area	12 U 283230 5442214	Unknown
Bat 22	20-Jul-2020	Crandell Lake	12 U 283274 5441558	22-Jul-2020	Crandell Lake	Burned	Bearing area	12 U 283230 5442214	Unknown
Bat 22	20-Jul-2020	Crandell Lake	12 U 283274 5441558	23-Jul-2020	Crandell Lake	Burned	Bearing area	12 U 283230 5442214	Unknown
Bat 22	20-Jul-2020	Crandell Lake	12 U 283274 5441558	24-Jul-2020	Crandell Lake	Burned	Likely rock crevice	12 U 282710 5442097	0.8
Bat 22	20-Jul-2020	Crandell Lake	12 U 283274 5441558	25-Jul-2020	Crandell Lake	Burned	Likely rock crevice	12 U 283007 5441426	0.3
Bat 23	20-Jul-2020	Crandell Lake	12 U 283274 5441558	21-Jul-2020	Crandell Lake	Burned	Bearing area	12 U 283230 5442214	Unknown
Bat 23	20-Jul-2020	Crandell Lake	12 U 283274 5441558	22-Jul-2020	Crandell Lake	Burned	Confirmed tree	12 U 282683 5443026	1.6
Bat 23	20-Jul-2020	Crandell Lake	12 U 283274 5441558	23-Jul-2020	Crandell Lake	Burned	Confirmed tree	12 U 282636 5443092	1.7

Bat 23	20-Jul-2020	Crandell Lake	12 U 283274 5441558	24-Jul-2020	Crandell Lake	Burned	Confirmed tree	12 U 282683 5443026	1.6
Bat 23	20-Jul-2020	Crandell Lake	12 U 283274 5441558	25-Jul-2020	Crandell Lake	Burned	Confirmed tree	12 U 282631 5443040	1.6
Bat 24	20-Jul-2020	Crandell Lake	12 U 283274 5441558	21-Jul-2020	Crandell Lake	Burned	Bearing area	12 U 283230 5442214	Unknown
Bat 24	20-Jul-2020	Crandell Lake	12 U 283274 5441558	22-Jul-2020	Crandell Lake	Burned	Bearing area	12 U 283230 5442214	Unknown
Bat 24	20-Jul-2020	Crandell Lake	12 U 283274 5441558	23-Jul-2020	Crandell Lake	Burned	Confirmed tree	12 U 283016 5442670	1.1
Bat 25	20-Jul-2020	Crandell Lake	12 U 283274 5441558	21-Jul-2020	Blakiston valley	Unknown	Bearing area	12 U 288364 5440614	Unknown
Bat 26	16-Jul-2020	Hay Barn	12 U 291820 5440280	18-Jul-2020	Cameron valley	Burned	Bearing area	12 U 286655 5437431	Unknown
Bat 26	16-Jul-2020	Hay Barn	12 U 291820 5440280	19-Jul-2020	Cameron valley	Burned	Likely rock crevice	12 U 286710 5437879	5.6
Bat 27	16-Jul-2020	Hay Barn	12 U 291820 5440280	17-Jul-2020	Hay barn	Unburned	Confirmed tree	12 U 292267 5440251	0.4
Bat 27	16-Jul-2020	Hay Barn	12 U 291820 5440280	18-Jul-2020	Hay barn	Unburned	Bearing area	12 U 292267 5440251	Unknown
Bat 27	16-Jul-2020	Hay Barn	12 U 291820 5440280	19-Jul-2020	Hay barn	Unburned	Bearing area	12 U 292267 5440251	Unknown

* Confirmed roosts are the exact roosts. Likely roosts are when the bat was tracked to an approximate roosting area and the surrounding habitat strongly suggested a certain roost type. Bearing areas are broad areas where the signal was detected from, but the roosting location was estimated using bearings only.

† Distances were calculated using the vector analysis Distance Matrix tool in QGIS (version 3.16.8-Hannover).

Appendix D: Metadata for the Acoustic Detector Sites, the Waterton Townsite, and the Weather Stations

Table D.1. Site metadata for the Waterton townsite and each acoustic detector site monitored from 2015 to 2020 in Waterton Lakes National Park, Alberta.

Acoustic site	Coordinates (NAD 83)	Elevation (m)	Natural subregion	Burn/unburn after wildfire	Townsite distance (km)	Distance to water (m)	Nearest water feature	Habitat description
Sewage Lagoon	12U 288320 5439394	1318	Foothills Parkland	Burn	2	5	Manmade “Lake”	Wastewater treatment lagoon, surrounded by buildings and sparse mixed forest
Bison Paddock	12U 290288 5446141	1357	Foothills Parkland	Burn	9	20	Wetland	Prairie field near wetland, surrounded by deciduous vegetation
Blakiston Roadside	12U 282448 5443621	1428	Montane	Burn	8	20	Wetland	Grassy meadow on hill above wetland, surrounded by sparse mixed forest
Red Rock	11U 716829 5446319	1503	Montane	Burn	12.5	60	Red Rock Creek	Grassy meadow near to canyon, surrounded by mixed forest
Dipper at Rowe	11U 718267 5437995	1589	Montane	Burn	7.5	30	Cameron Creek	Coniferous forest trail near stream
Cameron Lake	11U 715961 5433861	1670	Lower Subalpine	Burn	10.5	10	Wetland	Wetland surrounded by coniferous forest
Wishbone	12U 293743 5443049	1289	Foothills Parkland	Unburn	8.5	350	Wetland	Grassy meadow surrounded by deciduous forest
Waterton townsite	12U 287451 5437358	1282	Montane	Unburn	0	Surrounded by water	Waterton Lake	Townsite with a population of approximately 100 people
Belly Bend	12U 303521 5433244	1378	Montane	Unburn	16.5	5	Belly River	River surrounded by deciduous vegetation
Yarrow Cabin	12U 283197 5453750	1520	Montane	Unburn	17	75	Yarrow Creek	Grassy meadow near to gravel road, surrounded by deciduous forest
Sofa Burn	12U 298216 5438607	1534	Montane	Unburn	11	10	Wetland	Wetland surrounded by sparse mixed forest
Boundary Bay	12U 287550 5431047	1284	Lower Subalpine	Unburn	6.5	20	Boundary Creek	Dry riverbed surrounded by mixed forest
Lost Lake	11U 708233 5447777	1897	Upper Subalpine	Unburn	20	10	Lost Lake	Lake, surrounded by coniferous forest
Lone Lake	11U 709485 5441200	2033	Upper Subalpine	Unburn	16.5	10	Lone Lake	Lake, surrounded by coniferous forest

Table D.2. Site metadata for each weather station used to compile environmental data for the thirteen different acoustic detector sites in Waterton Lakes National Park, Alberta.

Weather station	Associated acoustic site(s)	Acoustic site to weather station distance (km)	Weather station coordinates (NAD 83)	Weather station elevation (m)	Weather station operator
Akamina 2	Cameron Lake	1.5	11U 715613 5435351	1676	Environment and Parks
	Dipper at Rowe	3.5			
Chief Customs	Belly Bend	3.0	12U 305576 5430894	1690	Environment and Parks
Goathaunt	Boundary Bay	4.5	12U 288425 5426669	1280	Environment and Parks
Waterton Park Gate	Bison Paddock	5.5	12U 295760 5446050	1296	Environment Canada
	Sewage Lagoon	10.0			
	Sofa Burn	8.0			
	Wishbone	3.5			
	Yarrow Cabin	15.0			
Waterton Red Rock	Blakiston Roadside	5.0	11U 716398 5446513	1524	Environment and Parks
	Lone Lake	8.5			
	Lost Lake	8.5			
	Red Rock	0.5			

Appendix E: Acoustic Detector Deployment and Call Processing Methodology

E.1 Acoustic Detector Deployment

The locations of the acoustic detectors used in the 2015 to 2020 study in WLNP, were selected using a probability-based sampling design, based on the North American Bat Monitoring Program (NABat) sampling structure. NABat is a collaborative monitoring program across North America that uses standardized protocols to collect acoustic bat data (Loeb et al. 2015). These data help assess population trends and inform conservation-based strategies to support bats due to their vulnerability to a variety of threats (e.g. white-nose syndrome, wind energy development, climate change, habitat loss) (Loeb et al. 2015). The NABat sampling design divides North America into a series of 10 by 10 km grid cells, which are further divided into four, five by five km quadrants within each cell (Loeb et al. 2015). WLNP staff followed the recommendations from Gros Morne National Park, Newfoundland, to increase the number of acoustic recording sites for parks with small areas (Parks Canada 2017a). Within the quadrants of the cells, the exact location of the detector was chosen based on bat survey work that had been completed in previous years (Lausen 2012, Parks Canada 2017a).

The following specifications regarding detector deployment setup and call processing were provided by H. Mahoney, pers. comm. Wildlife Acoustics' Song Meter SM2BAT+ detectors and SMX-U1 omnidirectional ultrasonic microphones were used throughout the six-year study. The detectors were deployed using poles, with the microphone oriented at a 45° angle downward, and raised five to six meters from the ground. The microphones were calibrated prior to the start of each field season to ensure they were working within normal parameters, and that the sensitivities were approximately equivalent between years. The detectors' recording settings were confirmed prior to each site deployment to ensure they were programmed correctly and would be recording during the appropriate times and with the appropriate settings.

The detectors were programmed to record from 30 minutes before sunset to 30 minutes after sunrise. The files were recorded in full spectrum in WAC0 or WAV file format (depending on the year), the sample rate was set to 384 kHz, the high pass filter was set to eight kHz (FS/48) and the low pass filter was off. The trigger level (i.e. signal to noise ratio) was

set to 18 dB, the trigger window was set to two seconds, and the trigger max length was set to zero seconds. The detectors' amplifier configuration switches were set to include: a 2.5V bias, 1000 Hz high-pass filter, and 12 dB gain for the microphone preamplifier.

E.2 Acoustic Call Processing

At the end of each field season, the raw acoustic files were processed using Wildlife Acoustics' Kaleidoscope Pro bat analysis software (version 3.1.1, www.wildlifeacoustics.com/products/kaleidoscope-pro). Both processed full-spectrum WAV files and zero-cross files were created from the raw WAC0 and WAV files. Full spectrum recordings are a digitized representation of the entire soundscape, which includes frequency, time, harmonic and amplitude information for each call (Szewczak 2010, Agranat 2013). A zero-cross recording is created by measuring the amount of time it takes a soundwave to cross the 'zero' mark a given number of times (Szewczak 2010, Agranat 2013). The time between the oscillations, or typically the time it takes for eight zero-crossings (called the division ratio), is recorded (Szewczak 2010). These data are then plotted as the average frequencies per time (Szewczak 2010). Zero-cross files can only display the dominant frequency at any one time, and only the frequency and time of each call (i.e. no amplitude or harmonic information) (Szewczak 2010, Agranat 2013).

During the acoustic processing, files were filtered to keep passes that had calls with frequencies between eight and 120 kHz, and durations between two and 500 ms, and had a minimum of two calls. Files were split to a maximum duration of 15 seconds and the zero-cross files created had a division ratio of eight. The processed full-spectrum and zero-cross files were dispersed into nightly subdirectories for each site and year. The advanced signal enhancement option was used and the files that did not pass the filter were automatically moved to a separate directory called "NOISE". The Bats of North America 3.1.0 classifier was used for auto-identification. Within the classifier, the little brown *Myotis* (*Myotis lucifugus*), long-eared *Myotis* (*Myotis evotis*), long-legged *Myotis* (*Myotis volans*), western small-footed *Myotis* (*Myotis ciliolabrum*), Yuma *Myotis* (*Myotis yumanensis*), California *Myotis* (*Myotis californicus*), big brown bat (*Eptesicus fuscus*), silver-haired bat (*Lasionycteris noctivagans*), eastern red bat (*Lasiurus borealis*), and hoary bat (*Lasiurus cinereus*) were included. The classifier was set to a

sensitivity of “+1 More Accurate (Conservative)”. All files were labelled with one of the species included in the classifier, or the additional categories of “NoID” (i.e. likely a bat pass but could not determine species) and “Noise” (i.e. likely not a bat pass). For all years included in the study, Parks Canada staff processed the raw acoustic bat files.

Once the bat files were processed in Kaleidoscope Pro, I inspected all of the data log files for each site and year. The data log files were automatically generated for every night a detector was deployed, recording the date, time, internal temperature, and external sensor readings every five minutes during the recording period. Through these data log files, I was able to confirm whether a specific detector was recording normally throughout the deployment period. I excluded nights that I suspected had recording issues (i.e. microphone concerns or were not recording during the appropriate times). In addition, I excluded nights when the detector stopped recording more than 30 minutes before sunrise. Although past research has shown that bat activity decreases with increased precipitation and low temperatures (Vonhof 2006), bats were detected on nights with low sunset temperatures (minimum temperature of 5.2 °C) and relatively high amounts of precipitation (maximum nightly precipitation of 19.7 mm). Therefore, I did not exclude nights due to environmental conditions and instead used sunset temperature and nightly precipitation as covariates in my models.

I compiled metadata for the surrounding habitat for each acoustic site and the Waterton townsite, as well as environmental conditions for each night a detector was deployed (Appendix D). For each site I compiled: the elevation (determined from Google Earth Pro, version 7.3.3.7786), natural subregion (determined using spatial data provided by Parks Canada), forest type (I recorded habitat information during site visits), distance to the Waterton townsite (determined using the point distance analysis tool in ArcGIS Desktop 10.7.1), distance to and type of nearest water source (determined during site visits, from Google Earth Pro, and using the Alberta River Basins web application [i.e. GOA 2021]), as well as whether it became burned or remained unburned after the wildfire (determined through spatial data provided by Parks Canada). I also recorded habitat descriptions for each site. Population census data for the Waterton townsite was determined using the Alberta Regional Dashboard web application (i.e. GOA 2020). Weather data were acquired from the Alberta Climate Information Service, Current

and Historical Alberta Weather Station Data Viewer (i.e. ACIS 2020). Five stations were used to represent the thirteen acoustic detector locations. The mean distance from an acoustic site to its associated weather station was six kilometers (range of 0.5 to 15 kms; determined using the point distance analysis tool in ArcGIS Desktop). The weather stations recorded hourly temperature and precipitation data. For temperature, I used the hourly “instantaneous ambient air temperature (°C) measured two meters above the ground surface”. This option records the temperature computed at the start of each hour. For precipitation, I used the hourly “precipitation totals (mm)”. This option records the hourly precipitation accumulated throughout the previous hour. To calculate the temperature at sunset for each night, I first recorded the sunset times in WLNP (i.e. Maplogs 2021). All sunset times throughout the study period were between 21:00 and 22:00. Therefore, using the temperatures recorded for 21:00 and 22:00, I used the sunset minute times to determine the percent value between the two temperatures, and thus the resulting approximate sunset temperature. To determine the total sum of precipitation for each night, I included the precipitation received from 21:00 to 6:00.

To analyze the acoustic files, I used Titley Scientific’s AnalookW bat analysis software (version 4.5s) and filters provided by J. Rae and C. Lausen, pers. comm. Similar to the auto-identification option in Kaleidoscope Pro, AnalookW has the option of creating and loading filters to aid with bat call analysis. The filters enable the user to create a list of specific call parameters (e.g. call frequency, slope, duration) to search for in each file. If the bat pass in the file is within those parameters, the file passes the filter and is marked differently compared to files that are rejected by the filter.

