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State-wide assessment of status, predicted distribution, and landscape-level habitat suitability of amphibians and reptiles in Montana

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STATE-WIDE ASSESSMENT OF STATUS, PREDICTED DISTRIBUTION, AND LANDSCAPE-LEVEL HABITAT SUITABILITY OF AMPHIBIANS AND REPTILES IN MONTANA

By

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Beginning in the late 1980s herpetologists began to realize that amphibians around the world had undergone, and were continuing to undergo, declines, extirpations, and extinctions. In most cases, detections of declines and determinations of the underlying causes has been hampered by a lack of available baseline information on distribution and status.

This project was a cooperative effort to address these data deficiencies for amphibians and reptiles in Montana. Watersheds with greater than 30 percent federal or state land ownership were randomly selected for survey in each of 11 geographic strata. Visual encounter and dipnet surveys of all standing water bodies on public lands within these watersheds yielded watershed and site occupancy estimates as a measure of status. Occupancy estimates from this first-ever state-wide base level assessment can be more validly used for future comparisons with future status assessment, provided additional support for declines in Western Toad (*Bufo boreas)* and Northern Leopard Frog (*Rana pipiens*) populations in western Montana, and identified a variety of conservation issues of concern that can be addressed through management actions (e.g., clear evidence for negative impacts of fish and importance of maintaining natural disturbance regimes such as flooding, beaver, and fire).

The information gathered during field inventories was combined with other existing information and used in maximum entropy modeling to predict state-wide distribution and habitat suitability for all of Montana's amphibians and reptiles. These models out performed GAP analysis models by simultaneously reducing the area predicted and omission error rates. Among other things, models identified scale dependent responses to environmental variables, potentially isolated populations in need of conservation efforts, and areas that are critical for maintaining landscape connectivity.

In conjunction with field inventories, a state-wide assessment of the distribution of the pathogenic chytrid fungus (*Batrachochytrium dendrobatidis*) (Bd) was undertaken using PCR-based detection in skin swabs or tissue samples. Bd was found across Montana in 6 of the 9 species tested at a variety of elevations, habitats, and distances from human activities. The widespread presence of Bd highlights the need for additional studies and measures to prevent the spread of Bd and other novel pathogens.

DEDICATION

Many wildlife professionals and the general public owe their first real connections to the natural world to an experience with amphibians and reptiles. Watching amphibian embryos develop through clear jelly layers, tadpoles sprout limbs while undergoing a radical rearrangement of body parts during metamorphosis, the margins of a wetland in motion with tens of thousands of newly metamorphosed amphibians, gartersnakes capture and eat frogs, or lizards move at seemingly impossible speeds across hot rocks to evade capture are all powerful experiences and… there is no going back. I dedicate this dissertation to the amphibians and reptiles of Montana and all individuals that have added to our understanding of them or worked for their protection.

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Finally, I would like to thank my family. My parents, Marvin Maxell and Diana Maxell surrounded me with every farm animal imaginable and put me behind a team of dogs to sled through the snow covered Uintah Mountains at an early age. My dad got me hooked on naming every animal in sight from the time that I could speak and my mom taught me flowers and to respect every living thing. My dad is the hardest working person I know and my mom is a biggest do-gooder I know – I hope I can survive this dangerous combination! I also want to thank my wife Sarah for putting up with my absences and graciously assisting me in the field until I managed to wear the skin off her heels - I promise to slow down in the future. I love you Sarah!

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION AND OVERVIEW: ASSESSING THE STATUS OF LENTIC BREEDING AMPHIBIANS AND AQUATIC REPTILES TO PROVIDE TOOLS FOR NATURAL RESOURCE MANAGERS

Importance of Amphibians

Amphibians play important ecological roles in shaping terrestrial and aquatic communities (Seale 1980, Wilbur 1980, 1997). As ectotherms with complex life histories they are up to 50 times more efficient than mammals or birds at processing prey into biomass and can, therefore, play key roles in transferring energy up the food chain and between aquatic and terrestrial communities (Burton and Likens 1975a, 1975b, Pough 1980, Wilbur 1980). Amphibians also contribute a great deal to human welfare. They are an important source of protein in many impoverished societies and have been key study organisms for vertebrate anatomy, neurology, physiology, embryology, developmental biology, genetics, evolutionary biology, animal behavior, and community ecology (Stebbins and Cohen 1995; Pough et al. 1998). Eggs, larvae, and adults have been extensively used in toxicology studies of chemical contaminants that may impact human health (Harfenist et al. 1989, Hayes et al. 2003). Skin secretions of some species show promise as antibiotics and as nonaddictive pain killers that are 200 times more powerful than morphine (Stebbins and Cohen 1995). Finally, some species are valuable bioindicators of the health of certain environments because they have complex life cycles with both aquatic and terrestrial life history stages that are philopatric to specific breeding, foraging, and overwintering sites connected by habitats suitable for migration (Dole 1965, Ewert 1969, Duellman and Trueb 1986, Sinsch 1990, Patla 1997, Welsh and Ollivier 1998).