I took the auto-identified zero-cross files created from Kaleidoscope Pro and ran a filter in AnalookW to further separate bat calls from noise. From this process, an ANL list file was also created. An ANL list file is a text file that lists all of the input files and where they are loaded from, and ‘virtually’ groups all files from all years by the species label that was assigned from Kaleidoscope Pro. Once the files were dispersed by species, I ran additional species/species groups’ filters on all of the files as a second auto-identification check before manually analyzing the files. I ran a hoary bat filter on the files Kaleidoscope Pro identified as hoary bats; if a file did not pass the filter, I reassigned the file to “low frequency”. I ran a big brown/silver-haired bat

filter on the files Kaleidoscope Pro identified as big brown bats or silver-haired bats; if a file did not pass the filter, I reassigned the file to “low frequency”. I ran a long-eared *Myotis* filter on the files Kaleidoscope Pro identified as long-eared *Myotis*; if a file did not pass the filter, I reassigned the file to “high frequency”. I ran an eastern red bat filter on the files Kaleidoscope Pro identified as eastern red bat; if a file did not pass the filter, I reassigned the file to “high frequency”. I ran a 40kHz *Myotis* filter on the files Kaleidoscope Pro identified as little brown *Myotis*, long-legged *Myotis*, and western small-footed *Myotis*; if a file did not pass the filter, I reassigned the file to “high frequency”. I ran a Yuma *Myotis*/California *Myotis* filter on the files Kaleidoscope Pro identified as Yuma *Myotis* and California *Myotis*; if a file did not pass the filter, I reassigned the file to “high frequency”. I ran both high and low frequency filters on the files Kaleidoscope Pro identified as “NoID”.

Once the files were filtered and redispersed, I manually identified all of the files. Manual species identification was based on resources provided from bat acoustic training courses I attended (i.e. Lausen 2016, pers. comm., Szewczak 2018). As there is no WLNP-specific bat echolocation reference library, I was conservative in assigning species specific identification and thus grouped species with similar echolocation characteristics (e.g. high frequency, 40kHz *Myotis*, big brown/silver-haired bat, low frequency). To ensure I was consistent, I analyzed all files from all years for a given species/species group at the same time. The echolocation call characteristics I used for each species/species group and sonogram examples are illustrated in Appendix F.

Files needed a minimum of two calls be considered a bat echolocation pass. In addition, the calls needed to occur within a reasonable time interval of one another. Most search phase calls occur at a call rate of approximately 3 to 12 calls per second (Griffin et al. 1960, Fenton 2013), therefore if the calls in the bat file were spaced at greater intervals than this, or very irregular intervals, it was not considered to be a bat. The shape of the call also needed to be indicative of a bat echolocation call, with the call typically starting at a high frequency and then sweeping down to a lower frequency (i.e. frequency modulated) (Fenton 2013). If only two calls were present, the file was identified as either high or low frequency. For files with three calls or more, echolocation passes could be assigned to a specific species/species group. However, due

to how diagnostic most hoary bat calls are, I was confident in assigning a species identification to hoary bat files containing only two calls. For all species, if a file appeared to contain only one call, the file was marked as noise. Some files had multiple species recorded. Each species or species group was identified and accounted for. Files that appeared to have multiple individuals of the same frequency class were not corrected for, and were therefore only counted as one pass.

Although I analyzed files for species using the zero-cross files created from Kaleidoscope Pro, I also used Wildlife Acoustics' Kaleidoscope Lite bat analysis software (version 5.4.0) to examine a subset of the processed full spectrum WAV files created from Kaleidoscope Pro. AnalookW only recognizes zero-cross files, however, Kaleidoscope Lite is able to load both zero-cross and full spectrum files. For full spectrum files to be zero-crossed, the signal to noise ratio (i.e. the difference between the detected signal and background noise) needs to be above a certain threshold to be converted. Therefore, many low amplitude calls that appear in full-spectrum are absent in zero-cross files. Due to feeding buzzes having low intensity (Szewczak 2018), they rarely appeared in the zero-cross files. I used Kaleidoscope Lite to determine the number of feeding buzzes in a subsample of files I manually identified in AnalookW as 40 kHz *Myotis*. Due to the quantity of data, I randomly selected 20 individual files or 10% of all files, whichever was greater in RStudio (version 1.3.1093, R Core Team 2020; package *dplyr*, Wickham et al. 2021) for each date for every site for each year. For each subsampled file, I manually identified whether a feeding buzz was present in the pass primarily by visually inspecting the file. I also listened to segments of the echolocation pass if it was unclear whether a feeding buzz was present. Bats also emit buzzes when drinking (Griffiths 2013, Russo et al. 2016) or landing (Russo et al. 2007), but the abrupt change/drop in frequency associated with feeding buzzes is typically absent in drinking and landing buzzes (Kalko 1995, Schnitzler and Kalko 2001, Jakobsen and Surlykke 2010, Russo et al. 2018a). Therefore, only buzzes with a drop in frequency were considered feeding buzzes. If a file contained more than one feeding buzz, it was still only counted once. I then determined the proportion of files with a feeding buzz (number of files with a feeding buzz/total number of files subsampled) for each date for every site for each year.

Appendix F: Bat Echolocation Call Characteristics

Table F.1. Echolocation call characteristics for the different bat species/species groups found in Waterton Lakes National Park, Alberta. Manual species identification was based on resources provided from bat acoustic training courses I attended (i.e. Lausen 2016, pers. comm., Szewczak 2018). The characteristic echolocation frequency (Fc) is defined as the frequency of the call at its lowest slope toward the end of the call (Szewczak 2018).

Common name or frequency group	Scientific name	Label	Echolocation call characteristics	Example
Hoary bat	<i>Lasiurus cinereus</i>	LACI	Low frequency calls with a Fc usually between 18 – 22 kHz. Search phase calls often have a shallow slope and a minimum frequency as low as 15 kHz. Also produce calls that have a variable and random Fc across the pass.	Fig. F.1.
Big brown/silver-haired bat	<i>Eptesicus fuscus</i> / <i>Lasionycteris noctivagans</i>	EPFULANO	Low frequency calls with a Fc usually between 25 – 30 kHz.	Fig. F.2.
Low frequency bat	-	LowF	Low frequency calls that have a Fc below 30kHz that are non-diagnostic to a species or a specific frequency group (e.g. non-diagnostic LACI, EPFULANO).	Fig. F.3.
30 kHz bat	-	30k	Sequences that have few, non-diagnostic pulses with a Fc of 30 kHz. They could belong to MYEV or be approach phase calls of a LowF bat.	Fig. F.4.
Long-eared <i>Myotis</i>	<i>Myotis evotis</i>	MYEV	High frequency calls with a Fc usually between 30 – 35 kHz. Calls have a steep slope with an almost linear call shape and sweep over a broad range of frequencies.	Fig. F.5.
Eastern red bat	<i>Lasiurus borealis</i>	LABO	High frequency calls with a Fc usually between 35 – 40 kHz. Search phase calls often have a shallow slope and a Fc as low as 30 kHz. Also produce calls that have a variable and random Fc across the pass.	Fig. F.6.
40 kHz <i>Myotis</i>	<i>Myotis spp.</i>	40kMyotis	High frequency calls with a Fc between 35 – 45 kHz, that could not be LABO. Species included in this group are the little brown <i>Myotis</i> (<i>Myotis lucifugus</i>), long-legged <i>Myotis</i> (<i>Myotis volans</i>), and non-diagnostic MYEV.	Fig. F.7.
High frequency bat	-	HighF	High frequency calls that have a Fc above 30kHz that are non-diagnostic to a species or a specific frequency group (e.g. non-diagnostic MYEV, LABO, <i>Myotis</i> spp.).	Fig. F.8.
Not a bat	-	NOTBAT	Ultrasonic sounds that were recorded that are not bats. Northern flying squirrels (<i>Glaucomys sabrinus</i>), insects, as well as ambient noise (wind, rain etc.) commonly create ultrasonic sounds.	Fig. F.9.

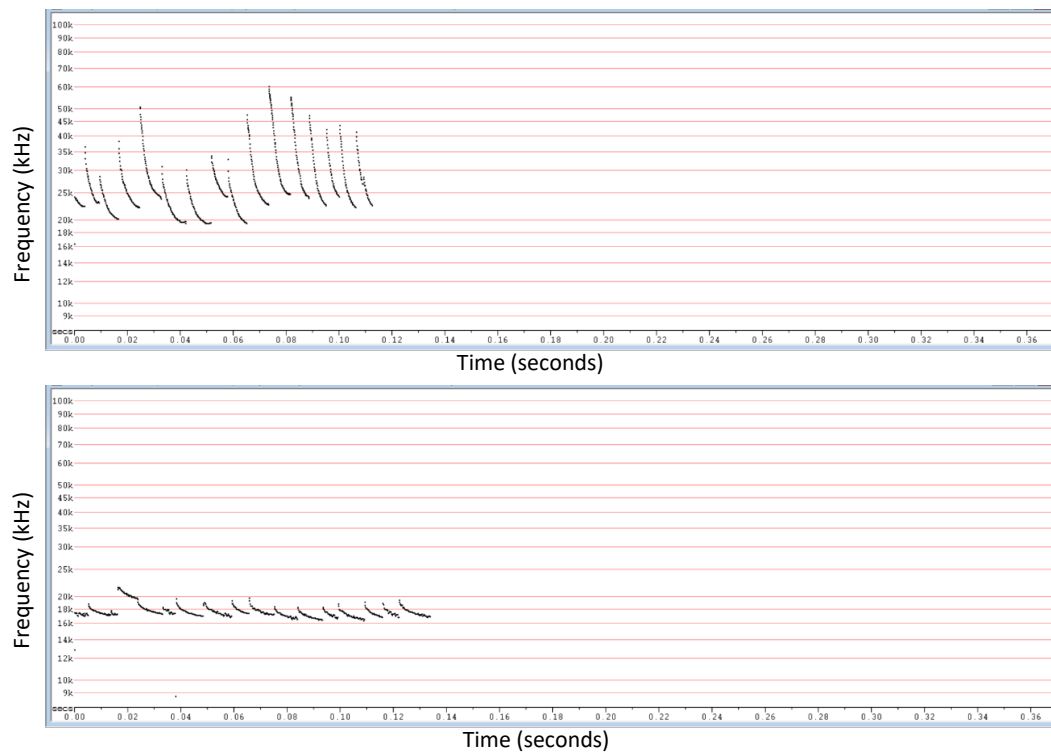


Fig. F.1. Examples of hoary bat passes. The sonograms are displayed in Titley Scientific's AnalookW bat analysis software (version 4.5s) in F7 in compressed time.

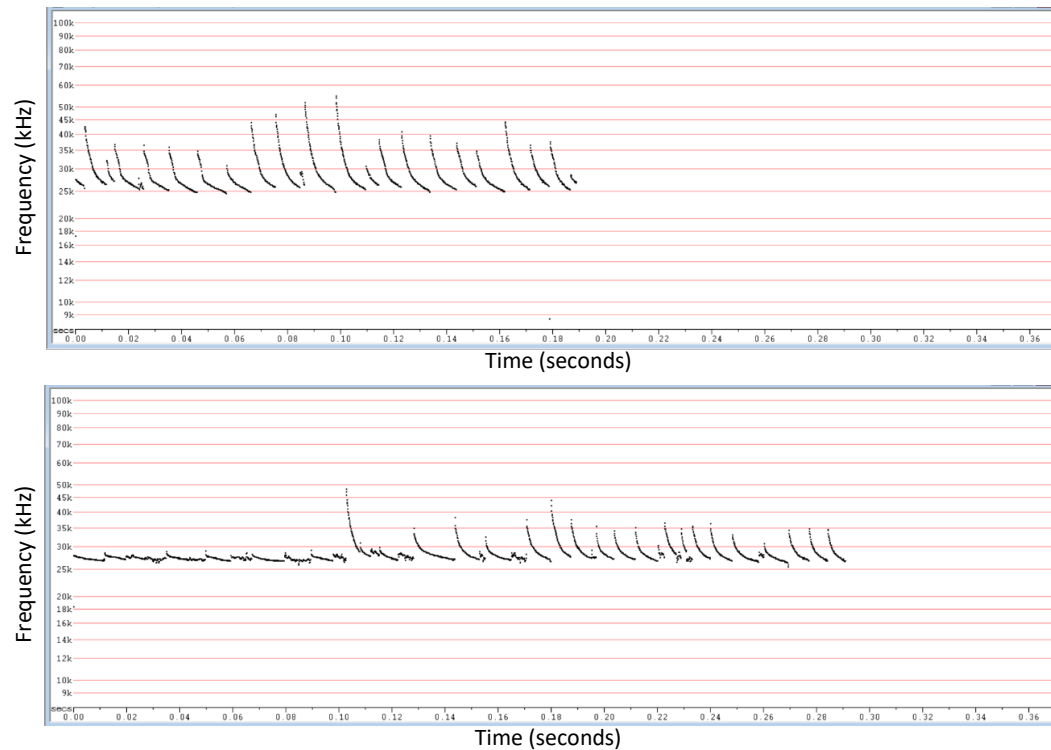


Fig. F.2. Examples of big brown/silver-haired bat passes. The sonograms are displayed in Titley Scientific's AnalookW bat analysis software (version 4.5s) in F7 in compressed time.

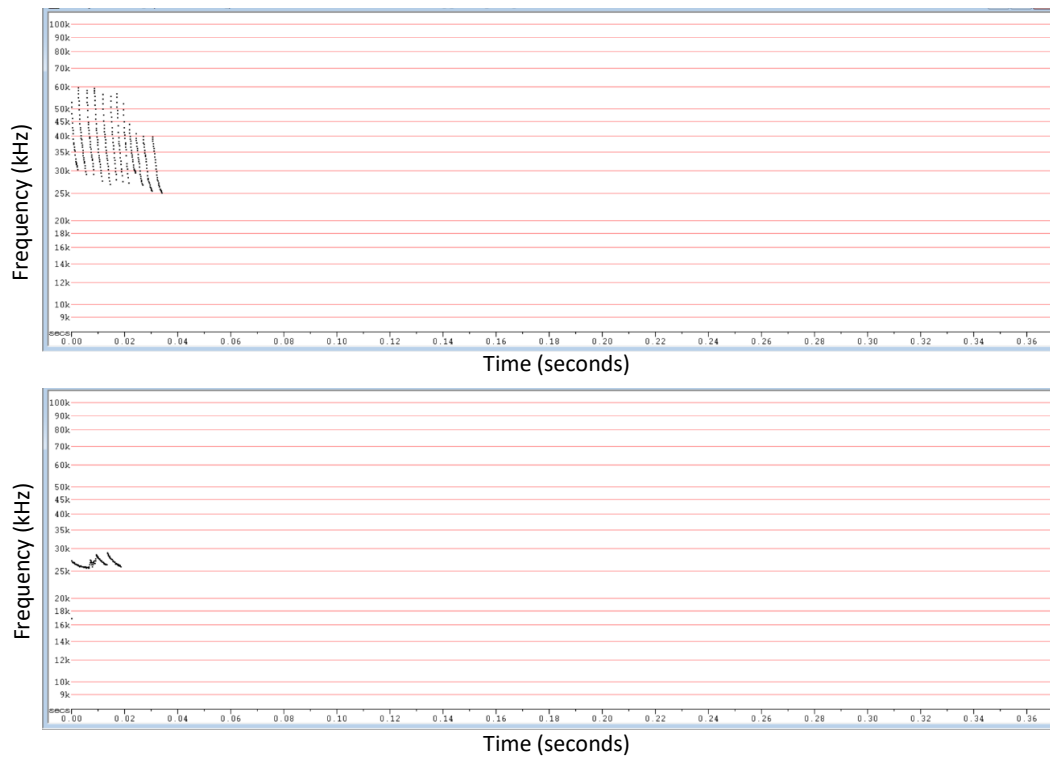


Fig. F.3. Examples of bat passes labeled as “LowF”. The sonograms are displayed in Titley Scientific’s AnalookW bat analysis software (version 4.5s) in F7 in compressed time.

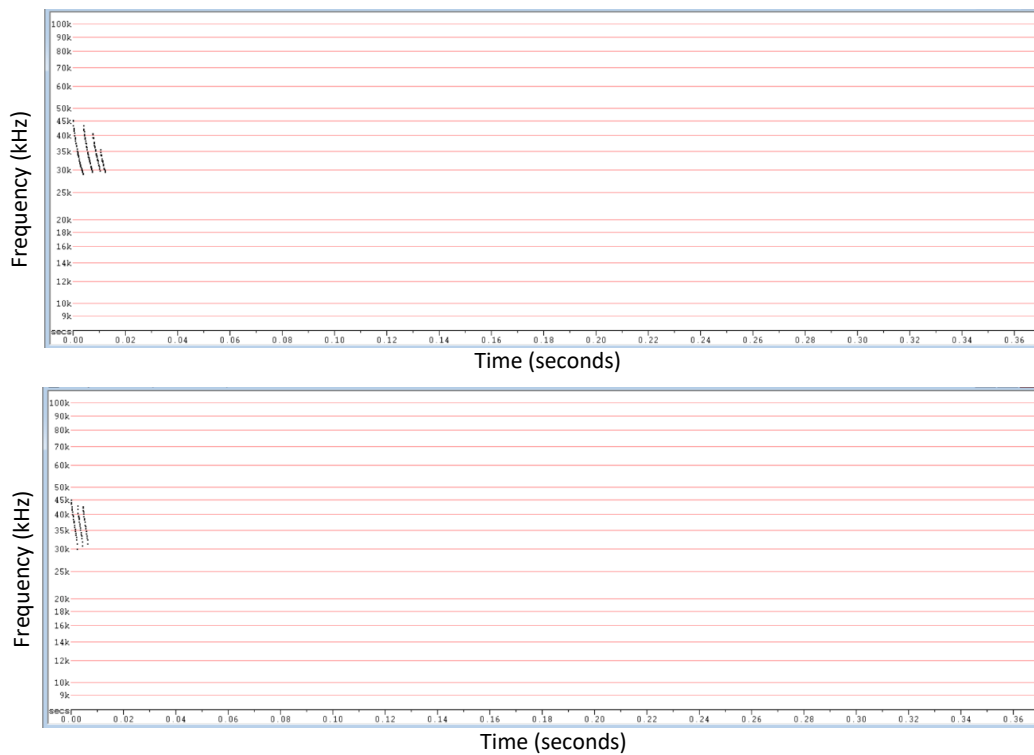


Fig. F.4. Examples of bat passes labeled as “30k”. The sonograms are displayed in Titley Scientific’s AnalookW bat analysis software (version 4.5s) in F7 in compressed time.

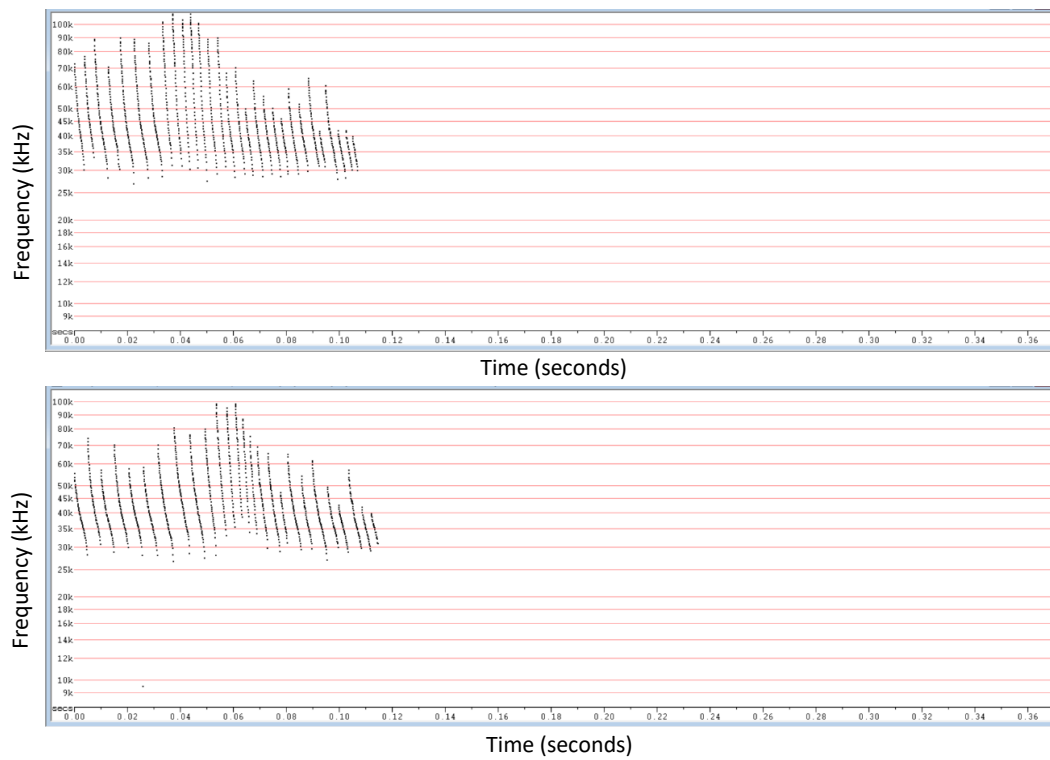


Fig. F.5. Examples of long-eared *Myotis* passes. The sonograms are displayed in Titley Scientific's AnalookW bat analysis software (version 4.5s) in F7 in compressed time.

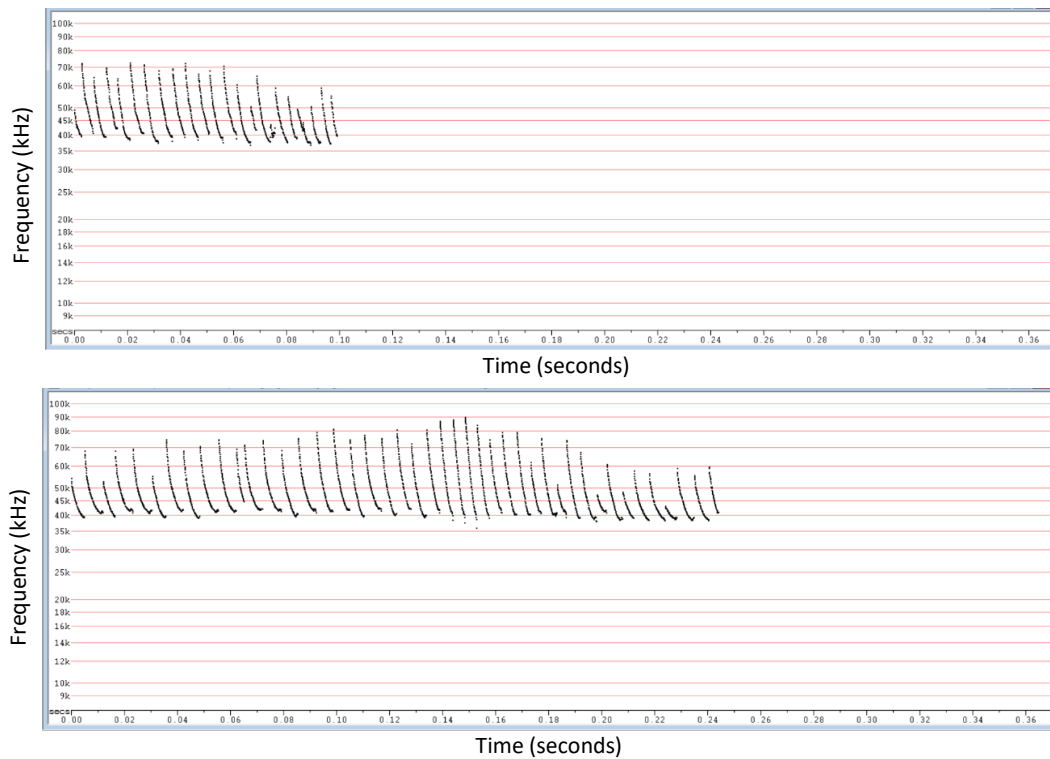


Fig. F.6. Examples of eastern red bat passes. The sonograms are displayed in Titley Scientific's AnalookW bat analysis software (version 4.5s) in F7 in compressed time.

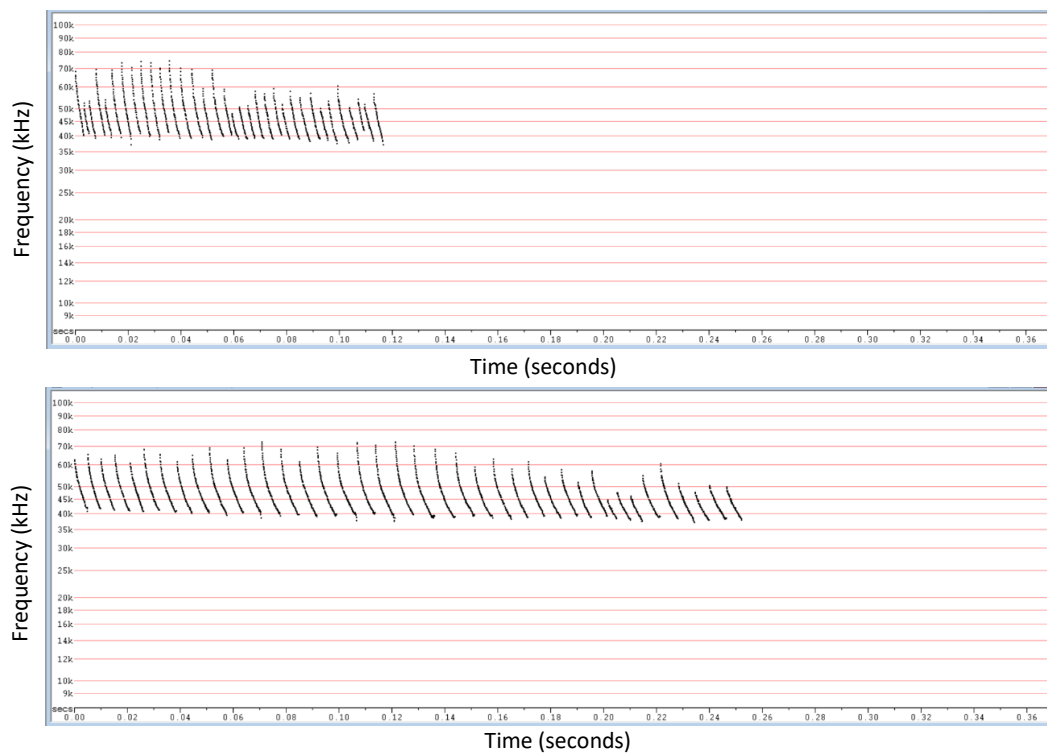


Fig. F.7. Examples of 40 kHz *Myotis* passes. The sonograms are displayed in Titley Scientific's AnlookW bat analysis software (version 4.5s) in F7 in compressed time.



Fig. F.8. Examples of bat passes labeled as "HighF". The sonograms are displayed in Titley Scientific's AnlookW bat analysis software (version 4.5s) in F7 in compressed time.

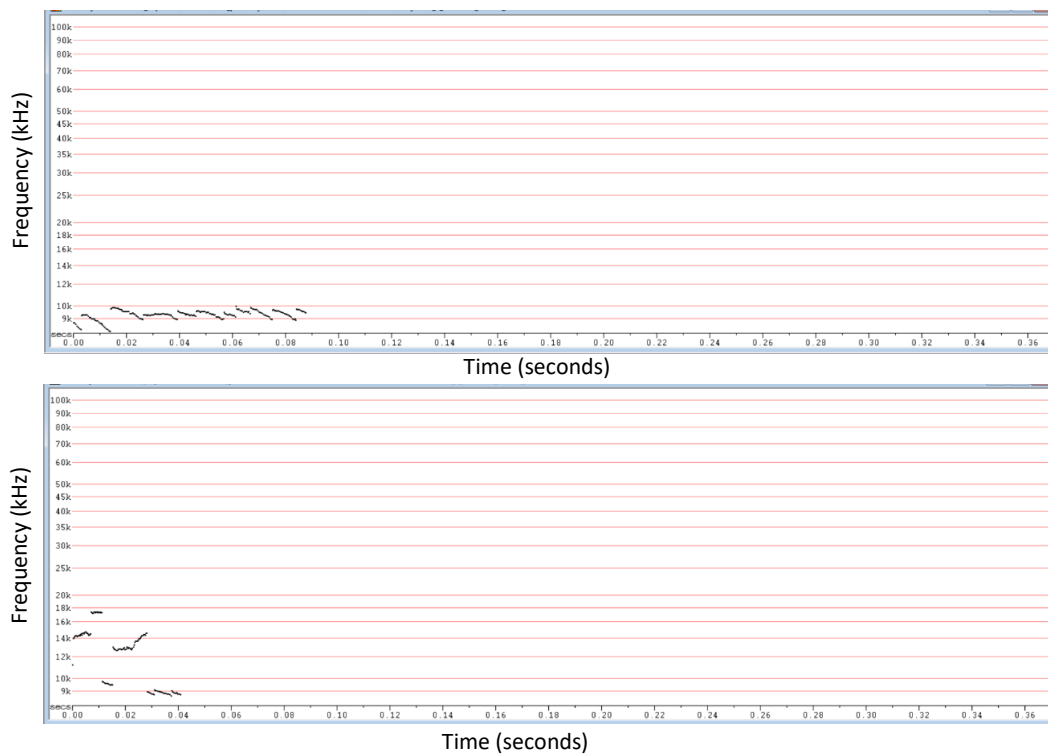


Fig. F.9. Examples of bat passes labeled as "NOTBAT". The sonograms are displayed in Titley Scientific's AnlookW bat analysis software (version 4.5s) in F7 in compressed time.

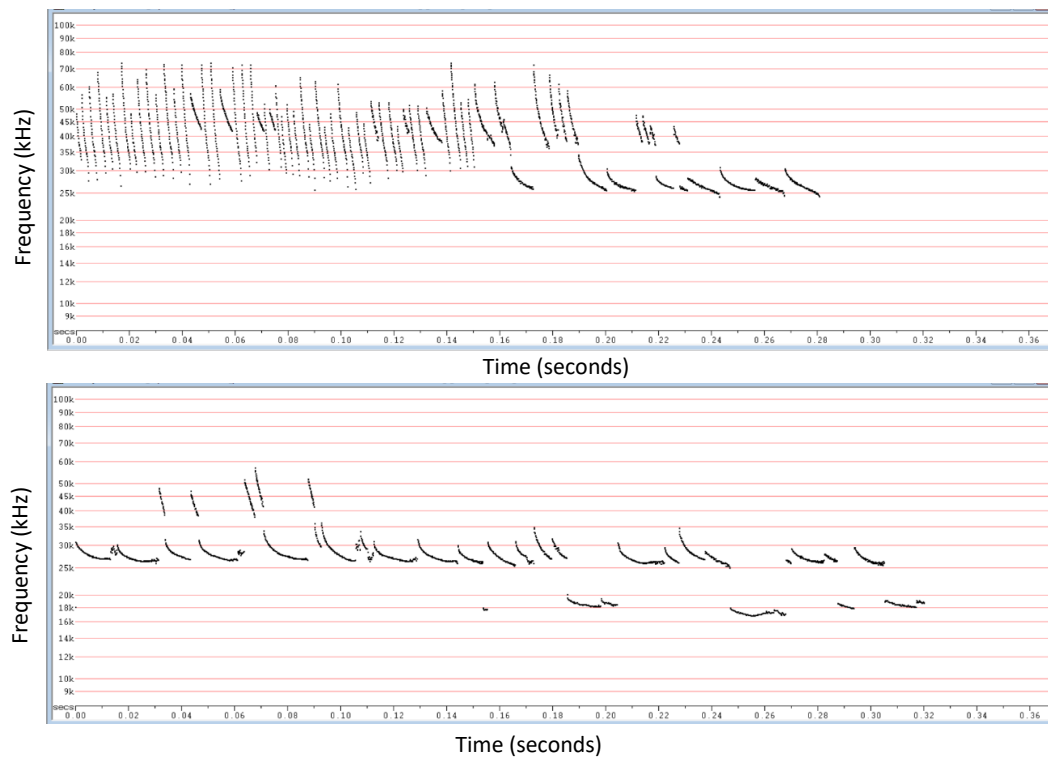


Fig. F.10. Examples of bat passes that contain multiple species. The sonograms are displayed in Titley Scientific's AnlookW bat analysis software (version 4.5s) in F7 in compressed time.