Amphibian Declines

At the First World Congress of Herpetology in 1989 herpetologists began to suspect that amphibians around the world were undergoing declines, extirpations, and extinctions (Blaustein and Wake 1990, Stebbins and Cohen 1995). Hypothesized causative agents of declines included: loss, deterioration, and fragmentation of aquatic and terrestrial habitats (Van Rooy and Stumpel 1995, Lind et al. 1996, Beebee 1997), introduction of nonindigenous species (Bradford 1989, Bradford et al. 1993, Fisher and Schaffer 1996, Gamradt and Kats 1996, Hecnar and M'Closkey 1997), environmental pollutants (Lewis et al. 1985, Dunson et al. 1992, Sparling et al. 2001, Hayes et al. 2003), increased ambient UV-B radiation (Blaustein et al. 1994, 1995), climate change (Pounds and Crump 1994, Stewart 1995, Pounds et al. 1999), pathogens (Carey 1993, Berger et al. 1998, Carey et al. 1999, Daszak et al. 1999, Lips 1999), human commerce (Jennings and Hayes 1985, Pough et al. 1998), and synergistic interactions between causative agents (Carey and Bryant 1995, Kiesecker and Blaustein 1995, Pounds et al. 2006). With the possible exception of increased ambient UV-B radiation (Adams et al. 2005), there has been growing support for these agents of decline over the past 20 years and there is a consensus that declines have occurred at a global scale and that amphibians are far more threatened than either birds or mammals (Houlahan et al. 2000, Wake 2003, Stuart et al. 2004, Lannoo 2005). A recent Global Amphibian Assessment (GAA) determined that 43% of amphibian species are experiencing some form of declines, 33% are threatened with extinction, and at least 427 species (7%) are critically endangered with extinction (Stuart et al. 2004).

The GAA also determined that 23% of amphibian species were so poorly understood that their status was unable to be ranked with even simplistic criteria; a far higher percentage than birds (1.8%) and mammals (3.8%). The lack of information available for amphibian species may be one of the greatest threats faced by the group because 30% of the 6,485 known species have been described in the last 17 years, after amphibian declines were first suspected (Köhler et al. 2005, AmphibiaWeb 2009). This raises the very real possibility that hundreds, perhaps thousands, of recently extant amphibian species may go extinct without ever having been formally described. While many of the newly described species are from the tropics, there is also a lack of information for most temperate amphibian species. Indeed, throughout much of the 1990s there was debate as to whether observed declines and losses were just a normal part of the natural variability of populations (Pechmann et al. 1991, Blaustein 1994, Pechmann and Wilbur 1994).

In Montana, almost 60% of the amphibians and over 40% of the aquatic reptiles are state Species of Concern and a lack of baseline surveys has severely hampered our ability to understand the status of populations (Maxell et al. 2003, Werner et al. 2004, Maxell et al. 2009, MNHP and MFWP 2009). Furthermore, as in much of western North America, relatively little is known about the demography and life history of most of these species because most have only been studied in detail at a handful of locations at best. Thus, for most of these species there is currently no way to place the results of experimental studies of suspected mechanisms of decline in a population level context or the context of the results of regional monitoring programs in order to thoroughly understand the causes of decline (Biek et al. 2002).

A Multi-tiered Strategy to Assess Status of Amphibians and Aquatic Reptiles in Montana

A meaningful sampling unit is the necessary core of a multi-tiered strategy for monitoring lentic breeding amphibians and aquatic reptiles. Local watersheds are an ideal sampling unit not only because they encompass networks of habitat patches and local breeding populations (e.g., Funk et al. 2005), but because they encompass natural disturbance regimes (e.g., flooding and beaver) which create new habitat patches and are commonly used as management units by federal and state agencies and tribal governments so often encompass anthropogenic disturbance regimes as well. The U.S. Geological Survey (USGS) has defined an integrated series of watersheds for much of the U.S. (Seaber et al. 1984) and the smallest watershed unit they have defined, a 6th code or 12-digit hydrologic unit code (HUC), represents a relatively uniform naturally defined portion of the environment.