Appendix G: Kaleidoscope Noise Scrubber and Zero-Cross Analysis Limitations

G.1 Overview

To increase efficiency and consistency when I was manually analyzing the acoustic files, I used auto-identification software and analyzed the files in zero-cross file format rather than in full spectrum. I used the noise scrubber and species classifiers in Kaleidoscope Pro, as well as filters in AnalookW as two methods of auto-identification prior to manual analysis. AnalookW is only able to read zero-cross file formats, requiring all full spectrum files to be converted to zero-cross files prior to acoustic analysis. For full spectrum files to be zero-crossed, the signal to noise ratio needs to be above a certain threshold to be converted. Therefore, many of the low amplitude calls that appear in full-spectrum are absent in zero-cross files.

G.2 Methods

I wanted to test the proportion of bat passes that were missed by using the Kaleidoscope noise scrubber and analyzing the zero-cross files instead of the full spectrum files. These bat passes appeared as additional species in the full spectrum bat files (Fig. G.1) as well as in the files that were auto-identified as “noise” by Kaleidoscope Pro (Fig. G.2) and the files I manually identified as “noise” in AnalookW (Fig. G.3). Files that were identified as “noise” by Kaleidoscope Pro were automatically moved to a separate directory called “NOISE” during the initial conversion process. None of these files were analyzed further in AnalookW. Files that were identified as “noise” in AnalookW were files that were identified as a bat by Kaleidoscope Pro, but “NOTBAT” (e.g. squirrel, insect, ambient noise) when manually verified in AnalookW. Although, AnalookW only recognizes zero-cross files, Kaleidoscope Lite is able to load both zero-cross and full spectrum files. The “NOTBAT” files, when viewed as a full spectrum file in Kaleidoscope Lite were sometimes “NOTBAT”, but there were also files that did contain bat passes. This happened if only part of a full spectrum pass was successfully zero-crossed, leaving either ambiguity in possible bat-pass identification, or too few echolocation calls to be considered a bat pass (i.e. two or more calls were needed to identify a bat pass).

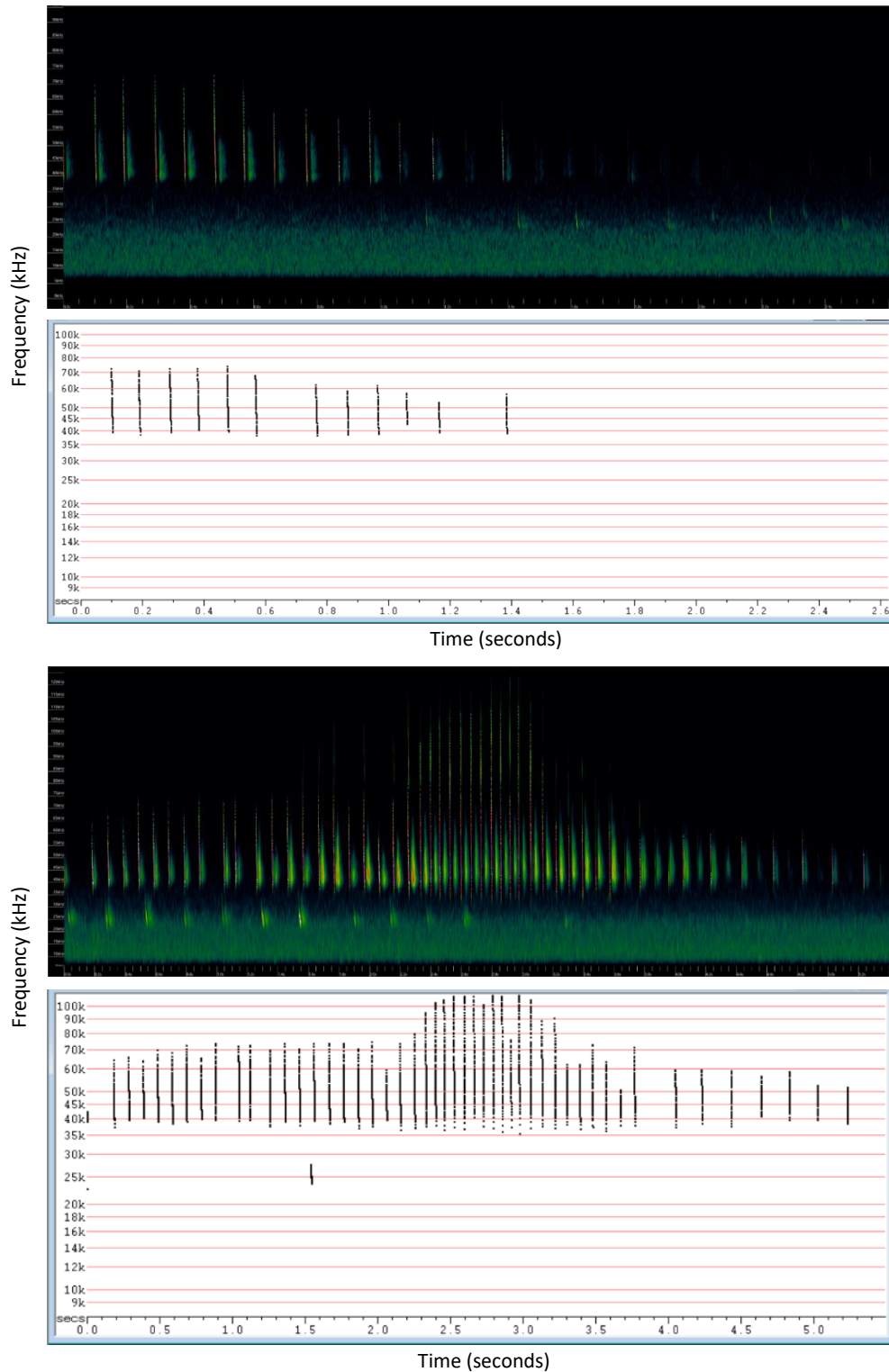


Fig. G.1. Examples of the same bat pass displayed in full spectrum (black background; Wildlife Acoustics' Kaleidoscope Lite bat analysis software, version 5.4.0) and zero-cross (white background; Titley Scientific's AnaloookW bat analysis software, version 4.5s). Both sonograms are displayed in true time with similar frequency and time axes. The full spectrum files display two species (40 kHz *Myotis* and low frequency bat), whereas only one species is present in the zero-cross file (40 kHz *Myotis* only).

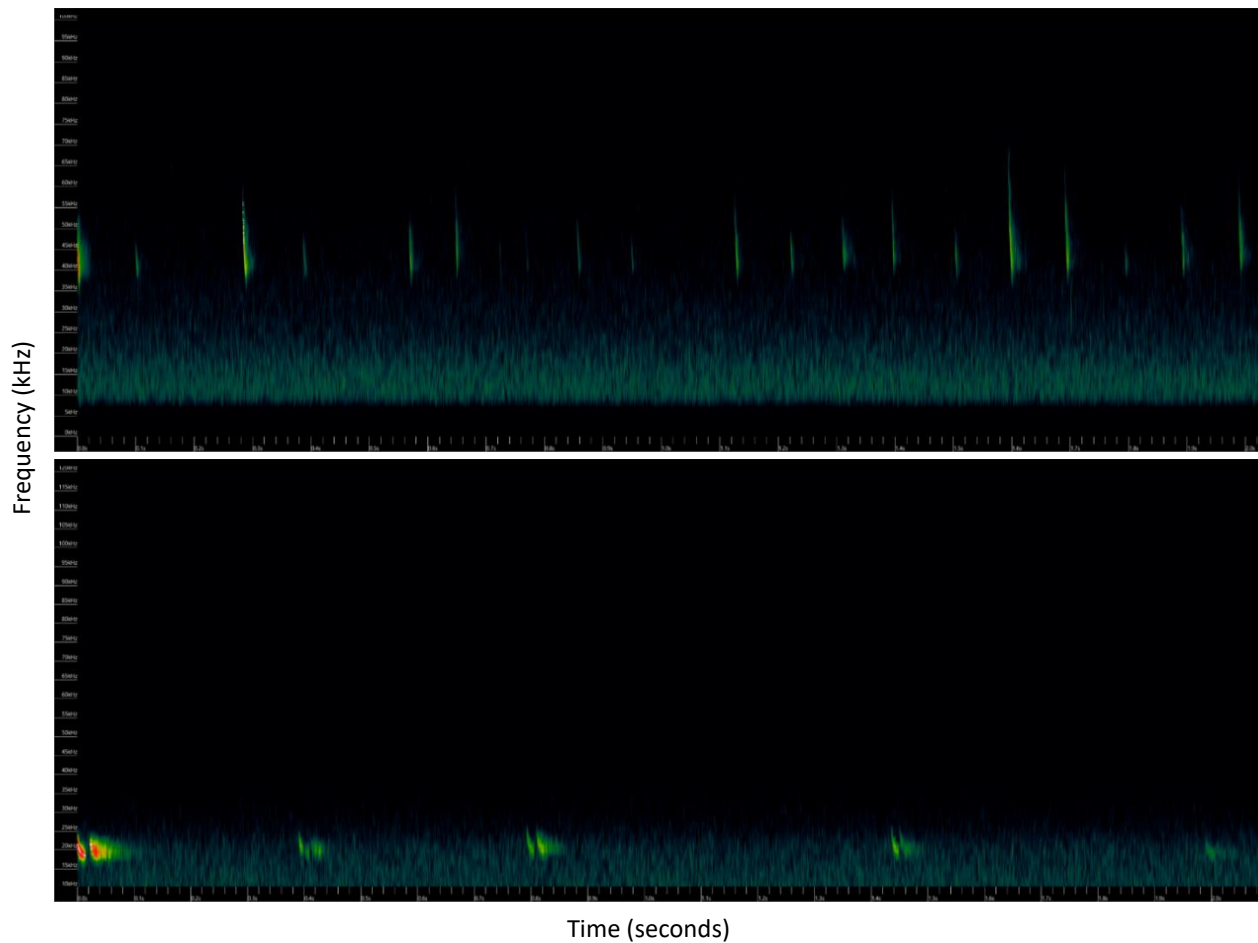


Fig. G.2. Examples of bat echolocation passes (Wildlife Acoustics’ Kaleidoscope Lite bat analysis software, version 5.4.0) that were identified as “noise” by Kaleidoscope Pro and were automatically moved to a separate directory called “NOISE” during the conversion process. The top sonogram contains a high frequency bat pass, and the bottom sonogram contains a hoary bat pass.

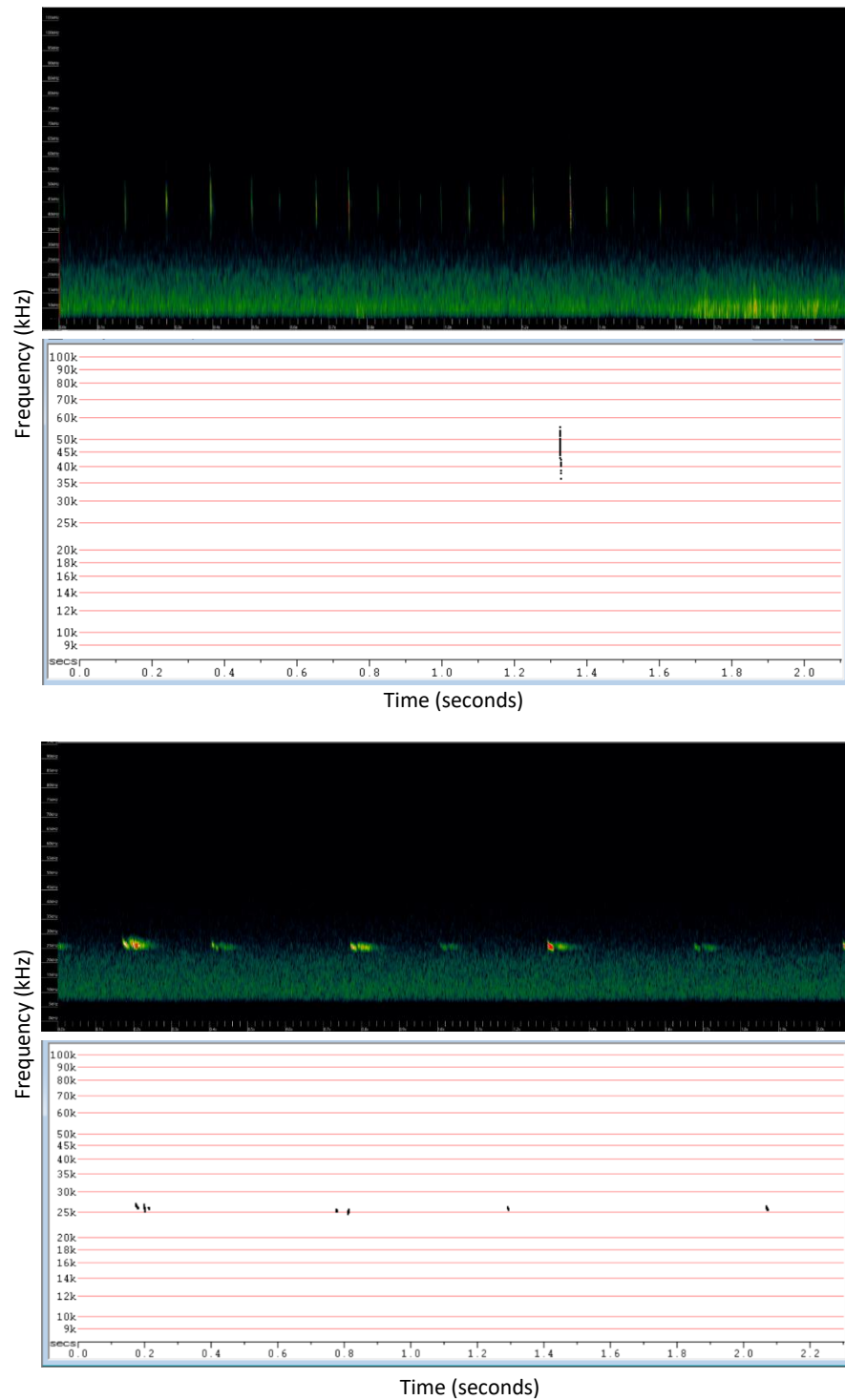


Fig. G.3. Examples of the same bat pass displayed in full spectrum (black background; Wildlife Acoustics' Kaleidoscope Lite bat analysis software, version 5.4.0) and zero-cross (white background; Titley Scientific's AnalookW bat analysis software, version 4.5s). Both sonograms are displayed in true time with similar frequency and time axes. The top comparison illustrates a full spectrum file labeled as a high frequency bat and the corresponding zero-cross file labeled as "NOTBAT" in AnalookW. The bottom comparison illustrates a full spectrum file labeled as a big brown/silver-haired bat and the corresponding zero-cross file labeled as "NOTBAT" in AnalookW.

To determine the number of full spectrum bat files that contained additional species, as well as the number of “noise” files (from both Kaleidoscope Pro and AnalookW) that contained bats, I viewed a subsample of full spectrum files in Kaleidoscope Lite. Due to the large quantity of files, I randomly selected a subset of files in RStudio (version 1.3.1093, R Core Team 2020; package *dplyr*, Wickham et al. 2021) for all dates for every site for each year to determine the proportion of files containing additional species not zero-crossed. For each analysis (i.e. the number of full spectrum bat files containing additional species, the number of “noise” files from Kaleidoscope Pro containing bats, and the number of “noise” files from AnalookW containing bats), ten individual files were selected for each date for every site for each year. Some dates had less than ten files, therefore I selected however many files were available for that date (i.e. one to nine).

To determine how many of the subsampled full spectrum files contained additional bat species that were not detected in the zero-cross files, I manually identified if ‘one’ or ‘more than one’ bat species/species group was present in each file. I then determined the proportion of missed additional species (i.e. file with additional species only identified in the full spectrum file/total number of files subsampled) for each date for every site for each year.

To determine the number of “noise” files from Kaleidoscope Pro that contained bats, I manually identified whether a bat species/species group was present in each file. Due to the limited diagnostic qualities that many of these low amplitude calls have, I identified the files with bat passes as either a high frequency bat, big brown/silver-haired bat, hoary bat, or non-diagnostic low frequency bat. If more than one species/species group was present in a file, both were identified and included. I then determined the proportion of misidentified noise files (i.e. file with a bat pass that was misidentified as noise/total number of files subsampled) for each date for every site for each year. I followed the same analysis steps to determine the number of “noise” files from AnalookW that contained bats when viewed in full spectrum.

G.3 Results

G.3.1 Missed additional species

In total, there were 65,180 full spectrum bat files, detected across 374 of the 380 detector nights. Due to file conversion issues, a small number of files (10) were missed when

converting files from Kaleidoscope Pro to AnalookW to Kaleidoscope Lite. Bats were not detected every night, therefore nights with no detections were excluded. By subsampling, I manually analyzed 3,626 full spectrum files to determine how many contained additional bat species that were not detected in the zero-cross files. For each detector night, between 0 – 60% of full spectrum files contained additional species that were not detected in the zero-cross files. The mean percentage across the six years was 3.5% (Table G.1 and G.2). The two sites with the highest bat activity, Lost Lake and Lone Lake, had much higher percentages than any of the other sites (12.6% and 7.9%, respectively). All of the remaining sites had percentages of 5.0% or less. The mean percentage across the six years, excluding Lost Lake and Lone Lake, was 2.0%.

G.3.2 Missed species from Kaleidoscope Pro noise files

In total, there were 64,979 noise files identified by Kaleidoscope Pro, detected across 380 detector nights. By subsampling, I analyzed 3,800 full spectrum noise files to determine the proportion that contained bats. For each detector night, between 0 – 70% of the full spectrum noise files contained bat passes. The mean percentage of noise files containing a bat pass (i.e. high frequency bat, big brown/silver-haired bat, hoary bat, or non-diagnostic low frequency bat) across the six years was 9.3% (Table G.3 and G.4). High frequency bats were detected in 7.2% of files, big brown/silver-haired bats were detected in 1.3% of files, hoary bats were detected in 0.8% of files, and non-diagnostic low frequency bat were detected in 0.3% of files. A number of the “noise” files (13) contained more than one species, resulting in a slightly higher total (9.6%) when species/species groups specific totals were summed. The two sites with highest proportion of missed passes from the noise files, Lost Lake and Lone Lake, had much higher percentages than any of the other sites (34.2% and 22.9%, respectively). All of the remaining sites had percentages of 11.2% or less. The mean percentage across the six years, excluding Lost Lake and Lone Lake, was 5.1%.

G.3.3 Missed species from AnalookW noise files

In total, there were 10,035 noise files manually identified in AnalookW, detected across 328 of the 380 detector nights. AnalookW noise files were not detected every night, therefore nights with no detections were excluded. By subsampling, I analyzed 2,055 full spectrum noise files to determine the proportion that contained bats. For each detector night, between 0 –

100% of the full spectrum noise files contained bat passes. The mean percentage of noise files containing a bat pass (i.e. high frequency bat, big brown/silver-haired bat, hoary bat, or non-diagnostic low frequency bat) across the six years was 21.3% (Table G.5 and G.6). High frequency bats were detected in 8.1% of files, big brown/silver-haired bat were detected in 5.1% of files, hoary bats were detected in 8.5% of files, and non-diagnostic low frequency bats were detected in 2.6% of files. A number of the “noise” files (41) contained more than one species, resulting in a slightly higher total (24.3%) when species/species groups specific totals were summed. The two sites with the highest bat activity, Lost Lake and Lone Lake, had much higher percentages than any of the other sites (65.3% and 53.3%, respectively). All of the remaining sites had percentages of 35.7% or less. The mean percentage across the six years, excluding Lost Lake and Lone Lake, was 12.5%.

Table G.1. The percent of subsampled (i.e. ten individual files selected for each date for every site for each year) full spectrum bat files for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020 that contained additional species that were missed in the zero-cross files. Note that the values represent weighted means (i.e. sites with larger sample sizes contribute more to the mean than sites with smaller sample sizes). For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	2.2%	2.4%	0.5%	1.8%	0.9%	2.6%	2.4%	1.9%	1.8%
Bison Paddock	0.0%	5.0%	2.0%	2.5%	0.0%	4.0%	5.0%	2.9%	2.7%
Blakiston Roadside	8.0%	3.3%	0.0%	4.0%	0.0%	0.0%	2.5%	0.7%	2.4%
Cameron Lake	0.0%	4.0%	0.0%	1.5%	0.0%	0.0%	2.0%	0.7%	1.1%
Dipper at Rowe	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	-	0.0%	0.0%
Red Rock	0.0%	0.0%	0.0%	0.0%	1.7%	5.0%	0.0%	2.5%	1.3%
Sewage Lagoon	3.3%	2.9%	0.0%	2.1%	3.3%	5.0%	2.5%	3.6%	2.9%
Unburn	5.8%	5.5%	6.6%	5.9%	3.2%	3.6%	4.5%	3.7%	4.7%
Belly Bend	7.5%	5.0%	5.0%	5.8%	5.0%	1.7%	-	3.3%	4.6%
Boundary Bay	-	5.0%	12.5%	8.8%	7.1%	2.5%	0.0%	3.3%	5.0%
Lone Lake	13.3%	10.0%	8.6%	10.0%	2.9%	8.3%	6.7%	5.8%	7.9%
Lost Lake	12.5%	15.7%	10.0%	12.8%	-	10.0%	14.3%	12.3%	12.6%
Sofa Burn	0.0%	0.0%	0.0%	0.0%	0.0%	3.3%	0.0%	1.3%	0.8%
Wishbone	2.0%	1.7%	0.0%	1.5%	1.7%	0.0%	0.0%	0.6%	1.1%
Yarrow Cabin	2.5%	0.0%	0.0%	0.7%	1.7%	0.0%	2.5%	1.1%	0.9%
Total (burn and unburn)	4.0%	4.3%	3.9%	4.1%	2.1%	3.2%	3.7%	2.9%	3.5%

Table G.2. The number of subsampled full spectrum bat files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the additional species that were missed in the zero-cross files.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	217	329	206	752	313	241	194	748	1500
Bison Paddock	30	40	50	120	41	50	40	131	251
Blakiston Roadside	50	59	26	135	59	14	31	104	239
Cameron Lake	50	50	30	130	50	50	43	143	273
Dipper at Rowe	30	40	20	90	44	27	-	71	161
Red Rock	27	70	40	137	59	60	40	159	296
Sewage Lagoon	30	70	40	140	60	40	40	140	280
Unburn	238	486	290	1014	373	416	323	1112	2126
Belly Bend	40	60	20	120	60	60	-	120	240
Boundary Bay	-	40	40	80	70	40	63	173	253
Lone Lake	30	90	70	190	70	60	60	190	380
Lost Lake	40	70	70	180	-	60	70	130	310
Sofa Burn	40	38	20	98	60	60	40	160	258
Wishbone	50	120	30	200	60	60	50	170	370
Yarrow Cabin	38	68	40	146	53	76	40	169	315
Total (burn and unburn)	455	815	496	1766	686	657	517	1860	3626

Table G.3. The percent of subsampled (i.e. ten individual files selected for each date for every site for each year) Kaleidoscope Pro noise files for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020 that contained bats. Note that the values represent weighted means (i.e. sites with larger sample sizes contribute more to the mean than sites with smaller sample sizes). For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	5.7%	5.7%	5.5%	5.6%	6.4%	2.4%	6.4%	5.0%	5.3%
Bison Paddock	3.3%	2.5%	2.0%	2.5%	0.0%	0.0%	17.5%	5.0%	3.8%
Blakiston Roadside	20.0%	5.7%	7.5%	10.6%	8.3%	0.0%	0.0%	3.1%	6.9%
Cameron Lake	2.0%	8.0%	0.0%	3.8%	22.0%	10.0%	8.3%	13.1%	9.0%
Dipper at Rowe	0.0%	10.0%	10.0%	6.7%	0.0%	0.0%	-	0.0%	3.5%
Red Rock	2.5%	6.3%	5.0%	5.0%	0.0%	1.7%	2.5%	1.3%	3.1%
Sewage Lagoon	0.0%	2.9%	10.0%	4.3%	8.3%	2.5%	2.5%	5.0%	4.6%
Unburn	12.5%	12.2%	13.4%	12.6%	8.2%	13.3%	14.7%	12.0%	12.3%
Belly Bend	5.0%	6.7%	10.0%	6.7%	6.7%	1.7%	-	4.2%	5.4%
Boundary Bay	-	17.5%	27.5%	22.5%	5.7%	17.5%	0.0%	6.1%	11.2%
Lone Lake	43.3%	20.0%	18.6%	23.2%	27.1%	28.3%	11.7%	22.6%	22.9%
Lost Lake	25.0%	31.4%	14.3%	23.3%	-	43.3%	54.3%	49.2%	34.2%
Sofa Burn	2.5%	7.5%	10.0%	6.0%	3.3%	0.0%	5.0%	2.5%	3.8%
Wishbone	2.0%	3.3%	0.0%	2.5%	0.0%	5.0%	4.0%	2.9%	2.7%
Yarrow Cabin	7.5%	2.9%	2.5%	4.0%	3.3%	2.5%	2.0%	2.6%	3.2%
Total (burn and unburn)	9.1%	9.5%	10.0%	9.6%	7.3%	8.9%	11.4%	9.0%	9.3%

Table G.4. The number of subsampled Kaleidoscope Pro noise files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the percent that contained bats.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	230	350	220	800	330	290	220	840	1640
Bison Paddock	30	40	50	120	50	50	40	140	260
Blakiston Roadside	50	70	40	160	60	60	40	160	320
Cameron Lake	50	50	30	130	50	50	60	160	290
Dipper at Rowe	30	40	20	90	50	30	-	80	170
Red Rock	40	80	40	160	60	60	40	160	320
Sewage Lagoon	30	70	40	140	60	40	40	140	280
Unburn	240	490	290	1020	380	420	340	1140	2160
Belly Bend	40	60	20	120	60	60	-	120	240
Boundary Bay	-	40	40	80	70	40	70	180	260
Lone Lake	30	90	70	190	70	60	60	190	380
Lost Lake	40	70	70	180	-	60	70	130	310
Sofa Burn	40	40	20	100	60	60	40	160	260
Wishbone	50	120	30	200	60	60	50	170	370
Yarrow Cabin	40	70	40	150	60	80	50	190	340
Total (burn and unburn)	470	840	510	1820	710	710	560	1980	3800

Table G.5. The percent of subsampled (i.e. ten individual files selected for each date for every site for each year) AnalookW noise files for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, that contained bats. Note that the values represent weighted means (i.e. sites with larger sample sizes contribute more to the mean than sites with smaller sample sizes). For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	13.1%	7.7%	12.1%	10.6%	8.0%	0.0%	10.5%	6.1%	8.2%
Bison Paddock	0.0%	3.6%	0.0%	1.2%	0.0%	0.0%	0.0%	0.0%	0.5%
Blakiston Roadside	40.0%	0.0%	2.5%	13.1%	1.7%	0.0%	2.5%	1.3%	7.4%
Cameron Lake	8.0%	0.0%	3.3%	3.8%	12.7%	0.0%	10.0%	7.7%	6.0%
Dipper at Rowe	0.0%	0.0%	0.0%	0.0%	2.0%	-	-	2.0%	0.8%
Red Rock	5.0%	10.0%	50.0%	23.6%	27.5%	0.0%	33.3%	16.2%	19.6%
Sewage Lagoon	8.3%	37.5%	11.1%	20.8%	9.3%	0.0%	12.5%	7.4%	13.3%
Unburn	17.1%	40.7%	37.8%	34.1%	28.1%	28.1%	32.2%	29.4%	31.7%
Belly Bend	5.0%	42.1%	48.3%	30.8%	42.7%	1.7%	-	22.2%	26.5%
Boundary Bay	-	100.0%	0.0%	66.7%	2.2%	0.0%	8.3%	5.1%	17.4%
Lone Lake	33.3%	64.6%	45.8%	52.5%	56.4%	40.0%	66.0%	54.3%	53.3%
Lost Lake	65.6%	53.2%	53.8%	55.6%	-	75.0%	81.8%	79.0%	65.3%
Sofa Burn	2.5%	12.5%	0.0%	6.0%	3.3%	31.3%	0.0%	11.2%	8.9%
Wishbone	0.0%	17.5%	3.3%	11.0%	0.0%	5.4%	2.9%	2.8%	7.2%
Yarrow Cabin	16.7%	37.5%	50.0%	35.0%	75.0%	33.3%	0.0%	36.4%	35.7%
Total (burn and unburn)	15.2%	27.7%	25.8%	23.8%	18.5%	15.3%	23.3%	18.9%	21.3%

Table G.6. The number of subsampled AnalookW noise files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the percent that contained bats.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	109	164	129	402	243	214	166	623	1025
Bison Paddock	25	25	43	93	50	50	40	140	233
Blakiston Roadside	13	43	32	88	60	50	40	150	238
Cameron Lake	43	50	30	123	49	50	53	152	275
Dipper at Rowe	6	16	7	29	35	-	-	35	64
Red Rock	13	15	6	34	13	60	21	94	128
Sewage Lagoon	9	15	11	35	36	4	12	52	87
Unburn	142	228	147	517	187	163	163	513	1030
Belly Bend	33	42	16	91	34	57	-	91	182
Boundary Bay	-	4	1	5	30	10	42	82	87
Lone Lake	6	44	46	96	30	13	21	64	160
Lost Lake	10	28	30	68	-	11	42	53	121
Sofa Burn	36	19	20	75	30	11	4	45	120
Wishbone	50	84	29	163	57	58	32	147	310
Yarrow Cabin	7	7	5	19	6	3	22	31	50
Total (burn and unburn)	251	392	276	919	430	377	329	1136	2055

G.4 Discussion

G.4.1 Missed additional species

In the full-spectrum version, there were acoustic files that contained two or more bat species, while the corresponding zero-cross version only showed one species. By examining the percent of subsampled full spectrum bat files that contained additional species that were missed in the zero-cross files, I determined that this was likely a relatively minor source of error. A mean of 3.5% of the subsampled files were missing additional species. However, for the sites with high activity such as Lost Lake and Lone Lake, a much larger percentage of files contained missing additional species; 12.6% and 7.9% respectively.