A multi-tiered strategy to assess the status of lentic breeding amphibians and aquatic reptiles in Montana that is based on a 12-digit HUC watershed would: (1) carry out base-level inventories of watershed and lentic site breeding rates as a measure of regional and local watershed status; (2) initiate long-term programs for monitoring status and trends in watershed and lentic site occupancy rates; (3) initiate long-term intensive monitoring of population dynamics and vital rates of individual species in a few sampling units representative of the range of latitudes and elevations occupied by each species; (4)

conduct experimental research at the scale of sampling units used for inventory and monitoring in order to understand underlying causes of the patterns of site occupancy and demography observed; and (5) use the data gathered to create easily interpretable models that can be used by natural resource managers to prioritize conservation efforts at multiple spatial scales across Montana. This strategy is compatible with that recently articulated by the U.S. Geological Survey's Amphibian Research and Monitoring Initiative (ARMI) (Hall and Langtimm 2001, Corn et al. 2005a-c, Muths et al. 2005)

Tools for Natural Resource Managers

The ultimate goal of fish and wildlife research is to provide management tools that will allow natural resource managers to conserve and protect fish and wildlife populations and their habitats for current and future generations. During the course of my graduate research, I have summarized existing information on amphibians and reptiles in books and reports readily accessible to resource managers and the general public (Maxell and Hokit 1999, Maxell 2000, Maxell et al. 2003, 2009, Werner et al. 2004), used a variety of modeling techniques to evaluate the status of fish, amphibian, and reptile populations (Maxell 1998, Hart et al. 1998, Maxell 1999, Biek et al. 2002), conducted my own research on the distribution, status, and natural history of Montana's amphibian and aquatic reptile species (Maxell 2002a-b, Maxell et al. 2002, Maxell 2004a-e, Maxell 2005a-b, Maxell 2006) and mentored undergraduate projects on the reproductive ecology, prey, and gartersnake predators of the Columbia Spotted Frog (*Rana luteiventris*) (Easley 2002, Wilson 2003, Thayer 2004, Thompson 2004). In this dissertation, I extended these efforts toward fulfillment of a multi-tiered strategy for assessing the status of Montana's amphibians and aquatic reptiles.

The main goals of my dissertation were to:

1. assess the statewide distribution and status of 10 lentic breeding amphibians and 4 aquatic reptile species using a probability-based sampling scheme centered on 12-digit HUC watersheds in order to make inference to regional watershed and site breeding and occupancy rates as a baseline measure of status

2. use detection data from our surveys to assess the suitability of local environmental variables for 10 lentic breeding amphibian and 4 aquatic reptile species in classification tree diagrams that are easily interpretable in a variety of ecological settings across Montana
3. use positive data from our surveys and other data sources to create models predicting the regional and landscape-level suitability of habitats for 31 individual species beyond areas that have been inventoried
4. assess the distribution of the amphibian pathogen *Batrachochytrium dendrobatidis* (Bd) in Montana amphibians
5. initiate long-term intensive monitoring of population dynamics and vital rates of the Columbia Spotted Frog (*Rana luteiventris*) in three 12-digit HUC watersheds in western Montana
6. make all inventory information and predictive models readily available to natural resource managers via web applications

Throughout my dissertation I use common and scientific names in the 5th edition of Scientific and Standard English Names of the Amphibians and Reptiles of North America North of Mexico (Crother 2000, Crother et al. 2001, 2003). I have chosen to use these in preference to those in the 6th edition (Crother 2008) because they currently have more of a consensus among herpetologists and because changes in the 6th edition have been questioned by several authors as not adequately reflecting a consensus among herpetologists and unnecessary to reflect evolutionary history (e.g., Smith and Chiszar 2006, Hillis 2007, Wiens 2007) (Table 1.1).

In chapter 2, I summarize the site and watershed occupancy rates from surveys of 6,741 lentic sites within 429 randomly selected watersheds between 2000 and 2008. Surveys provided additional evidence for declines in Western Toad (*Bufo boreas*) and Northern Leopard Frog (*Rana pipiens*) populations; *R. pipiens* was only detected at 1 site in western Montana and *B. boreas* was not detected in portions of the eastern edge of its former range and was detected breeding in only 17% of watersheds and 2% of sites. However, classification trees for *B. boreas* showed indications that the species preferentially uses disturbed landscapes; breeding at 29% of permanent sites associated with recent forest fires and 6% of ephemeral sites associated with recent timber harvest. Seven of the 10 amphibian species and the Common Gartersnake (*Thamnophis sirtalis*) were detected at significantly fewer sites when fish were detected. The presence of emergent vegetation was positively associated with breeding or occupancy rates for all but one of the species examined and appeared to partially mitigate the effects of fish. Classification trees indicate that resource managers could enhance habitats for wetland herpetofauna by (1) creating new lentic sites on the landscape either directly or through protection or reestablishment of natural disturbances such as beaver, floods, and bison, (2) creating emergent vegetation at portions of existing sites that currently lack it via rotational fencing to temporarily exclude grazing, and (3) eliminating introduced fish populations.