G.4.2 Missed species from Kaleidoscope Pro noise files

Although the Kaleidoscope noise scrubber has been used by a number of acoustic studies (e.g. Austin et al. 2018a, Smith 2019, Allagas 2020, Tena et al. 2020, Li et al. 2021), it does misidentify bat passes as noise (Braun de Torrez et al. 2018b). For this study, the mean percentage of noise files containing a bat pass was 9.3%, with 77.4% of the passes being high frequency bats, 14.0% being big brown/silver-haired bat, and 8.6% being hoary bats. If the noise scrubber was equally biased across all frequency groups, then I would expect proportions of missed species to be similar to proportions of actual detections. High frequency bats made up 73.1% of all detections, big brown/silver-haired bat made up 15.4%, and hoary bats made up 4.2%. Although high frequency bats and big brown/silver-haired bat had similar proportions between missed species and overall detections, hoary bats were over twice as common in the noise files compared to overall detections. There was a significant difference in the expected versus observed frequency of missed hoary bat detections compared to all other species detections (Chi-square goodness-of-fit, $\chi^2 = 18.97$, $df = 1$, $P < 0.001$). This suggests that the noise scrubber is biasing results and underrepresenting hoary bats. Sites were also not equally represented in terms of the proportion of noise files that contained bats. Sites with high levels of activity, such as Lost Lake and Lone Lake, had over twice the proportion of noise files that contained bats compared to any other site.

G.4.3 Missed species from AnalookW noise files

In addition to the files that Kaleidoscope identified as noise, there were files that were considered bat passes by Kaleidoscope but appeared to be noise when I manually identified them as zero-cross files in AnalookW. When I re-examined a subsample of the AnalookW noise files in full spectrum, a mean of 21.3% contained bat passes. There was also a strong bias for which species were most commonly misidentified as noise in AnalookW. The issue was most apparent for hoary bats, where they were found to be misidentified 9.5 times more often than would be expected, relative to how often they were detected in the study area (i.e. hoary bats made up 4.2% of overall activity in the study area but 39.9% of all misidentified AnalookW noise files). There was a significant difference in the expected versus observed frequency of missed hoary bat detections compared to all other species detections (Chi-square goodness-of-fit, $\chi^2 = 653.10$, $df = 1$, $P < 0.001$). Big brown/silver-haired bats were moderately overrepresented appearing 1.6 times more often than would be expected (i.e. big brown/silver-haired bat made up 15.4% of overall activity in the study area but 23.9% of all misidentified AnalookW noise files). High frequency bats were underrepresented, appearing half as often as would be expected (i.e. high frequency bats made up 73.1% of overall activity in the study area but 38.0% of all misidentified AnalookW noise files). This suggests that converting files from full spectrum to zero-cross biases results, primarily for the low frequency bats. Sites were also not equally represented in terms of the proportion of AnalookW noise files that contained bats. Sites with high levels of activity, such as Lost Lake and Lone Lake, had ≥ 1.5 times as many AnalookW noise files that contained bats compared to any other site.

G.5 Conclusion

As the same software settings and workflow were used for each year of the acoustic data, I do not believe that using the Kaleidoscope noise scrubber or analyzing the zero-cross files instead of the full spectrum files would have led to biased results when comparing bat activity levels pre- versus post-wildfire. I believe that hoary bat activity levels were greatly underestimated, and Lost Lake and Lone Lake had higher activity levels than was reported, but it was consistently biased amongst all years of the study. It should also be noted that depending on the sample size, manually reviewing all of the noise files would significantly increase the

amount of time required for the acoustic analysis. In the six-year study, 75,225 files were identified as bats by Kaleidoscope Pro and 64,979 files were identified as noise.

Although studies have examined the limitations of the auto-identification function in bat acoustic analysis (e.g. Lemen et al. 2015, Russo and Voight 2016, Rydell et al. 2017, Nocera et al. 2019), to my knowledge no one has looked at error rates of bat passes missed in zero-cross conversions, or rates of bat passes incorrectly identified as noise. The workflow I chose to follow (i.e. using the auto-identifications provided by Kaleidoscope and completing the acoustic analysis in AnalookW) is a method suggested by the North American Bat Monitoring Program (NABat) (Reichert et al. 2018). NABat is a collaborative monitoring program that uses standardized protocols to collect acoustic bat data (Loeb et al. 2015). The primary argument offered for ignoring missed or incorrectly identified bat passes is that these passes are of low quality and difficult to reliably identify (Frick 2013, Reichert et al. 2018). Noise scrubbing also reduces storage requirements and increases acoustic analysis efficiency (Reichert et al. 2018).

Despite some of the missed/incorrectly identified bat passes in this study being of low quality, there were also passes easily identifiable to species (i.e. hoary bat) and species groups (i.e. big brown/silver-haired bat). Some researchers recognize the issue with Kaleidoscope incorrectly identifying bat passes as noise (e.g. Braun de Torrez et al. 2018b, Perea and Tena 2020), but most acoustic studies using Kaleidoscope exclude the noise files without any manual review (e.g. Austin et al. 2018a, Smith 2019, Tena et al. 2020, Allagas 2020, Li et al. 2021). Accounting for the bias in missed species could be particularly important for studies on endangered species, when conducting presence/absence analyses, or estimating abundance to inform mitigation decisions.

Although the proportions of bat passes missed in the zero-cross conversions or proportions of bat passes incorrectly identified as noise were relatively high in this study, it is probable that it was influenced by the environmental conditions of Waterton Lakes National Park (WLNP) and the deployment locations. WLNP is situated in a very windy area (Parks Canada 2018), with most nights likely experiencing a large amount of background noise (i.e. wind). As such, bats likely had to be near to and echolocating towards the detector's microphone to meet the signal to noise ratio threshold required by Kaleidoscope to convert a

pass from full spectrum to zero-cross. In addition, most detector sites were located near to waterbodies, areas with high levels of bat activity (Salvarina 2016). If detectors had been deployed in flight corridors (e.g. forest trails that lead to a water source) rather than feeding/drinking congregation areas like a waterbody, less bats would have been detected but the signals would have likely been stronger and more direct.

Appendix H: Species/Species Groups' Mean Passes/Night

Table H.1. The mean passes/night (\pm SE) for 40 kHz *Myotis* for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta. Results are displayed for each year for the burned and unburned sites. For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015		2016		2017		2018		2019		2020	
	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n
Burn	23.1 \pm 5.9	23	35.2 \pm 8.9	35	46.9 \pm 11.6	22	76.9 \pm 20.5	33	33.2 \pm 8.0	29	108.7 \pm 58.1	22
Bison Paddock	72.7 \pm 19.4	3	73.5 \pm 40.3	4	118.2 \pm 26.7	5	31.0 \pm 12.6	5	15.6 \pm 4.8	5	445.2 \pm 278.5	4
Blakiston Roadside	8.2 \pm 2.5	5	3.1 \pm 1.4	7	2.2 \pm 1.3	4	32.7 \pm 12.1	6	0.0 \pm 0.0	6	3.0 \pm 2.0	4
Cameron Lake	14.4 \pm 5.0	5	99.0 \pm 36.5	5	10.3 \pm 3.8	3	153.4 \pm 84.9	5	80.8 \pm 23.4	5	68.7 \pm 43.0	6
Dipper at Rowe	8.7 \pm 2.4	3	28.5 \pm 8.3	4	29.5 \pm 3.5	2	6.8 \pm 2.2	5	10.0 \pm 1.7	3	-	0
Red Rock	2.8 \pm 1.2	4	3.6 \pm 1.3	8	12.0 \pm 2.1	4	4.0 \pm 1.1	6	15.2 \pm 5.6	6	3.5 \pm 1.4	4
Sewage Lagoon	54.3 \pm 7.8	3	39.6 \pm 12.4	7	73.5 \pm 11.7	4	227.0 \pm 40.1	6	89.8 \pm 18.4	4	43.2 \pm 17.8	4
Unburn	194.9 \pm 86.6	24	142.0 \pm 38.0	49	64.7 \pm 8.9	29	60.8 \pm 14.9	38	140.0 \pm 42.2	42	172.0 \pm 51.7	34
Belly Bend	23.0 \pm 11.7	4	25.2 \pm 6.6	6	8.5 \pm 2.5	2	30.7 \pm 7.5	6	52.0 \pm 13.7	6	-	0
Boundary Bay	-	0	170.0 \pm 70.6	4	79.0 \pm 7.7	4	38.0 \pm 6.5	7	50.5 \pm 15.0	4	6.1 \pm 1.7	7
Lone Lake	358.7 \pm 170.4	3	137.2 \pm 34.4	9	77.4 \pm 16.0	7	144.3 \pm 29.0	7	91.2 \pm 24.9	6	200.2 \pm 24.3	6
Lost Lake	720.2 \pm 439.8	4	594.3 \pm 184.9	7	88.7 \pm 13.6	7	-	0	684.5 \pm 174.2	6	638.0 \pm 145.6	7
Sofa Burn	121.0 \pm 49.7	4	37.0 \pm 21.3	4	105.0 \pm 92.0	2	118.2 \pm 74.2	6	53.7 \pm 30.4	6	15.5 \pm 4.5	4
Wishbone	15.4 \pm 6.5	5	35.9 \pm 7.3	12	9.3 \pm 0.9	3	8.3 \pm 3.4	6	19.0 \pm 5.5	6	10.8 \pm 1.5	5
Yarrow Cabin	23.0 \pm 11.7	4	21.7 \pm 7.5	7	35.8 \pm 7.6	4	15.2 \pm 5.7	6	34.6 \pm 10.4	8	4.2 \pm 1.2	5
Total (burn and unburn)	110.8 \pm 45.6	47	97.5 \pm 23.1	84	57.0 \pm 7.2	51	68.3 \pm 12.4	71	96.4 \pm 25.8	71	147.1 \pm 38.7	56

Table H.2. The mean passes/night (\pm SE) for big brown/silver-haired bats for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta. Results are displayed for each year for the burned and unburned sites. For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015		2016		2017		2018		2019		2020	
	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n
Burn	22.9 \pm 12.2	23	16.5 \pm 6.1	35	16.0 \pm 4.7	22	14.6 \pm 3.9	33	6.8 \pm 1.3	29	16.0 \pm 7.0	22
Bison Paddock	16.0 \pm 4.5	3	22.0 \pm 8.4	4	17.2 \pm 3.4	5	2.6 \pm 1.9	5	8.2 \pm 2.8	5	50.2 \pm 35.4	4
Blakiston Roadside	85.8 \pm 49.4	5	30.1 \pm 26.4	7	3.8 \pm 2.8	4	7.0 \pm 3.7	6	0.0 \pm 0.0	6	10.0 \pm 8.7	4
Cameron Lake	3.6 \pm 1.2	5	32.8 \pm 20.7	5	4.0 \pm 3.1	3	45.2 \pm 20.0	5	9.2 \pm 2.9	5	3.2 \pm 3.0	6
Dipper at Rowe	1.3 \pm 0.3	3	1.2 \pm 0.2	4	2.5 \pm 1.5	2	0.0 \pm 0.0	5	0.3 \pm 0.3	3	-	0
Red Rock	4.0 \pm 3.4	4	10.5 \pm 4.2	8	53.2 \pm 13.4	4	16.8 \pm 3.9	6	12.2 \pm 2.5	6	20.0 \pm 2.3	4
Sewage Lagoon	4.0 \pm 1.5	3	3.6 \pm 0.9	7	5.0 \pm 3.4	4	16.7 \pm 4.3	6	9.0 \pm 3.7	4	2.8 \pm 1.2	4
Unburn	50.0 \pm 15.7	24	36.8 \pm 9.1	49	53.2 \pm 10.0	29	14.4 \pm 3.2	38	39.0 \pm 10.5	42	52.8 \pm 15.0	34
Belly Bend	63.5 \pm 40.0	4	5.8 \pm 1.9	6	24.5 \pm 8.5	2	9.7 \pm 2.5	6	10.2 \pm 4.1	6	-	0
Boundary Bay	-	0	9.2 \pm 3.6	4	49.0 \pm 23.9	4	14.4 \pm 5.0	7	2.0 \pm 0.9	4	1.7 \pm 0.7	7
Lone Lake	28.3 \pm 20.6	3	94.6 \pm 35.2	9	88.0 \pm 23.7	7	42.7 \pm 11.8	7	58.2 \pm 23	6	61.0 \pm 19.8	6
Lost Lake	170.8 \pm 53.3	4	78.4 \pm 29.5	7	86.0 \pm 18.9	7	-	0	178.7 \pm 27.9	6	195.9 \pm 32.2	7
Sofa Burn	27.0 \pm 8.4	4	3.0 \pm 1.7	4	14.0 \pm 9.0	2	3.2 \pm 1.0	6	5.8 \pm 1.8	6	0.0 \pm 0.0	4
Wishbone	8.6 \pm 2.4	5	25.0 \pm 3.9	12	3.7 \pm 0.9	3	9.5 \pm 1.9	6	9.0 \pm 2.3	6	5.8 \pm 1.2	5
Yarrow Cabin	7.0 \pm 3.0	4	2.9 \pm 0.4	7	10.2 \pm 4.1	4	2.2 \pm 0.9	6	7.5 \pm 2.4	8	3.6 \pm 1.3	5
Total (burn and unburn)	36.8 \pm 10.1	47	28.3 \pm 5.9	84	37.1 \pm 6.5	51	14.5 \pm 2.5	71	25.9 \pm 6.5	71	38.3 \pm 9.7	56

Table H.3. The mean passes/night (\pm SE) for hoary bats for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta. Results are displayed for each year for the burned and unburned sites. For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015		2016		2017		2018		2019		2020	
	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n
Burn	7.9 \pm 2	23	4.2 \pm 1.2	35	3.2 \pm 0.9	22	0.3 \pm 0.1	33	0.4 \pm 0.2	29	1.5 \pm 0.5	22
Bison Paddock	10.0 \pm 1.0	3	7.5 \pm 3.2	4	4.6 \pm 1.3	5	0.0 \pm 0.0	5	1.4 \pm 0.7	5	4.5 \pm 2.2	4
Blakiston Roadside	23.6 \pm 3.5	5	10.0 \pm 5.1	7	0.0 \pm 0.0	4	0.0 \pm 0.0	6	0.0 \pm 0.0	6	1.0 \pm 0.7	4
Cameron Lake	2.0 \pm 0.8	5	2.2 \pm 0.2	5	0.0 \pm 0.0	3	0.0 \pm 0.0	5	0.8 \pm 0.4	5	0.3 \pm 0.2	6
Dipper at Rowe	0.7 \pm 0.3	3	0.5 \pm 0.3	4	0.0 \pm 0.0	2	0.0 \pm 0.0	5	0.0 \pm 0.0	3	-	0
Red Rock	1.0 \pm 0.7	4	2.2 \pm 0.6	8	9.0 \pm 2.1	4	0.8 \pm 0.3	6	0.0 \pm 0.0	6	1.8 \pm 0.5	4
Sewage Lagoon	5.7 \pm 0.9	3	2.1 \pm 1.0	7	2.8 \pm 1.2	4	0.7 \pm 0.2	6	0.5 \pm 0.3	4	0.8 \pm 0.5	4
Unburn	15.0 \pm 4.8	24	18.8 \pm 5.4	49	16.8 \pm 4.3	29	8.1 \pm 1.8	38	3.1 \pm 0.5	42	9.7 \pm 3.4	34
Belly Bend	55.8 \pm 17.9	4	24.2 \pm 3.6	6	73.0 \pm 12.0	2	27.8 \pm 6.6	6	6.2 \pm 1.9	6	-	0
Boundary Bay	-	0	1.5 \pm 0.6	4	13.8 \pm 6.4	4	5.9 \pm 2.3	7	0.5 \pm 0.5	4	5.0 \pm 2.7	7
Lone Lake	0.3 \pm 0.3	3	50.9 \pm 26.4	9	26.7 \pm 10.5	7	0.6 \pm 0.3	7	1.7 \pm 0.9	6	10.2 \pm 3.0	6
Lost Lake	1.8 \pm 1.1	4	12.7 \pm 8.4	7	8.9 \pm 4.2	7	-	0	1.7 \pm 0.8	6	29.4 \pm 14.5	7
Sofa Burn	13.5 \pm 4.6	4	5.2 \pm 1.2	4	10.5 \pm 4.5	2	4.5 \pm 0.6	6	5.5 \pm 0.7	6	0.8 \pm 0.2	4
Wishbone	9.8 \pm 2.6	5	13.8 \pm 2.0	12	2.0 \pm 0.6	3	5.0 \pm 0.9	6	3.7 \pm 1.3	6	1.8 \pm 0.6	5
Yarrow Cabin	6.2 \pm 1.1	4	5.1 \pm 1.1	7	2.5 \pm 0.6	4	6.2 \pm 1.6	6	2.1 \pm 1.1	8	3.0 \pm 1.2	5
Total (burn and unburn)	11.5 \pm 2.7	47	12.7 \pm 3.2	84	10.9 \pm 2.6	51	4.4 \pm 1.1	71	2.0 \pm 0.3	71	6.5 \pm 2.1	56

Table H.4. The mean passes/night (\pm SE) for long-eared *Myotis* for each acoustic detector site from 2015 to 2020 in Waterton Lakes National Park, Alberta. Results are displayed for each year for the burned and unburned sites. For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015		2016		2017		2018		2019		2020	
	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n	mean passes/night	n
Burn	4.0 \pm 1.5	23	8.4 \pm 3.3	35	1.9 \pm 0.3	22	6.6 \pm 2.5	33	1.2 \pm 0.3	29	1.7 \pm 0.6	22
Bison Paddock	0.7 \pm 0.3	3	2.0 \pm 0.6	4	2.4 \pm 0.7	5	0.0 \pm 0.0	5	0.2 \pm 0.2	5	1.0 \pm 1.0	4
Blakiston Roadside	4.4 \pm 0.7	5	3.7 \pm 1.1	7	1.2 \pm 0.2	4	26.3 \pm 10.7	6	0.8 \pm 0.7	6	3.5 \pm 2.9	4
Cameron Lake	2.0 \pm 0.5	5	8.0 \pm 4.8	5	3.3 \pm 0.7	3	9.0 \pm 3.0	5	1.2 \pm 0.7	5	0.3 \pm 0.3	6
Dipper at Rowe	4.7 \pm 1.3	3	44.5 \pm 22.7	4	0.0 \pm 0.0	2	0.2 \pm 0.2	5	0.3 \pm 0.3	3	-	0
Red Rock	9.8 \pm 8.8	4	3.1 \pm 0.8	8	2.8 \pm 0.5	4	1.3 \pm 0.2	6	2.2 \pm 0.5	6	4.0 \pm 1.2	4
Sewage Lagoon	1.7 \pm 0.9	3	2.6 \pm 0.9	7	0.8 \pm 0.2	4	1.0 \pm 0.4	6	2.0 \pm 1.4	4	0.2 \pm 0.2	4
Unburn	1.5 \pm 0.6	24	1.3 \pm 0.2	49	2.3 \pm 0.6	29	1.2 \pm 0.4	38	1.2 \pm 0.3	42	1.3 \pm 0.4	34
Belly Bend	3.8 \pm 3.4	4	0.3 \pm 0.2	6	0.5 \pm 0.5	2	0.5 \pm 0.2	6	0.2 \pm 0.2	6	-	0
Boundary Bay	-	0	2.0 \pm 0.9	4	1.2 \pm 0.9	4	0.6 \pm 0.2	7	4.5 \pm 0.6	4	0.0 \pm 0.0	7
Lone Lake	1.0 \pm 0.6	3	3.2 \pm 0.7	9	5.7 \pm 2.0	7	5.1 \pm 1.7	7	0.8 \pm 0.4	6	1.3 \pm 0.8	6
Lost Lake	2.2 \pm 1.3	4	1.6 \pm 0.7	7	2.4 \pm 0.9	7	-	0	2.3 \pm 0.9	6	4.7 \pm 1.4	7
Sofa Burn	1.5 \pm 0.9	4	0.0 \pm 0.0	4	0.0 \pm 0.0	2	0.3 \pm 0.2	6	0.2 \pm 0.2	6	0.0 \pm 0.0	4
Wishbone	0.0 \pm 0.0	5	0.9 \pm 0.3	12	0.0 \pm 0.0	3	0.3 \pm 0.2	6	0.0 \pm 0.0	6	0.0 \pm 0.0	5
Yarrow Cabin	1.0 \pm 0.6	4	0.6 \pm 0.2	7	1.2 \pm 0.6	4	0.0 \pm 0.0	6	1.6 \pm 1.1	8	0.8 \pm 0.5	5
Total (burn and unburn)	2.7 \pm 0.8	47	4.3 \pm 1.4	84	2.1 \pm 0.4	51	3.7 \pm 1.2	71	1.2 \pm 0.2	71	1.5 \pm 0.4	56

Appendix I: 40 kHz *Myotis* Feeding Buzz Analysis

Table I.1. The percent of subsampled (i.e. 20 individual files or 10% of all files, whichever was greater) 40 kHz *Myotis* passes that contained feeding buzzes, for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020. Note that the values represent weighted means (i.e. sites with larger sample sizes contribute more to the mean than sites with smaller sample sizes). For cells with no entries, no data for that site and year was included in the analysis.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	1.0%	7.6%	11.1%	6.7%	11.0%	8.5%	11.8%	10.4%	8.5%
Bison Paddock	1.7%	15.3%	25.0%	15.9%	12.0%	4.0%	10.3%	8.6%	12.0%
Blakiston Roadside	1.4%	7.8%	5.7%	5.1%	15.8%	-	40.7%	24.1%	12.5%
Cameron Lake	0.0%	6.8%	4.7%	3.7%	12.4%	5.0%	9.2%	8.9%	6.5%
Dipper at Rowe	0.0%	5.5%	0.0%	2.4%	4.0%	19.3%	-	9.8%	5.9%
Red Rock	0.0%	8.3%	16.8%	9.0%	9.7%	11.3%	0.0%	8.4%	8.7%
Sewage Lagoon	3.3%	4.3%	2.5%	3.6%	11.3%	6.3%	3.8%	7.7%	5.6%
Unburn	12.0%	8.3%	9.6%	9.5%	13.3%	10.6%	5.2%	9.8%	9.7%
Belly Bend	8.8%	11.3%	8.5%	10.0%	17.2%	16.7%	-	16.9%	13.5%
Boundary Bay	-	8.5%	10.0%	9.3%	2.9%	7.5%	3.6%	4.2%	5.7%
Lone Lake	24.0%	4.9%	14.7%	11.5%	17.3%	8.3%	6.7%	11.1%	11.3%
Lost Lake	16.8%	19.0%	5.7%	13.3%	-	9.7%	10.3%	10.0%	11.9%
Sofa Burn	20.8%	14.5%	19.0%	17.9%	27.8%	10.3%	8.5%	16.4%	17.0%
Wishbone	4.0%	4.4%	10.3%	5.2%	3.8%	16.2%	0.0%	7.5%	6.2%
Yarrow Cabin	0.0%	2.3%	2.5%	1.9%	8.2%	6.0%	0.0%	5.2%	3.7%
Total (burn and unburn)	6.6%	8.0%	10.2%	8.3%	12.2%	9.9%	7.6%	10.1%	9.2%

Table I.2. The number of subsampled files examined for each acoustic detector site in Waterton Lakes National Park, Alberta from 2015 to 2020, for the analyses examining the percent of passes that contained 40 kHz *Myotis* feeding buzzes.

Site name	2015	2016	2017	Pre-fire total	2018	2019	2020	Post-fire total	Total (pre and post)
Burn	258	419	308	985	490	344	367	1201	2186
Bison Paddock	60	79	100	239	69	65	194	328	567
Blakiston Roadside	41	22	9	72	82	-	12	94	166
Cameron Lake	60	103	31	194	128	100	80	308	502
Dipper at Rowe	26	70	40	136	34	30	-	64	200
Red Rock	11	29	48	88	24	69	14	107	195
Sewage Lagoon	60	116	80	256	153	80	67	300	556
Unburn	645	1165	511	2321	630	1001	753	2384	4705
Belly Bend	54	97	17	168	105	120	-	225	393
Boundary Bay	-	98	80	178	139	75	43	257	435
Lone Lake	115	199	133	447	146	120	133	399	846
Lost Lake	305	418	140	863	-	410	446	856	1719
Sofa Burn	70	57	33	160	120	78	56	254	414
Wishbone	58	204	28	290	50	86	54	190	480
Yarrow Cabin	43	92	80	215	70	112	21	203	418
Total (burn and unburn)	903	1548	819	3306	1120	1345	1120	3585	6891

Table I.3. Kruskal-Wallis rank sum test results for the proportion of echolocation calls that contained feeding buzzes for 40kHz *Myotis* (40kFeedBuzz) in Waterton Lakes National Park, Alberta from 2015 to 2020. The three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) were tested separately. Significant values ($P < 0.05$) are bolded.

Response variable	Fire metric	Chi-squared	df	P-value
40kFeedBuzz proportion	PrePost123	6.621	3	0.085
40kFeedBuzz proportion	PrePostUnburnBurn	8.800	3	0.032
40kFeedBuzz proportion	PrePost123UnburnBurn	17.273	7	0.016

Table I.4. Dunn’s test of multiple comparisons for the three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) tested in the 40k *Myotis* feeding buzz (40kFeedBuzz) Kruskal-Wallis rank sum test. A Bonferroni correction to control for the experiment-wise error rate was added. The significant value ($P < 0.05$) is bolded.

Fire metric	Variable	Effect direction	Z – test statistic	Bonferroni adjusted P -value
40kFeedBuzz PrePost123	PrePost123 groups are not significantly different			
40kFeedBuzz PrePostUnburnBurn	PreBurn – PreUnburn	PreBurn < PreUnburn	-2.868	0.025
	PostUnburn – PreUnburn		-0.732	1.000
	PostBurn – PreBurn		1.967	0.295
	PostBurn – PostUnburn		-0.101	1.000
40kFeedBuzz PrePost123UnburnBurn	PreBurn – PreUnburn		-2.868	0.116*
	PostUnburn1 – PreUnburn		-0.114	1.000
	PostUnburn2 – PreUnburn		0.769	1.000
	PostUnburn3 – PreUnburn		-2.456	0.393*
	PostBurn1 – PreBurn		2.065	1.000*
	PostBurn2 – PreBurn		1.065	1.000
	PostBurn3 – PreBurn		0.823	1.000
	PostUnburn1 – PostUnburn2		-0.715	1.000
	PostUnburn1 – PostUnburn3		1.938	1.000
	PostUnburn2 – PostUnburn3		2.724	0.180*
40kFeedBuzz PrePost123UnburnBurn	PostBurn1 – PostBurn2		0.654	1.000
	PostBurn1 – PostBurn3		0.764	1.000
	PostBurn2 – PostBurn3		0.137	1.000

*Variable comparisons were significant if the unadjusted P -values were used.

Appendix J: 40 kHz *Myotis* Activity Analysis

Table J.1. Variable names, effect direction, parameter estimates, standard errors, and *P*-values for each 40 kHz *Myotis* (40kMyotis) fire metric model. Numerical variables (i.e. WaterDistance, NightSumPrecip and SunsetTemp) are scaled. For the fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn), the significant differences are explored in the associated post hoc table. A "positive" effect direction denotes an increase in the activity level as the variable increases; a "negative" effect direction denotes a decrease. For "NearestWaterFeature" only the most significantly different feature is displayed. R^2_c and R^2_m are conditional and marginal R^2 , respectively.

Model	Variable	Effect direction	Parameter estimate	Standard error	<i>P</i> -value	R^2_c	R^2_m
40kMyotis PrePost123	NearestWaterFeature	Lake	2.307	0.626	<0.001	0.665	0.382
	NightSumPrecip	Negative	-0.174	0.061	0.005		
	SunsetTemp	Positive	0.166	0.068	0.014		
40kMyotis PrePostUnburnBurn	PrePostUnburnBurn	Significant – see Table J.2				0.686	0.437
	NearestWaterFeature	Lake	2.139	0.604	<0.001		
	NightSumPrecip	Negative	-0.154	0.060	0.011		
	SunsetTemp	Positive	0.225	0.068	<0.001		
40kMyotis PrePost123UnburnBurn	PrePost123UnburnBurn	Significant – see Table J.2				0.701	0.513
	WaterDistance	Negative	-0.506	0.247	0.040		
	NearestWaterFeature	Lake	1.891	0.551	<0.001		
	SunsetTemp	Positive	0.261	0.068	<0.001		

Table J.2. Post hoc results for the 40 kHz *Myotis* (40kMyotis) fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn). The variable comparisons, effect direction, test type (i.e. summary output or estimated marginal means post hoc), parameter estimates, standard errors and *P*-values are reported. The estimated marginal means (EMMs) post-hoc tests have lower statistical power compared to the summary output due to how many groups are compared. Therefore, the summary output was used when available. Note, due to how the comparison was conducted in the summary versus EMMs tests, the signs for the parameter estimates are opposite. For the summary output, a "positive" parameter estimate denotes that the second listed factor is greater than the first factor; a "negative" parameter estimate denotes that the first listed factor is greater than the second factor. For the EMMs, a "positive" parameter estimate denotes that the first listed factor is greater than the second factor; a "negative" parameter estimate denotes that the second listed factor is greater than the first factor. Significant values ($P < 0.05$) are bolded.

Model	Variable	Effect direction	Summary or EMMs	Parameter estimate	Standard error	<i>P</i> -value
40kMyotis PrePost123	PrePost123 not significant in model					
40kMyotis PrePostUnburnBurn	PreUnburn – PreBurn	PreUnburn > PreBurn	Summary	-1.073	0.476	0.024
	PreUnburn – PostUnburn	PreUnburn > PostUnburn	Summary	-0.312	0.150	0.037
	PreBurn – PostBurn	PreBurn < PostBurn	EMMs	-0.624	0.172	0.002
	PostUnburn – PostBurn		EMMs	0.136	0.480	0.992
40kMyotis PrePost123UnburnBurn	PreUnburn – PreBurn	PreUnburn > PreBurn	Summary	-1.442	0.462	0.002
	PreUnburn – PostUnburn1		Summary	-0.352	0.206	0.087
	PreUnburn – PostUnburn2		Summary	-0.035	0.196	0.859
	PreUnburn – PostUnburn3	PreUnburn > PostUnburn3	Summary	-0.821	0.221	<0.001
	PreBurn – PostBurn1	PreBurn < PostBurn1	EMMs	-0.937	0.231	0.001
	PreBurn – PostBurn2		EMMs	-0.251	0.238	0.966
	PreBurn – PostBurn3		EMMs	-0.667	0.271	0.214
	PostUnburn1 – PostUnburn2		EMMs	-0.317	0.238	0.887
	PostUnburn1 – PostUnburn3		EMMs	0.468	0.262	0.628
	PostUnburn2 – PostUnburn3	PostUnburn2 > PostUnburn3	EMMs	0.786	0.257	0.047
	PostBurn1 – PostBurn2		EMMs	0.686	0.287	0.246
	PostBurn1 – PostBurn3		EMMs	0.270	0.322	0.991
	PostBurn2 – PostBurn3		EMMs	-0.416	0.325	0.906

Appendix K: Big Brown/Silver-Haired Bat Activity Analysis

Table K.1. Variable names, effect direction, parameter estimates, standard errors, and *P*-values for each big brown/silver-haired bat (EPFULANO) fire metric model. Numerical variables (i.e. TownsiteDistance, and SunsetTemp) are scaled. For the fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn), the significant differences are explored in the associated post hoc table. A "positive" effect direction denotes an increase in the activity level as the variable increases; a "negative" effect direction denotes a decrease. For "NearestWaterFeature" only, the most significantly different feature is displayed. R^2_c and R^2_m are conditional and marginal R^2 , respectively.

Model	Variable	Effect direction	Parameter estimate	Standard error	P-value	R^2_c	R^2_m
EPFULANO PrePost123	PrePost123		Significant – see Table K.2			0.681	0.524
	TownsiteDistance	Positive	0.563	0.207	0.006		
	NearestWaterFeature	Lake	1.734	0.522	<0.001		
	SunsetTemp	Positive	0.547	0.074	<0.001		
EPFULANO PrePostUnburnBurn	PrePostUnburnBurn		Significant – see Table K.2			0.680	0.533
	TownsiteDistance	Positive	0.635	0.227	0.005		
	NearestWaterFeature	Lake	1.712	0.506	<0.001		
	SunsetTemp	Positive	0.566	0.074	<0.001		
EPFULANO PrePost123UnburnBurn	PrePost123UnburnBurn		Significant – see Table K.2			0.690	0.539
	TownsiteDistance	Positive	0.624	0.232	0.007		
	NearestWaterFeature	Lake	1.729	0.515	<0.001		
	SunsetTemp	Positive	0.571	0.075	<0.001		

Table K.2. Post hoc results for the big brown/silver-haired bat (EPFULANO) fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn). The variable comparisons, effect direction, test type (summary output or estimated marginal means post hoc), parameter estimates, standard errors and *P*-values are reported. The estimated marginal means (EMMs) post-hoc tests have lower statistical power compared to the summary output due to how many groups are compared. Therefore, the summary output was used when available. Note, due to how the comparison was conducted in the summary versus EMMs tests, the signs for the parameter estimates are opposite. For the summary output, a "positive" parameter estimate denotes that the second listed factor is greater than the first factor; a "negative" parameter estimate denotes that the first listed factor is greater than the second factor. For the EMMs, a "positive" parameter estimate denotes that the first listed factor is greater than the second factor; a "negative" parameter estimate denotes that the second listed factor is greater than the first factor. Significant values ($P < 0.05$) are bolded.

Model	Variable	Effect direction	Summary or EMMs	Parameter estimate	Standard error	<i>P</i> -value
EPFULANO PrePost123	Pre – Post1	Pre > Post1	Summary	-0.452	0.171	0.008
	Pre – Post2	Pre > Post2	Summary	-0.478	0.167	0.004
	Pre – Post3	Pre > Post3	Summary	-0.615	0.187	0.001
	Post1 – Post2		EMMs	0.026	0.205	0.999
	Post1 – Post3		EMMs	0.162	0.223	0.885
	Post2 – Post3		EMMs	0.137	0.217	0.922
EPFULANO PrePostUnburnBurn	PreUnburn – PreBurn		Summary	0.001	0.455	0.998
	PreUnburn – PostUnburn	PreUnburn > PostUnburn	Summary	-0.721	0.158	<0.001
	PreBurn – PostBurn		EMMs	0.178	0.194	0.797
	PostUnburn – PostBurn		EMMs	-0.545	0.460	0.638
EPFULANO PrePost123UnburnBurn	PreUnburn – PreBurn		Summary	-0.007	0.463	0.988
	PreUnburn – PostUnburn1	PreUnburn > PostUnburn1	Summary	-0.775	0.220	<0.001
	PreUnburn – PostUnburn2	PreUnburn > PostUnburn2	Summary	-0.481	0.210	0.022
	PreUnburn – PostUnburn3	PreUnburn > PostUnburn3	Summary	-1.050	0.239	<0.001
	PreBurn – PostBurn1		EMMs	0.068	0.255	1.000
	PreBurn – PostBurn2		EMMs	0.452	0.271	0.706
	PreBurn – PostBurn3		EMMs	0.062	0.291	1.000
	PostUnburn1 – PostUnburn2		EMMs	-0.294	0.264	0.954
	PostUnburn1 – PostUnburn3		EMMs	0.275	0.285	0.979
	PostUnburn2 – PostUnburn3		EMMs	0.569	0.272	0.419
	PostBurn1 – PostBurn2		EMMs	0.384	0.308	0.918
	PostBurn1 – PostBurn3		EMMs	-0.006	0.343	1.000
	PostBurn2 – PostBurn3		EMMs	-0.390	0.339	0.945

Appendix L: Hoary Bat Activity Analysis

Table L.1. Variable names, effect direction, parameter estimates, standard errors, and *P*-values for each hoary bat (LACI) fire metric model. The numerical variable (i.e. SunsetTemp) is scaled. For the fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn), the significant differences are explored in the associated post hoc table. A “positive” effect direction denotes an increase in the activity level as the variable increases; a “negative” effect direction denotes a decrease. For “NearestWaterFeature” only the most significantly different feature is displayed. R^2_c and R^2_m are conditional and marginal R^2 , respectively.

Model	Variable	Effect direction	Parameter estimate	Standard error	<i>P</i> -value	R^2_c	R^2_m
LACI PrePost123	PrePost123		Significant – see Table L.2			0.676	0.469
	NearestWaterFeature	River	2.920	0.891	0.001		
	SunsetTemp	Positive	0.580	0.081	<0.001		
LACI PrePostUnburnBurn		Model failed to converge – see Table L.3 and L.4					
LACI PrePost123UnburnBurn	PrePost123UnburnBurn		Significant – see Table L.2			0.715	0.704
	NearestWaterFeature	River	2.101	0.351	<0.001		
	SunsetTemp	Positive	0.572	0.081	<0.001		

Table L.2. Post hoc results for the hoary bat (LACI) fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn). The variable comparisons, effect direction, test type (i.e. summary output or estimated marginal means post hoc), parameter estimates, standard errors and *P*-values are reported. The estimated marginal means (EMMs) post-hoc tests have lower statistical power compared to the summary output due to how many groups are compared. Therefore, the summary output was used when available. Note, due to how the comparison was conducted in the summary versus EMMs tests, the signs for the parameter estimates are opposite. For the summary output, a “positive” parameter estimate denotes that the second listed factor is greater than the first factor; a “negative” parameter estimate denotes that the first listed factor is greater than the second factor. For the EMMs, a “positive” parameter estimate denotes that the first listed factor is greater than the second factor; a “negative” parameter estimate denotes that the second listed factor is greater than the first factor. Significant values ($P < 0.05$) are bolded.

Model	Variable	Effect direction	Summary or EMMs	Parameter estimate	Standard error	<i>P</i> -value
LACI PrePost123	Pre – Post1	Pre > Post1	Summary	-1.281	0.196	<0.001
	Pre – Post2	Pre > Post2	Summary	-1.659	0.201	<0.001
	Pre – Post3	Pre > Post3	Summary	-0.533	0.207	0.010
	Post1 – Post2		EMMs	0.377	0.238	0.385
	Post1 – Post3	Post1 < Post3	EMMs	-0.749	0.251	0.015
	Post2 – Post3	Post2 < Post3	EMMs	-1.126	0.255	<0.001
LACI PrePostUnburnBurn		Model failed to converge – see Table L.3 and L.4				
LACI PrePost123UnburnBurn	PreUnburn – PreBurn	PreUnburn > PreBurn	Summary	-0.986	0.226	<0.001
	PreUnburn – PostUnburn1	PreUnburn > PostUnburn1	Summary	-0.852	0.240	<0.001
	PreUnburn – PostUnburn2	PreUnburn > PostUnburn2	Summary	-1.615	0.234	<0.001
	PreUnburn – PostUnburn3		Summary	-0.403	0.250	0.106
	PreBurn – PostBurn1	PreBurn > PostBurn1	EMMs	2.709	0.425	<0.001
	PreBurn – PostBurn2	PreBurn > PostBurn2	EMMs	1.779	0.385	<0.001
	PreBurn – PostBurn3		EMMs	0.819	0.332	0.209
	PostUnburn1 – PostUnburn2		EMMs	0.762	0.277	0.109
	PostUnburn1 – PostUnburn3		EMMs	-0.449	0.299	0.807
	PostUnburn2 – PostUnburn3	PostUnburn2 < PostUnburn3	EMMs	-1.211	0.296	0.001
	PostBurn1 – PostBurn2		EMMs	-0.929	0.541	0.675
	PostBurn1 – PostBurn3	PostBurn1 < PostBurn3	EMMs	-1.890	0.505	0.005
	PostBurn2 – PostBurn3		EMMs	-0.961	0.462	0.428

Table L.3. Kruskal-Wallis rank sum test results for hoary bats (LACI) in Waterton Lakes National Park, Alberta from 2015 to 2020 for the fire metric PrePostUnburnBurn. The significant value ($P < 0.05$) is bolded.

Response variable	Fire metric	Chi-squared	df	P-value
LACI echolocation count	PrePostUnburnBurn	111.78	3	<0.001

Table L.4. Dunn's test of multiple comparisons for the fire metric PrePostUnburnBurn tested in the hoary bat (LACI) Kruskal-Wallis rank sum test. A Bonferroni correction to control for the experiment-wise error rate was added. Significant values ($P < 0.05$) are bolded.

Fire metric	Variable	Effect direction	Z – test statistic	Bonferroni adjusted P-value
LACI PrePostUnburnBurn	PreBurn – PreUnburn	PreBurn < PreUnburn	-4.553	<0.001
	PostUnburn – PreUnburn	PostUnburn < PreUnburn	-3.660	0.002
	PostBurn – PreBurn	PostBurn < PreBurn	-5.488	<0.001
	PostBurn – PostUnburn	PostBurn < PostUnburn	-7.221	<0.001

Appendix M: Long-Eared *Myotis* Activity Analysis

Table M.1. Kruskal-Wallis rank sum test results for long-eared *Myotis* (MYEV) in Waterton Lakes National Park, Alberta from 2015 to 2020. The three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) were tested separately. Significant values ($P < 0.05$) are bolded.

Response variable	Fire metric	Chi-squared	df	P-value
MYEV echolocation count	PrePost123	20.520	3	<0.001
MYEV echolocation count	PrePostUnburnBurn	43.401	3	<0.001
MYEV echolocation count	PrePost123UnburnBurn	47.011	7	<0.001

Table M.2. Dunn’s test of multiple comparisons for the three different fire metrics (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) tested in the long-eared *Myotis* (MYEV) Kruskal-Wallis rank sum test. A Bonferroni correction to control for the experiment-wise error rate was added. Significant values ($P < 0.05$) are bolded.

Fire metric	Variable	Effect direction	Z – test statistic	Bonferroni adjusted <i>P</i> -value
MYEV PrePost123	Post1 – Pre		-2.333	0.118
	Post2 – Pre	Post2 < Pre	-3.551	0.002
	Post3 – Pre	Post3 < Pre	-3.495	0.003
	Post1 – Post2		1.015	1.000
	Post1 – Post3		1.162	1.000
	Post2 – Post3		0.208	1.000
MYEV PrePostUnburnBurn	PreBurn – PreUnburn	PreBurn > PreUnburn	4.606	<0.001
	PostUnburn – PreUnburn		-1.868	0.370
	PostBurn – PreBurn	PostBurn < PreBurn	-4.363	<0.001
	PostBurn – PostUnburn		1.815	0.417
MYEV PrePost123UnburnBurn	PreBurn – PreUnburn	PreBurn > PreUnburn	4.606	<0.001
	PostUnburn1 – PreUnburn		-1.433	1.000
	PostUnburn2 – PreUnburn		-1.226	1.000
	PostUnburn3 – PreUnburn		-1.372	1.000
	PostBurn1 – PreBurn		-2.060	1.000
	PostBurn2 – PreBurn	PostBurn2 < PreBurn	-3.868	0.003
	PostBurn3 – PreBurn	PostBurn3 < PreBurn	-3.563	0.010
	PostUnburn1 – PostUnburn2		-0.212	1.000
	PostUnburn1 – PostUnburn3		-0.003	1.000
	PostUnburn2 – PostUnburn3		0.203	1.000
	PostBurn1 – PostBurn2		1.620	1.000
	PostBurn1 – PostBurn3		1.568	1.000
	PostBurn2 – PostBurn3		0.068	1.000

Appendix N: Significant Site and Environmental Variables for the Generalized Linear Mixed Models

The site and environmental variables that were significant covariates in the generalized linear mixed models examining the effect of each fire metric (i.e. PrePost123, PrePostUnburnBurn, PrePost123UnburnBurn) on activity levels of each species/species group (i.e. 40 kHz *Myotis*, big brown/silver-haired bat, hoary bat) included: sunset temperature (°C), total nightly precipitation (mm), distance from the acoustic station to the Waterton townsite (km), nearest water feature (i.e. creek, lake, river or wetland), and distance from the acoustic station to the nearest water source (km). Including site as a random effect improved the performance of all models for all species/species groups (i.e. all conditional R^2 values were greater than the marginal R^2 values).

All species/species groups showed the expected trends in relation to the environmental variables; activity was highest on warm nights without rain. This trend has been well established in previous studies (e.g., Erickson and West 2002, Vonhof 2006, Muthersbaugh et al. 2019). Big brown/silver-haired bat activity increased as the distance to the Waterton townsite increased. It is unclear why this variable was significant as the sites that were located at the greatest distance from the townsite had variable habitats (e.g. coniferous and deciduous forest types, river, lake and creek sites, and elevations that differed by more than 650 meters).

The nearest water feature was a significant variable in all models for all species/species groups. 40 kHz *Myotis* and big brown/silver-haired bats were most associated with lake sites, and hoary bat activity was highest at the river site. The top two sites with the highest activity were lake sites (i.e. Lone Lake and Lost Lake). The mean bat passes per night for these sites were between 1.9 (Lone Lake) and 5.5 (Lost Lake) times higher than the site with the third highest activity level (i.e. Bison Paddock, a wetland site). Both Lone Lake and Lost Lake are similar in terms of habitat (i.e. high elevation, unburned coniferous sites that are far from the Waterton townsite and close to water) and were monitored later in the season (i.e. end of July). As there was only one river site (i.e. Belly Bend), it is difficult to conclude if hoary bats were drawn to the river as a suitable flyway corridor or foraging area, or if there was a different site-specific characteristic that made it preferable.

Although distance to the nearest water source was only significant in one model (40 kHz *Myotis* PrePost123UnburnBurn), water distance may have influenced the significance of the nearest water feature variable. Previous studies have found that bats are closely associated with water (Mackey and Barclay 1988, Grindal et al. 1999, Evelyn et al. 2004). All of the acoustic stations at the lake sites were within ten meters of water and the river site was five meters from water. Both the creek and wetland sites were typically further from water, with the creek sites ranging from 20 to 75 meters from water and the wetland sites ranging from ten to 350 meters from water. As high frequency sounds attenuate quickly, bats would likely need to be within approximately 25 meters of the microphone for their echolocation calls to be recorded (Adams et al. 2012). However, this value varies depending on several factors, including the atmospheric conditions (e.g. temperature and relative humidity), and frequency of the echolocation call (Goerlitz 2018, Russo et al. 2018a).