In chapter 3, I used presence-only data in conjunction with pseudo-absences in program Maxent to model distribution and habitat suitability for 31 species of amphibians and reptiles in Montana to inform management and conservation efforts. My primary goals were to: (1) identify variables that limit species' distributions; (2) identify areas in need of field surveys; (3) create lists of predicted species within administrative boundaries at the regional (>10,000 km^2), landscape (township or 100 km^2), and large local habitat patch (>16 Ha) scales; and (4) identify marginal, suitable, and optimal habitat classes for species at various spatial scales. Models identified scale dependent responses to environmental variables, opportunities to extend the known ranges of species, areas that support potentially isolated populations in need of conservation efforts, areas that are critical for maintaining landscape connectivity, areas that may provide the best habitat for reintroduction of species that have declined, and areas where exotic and nonindigenous species are most likely to become established. When compared to predictions from the deductively based models produced by the Montana Gap Analysis Project, continuous Maxent models offered more realistic depictions of amphibian and reptile species distributions when survey data was available for a region and in most cases reduced predicted area while simultaneously increasing predictive accuracy. However, deductive models like those produced by GAP are still important for representing some species distributions in areas lacking survey effort.

In chapter 4, I evaluate the distribution of the chytrid fungus, (Bd), which is pathogenic to many amphibians and has been linked to declines and extinctions in a number of species around the globe. Tissue samples and swabs of ventral surfaces were analyzed for the presence of Bd using PCR primers for the internal transcribed spacer region of rDNA. Bd was detected in 218 samples taken at 68 sites between 1998 and 2008 for 6 of the 9 species tested; *A. tigrinum*, *B. boreas*, *B. woodhousii*, *P. maculata*, *R. luteiventris*, and *R. pipiens*. Bd was found in samples taken throughout the active season of the species, is widespread across Montana, and was found at a variety of elevations, habitats, and distances from, and intensities of, human activity. In light of its association with other amphibian declines, Bd should be regarded as an ongoing threat to Montana amphibians.

My dissertation demonstrates how national efforts such as the U.S. Geological Survey's Amphibian Research and Monitoring Initiative can expand research and monitoring efforts beyond national park and wildlife refuge boundaries to develop a research and monitoring program that has broader spatial inference on the status of lentic breeding amphibians and aquatic reptiles. Natural resources managers now have a state-wide assessment of watershed and site occupancy rates that can be used as a current measure of status and as a baseline for future evaluations within various geographic and administrative strata. In areas where watershed surveys have been performed, resource managers can use detections and habitat evaluations to make preliminary decisions regarding a variety of land management actions. A website at the Montana Natural Heritage Program allows resource managers to access these survey results, including digital photographs of sites surveyed, in the context of a variety of map layers. In watersheds that have not been surveyed, resource managers can use predicted distribution models in combination with classification trees to determine the likely suitability of the landscape and site, respectively, for individual species as well as evaluate potential habitat enhancements for a site or local set of sites. While results from the intensive monitoring of population dynamics and vital rates of *R. luteiventris* in three 12-digit HUC watersheds in western Montana are not summarized in my dissertation due to limited time and funding, these efforts are continuing and will eventually be summarized consistent with the original goal of the integrated multi-tiered monitoring strategy articulated above.

Table 1.1. Common and scientific names of amphibians and reptiles of Montana. Names in the 5th edition (Crother 2000, Crother et al. 2001, 2003) and 6th edition (Crother 2008) of Scientific and Standard English Names of the Amphibians and Reptiles of North America North of Mexico are shown on the left and right, respectively, with changes in the 6th edition in bold. Common and scientific names in the 5th edition are used throughout my dissertation because changes in the 6th edition are under debate (e.g., Smith and Chiszar 2006, Hillis 2007, Wiens 2007).

Scientific and Standard English Names 5th Edition		Scientific and Standard English Names 6th Edition	
Amphibians			
Long-toed Salamander	*Ambystoma macrodactylum*	Long-toed Salamander	*Ambystoma macrodactylum*
Tiger Salamander	*Ambystoma tigrinum*	**Barred Tiger Salamander**	***Ambystoma mavortium***
Idaho Giant Salamander	*Dicamptodon aterrimus*	Idaho Giant Salamander	*Dicamptodon aterrimus*
Coeur d'Alene Salamander	*Plethodon idahoensis*	Coeur d'Alene Salamander	*Plethodon idahoensis*
Rocky Mountain Tailed Frog	*Ascaphus montanus*	Rocky Mountain Tailed Frog	*Ascaphus montanus*
Plains Spadefoot	*Spea bombifrons*	Plains Spadefoot	*Spea bombifrons*
Western Toad	*Bufo boreas*	Western Toad	***Anaxyrus boreas***
Great Plains Toad	*Bufo cognatus*	Great Plains Toad	***Anaxyrus cognatus***
Woodhouse's Toad	*Bufo woodhousii*	Woodhouse's Toad	***Anaxyrus woodhousii***
Boreal Chorus Frog	*Pseudacris maculata*	Boreal Chorus Frog	*Pseudacris maculata*
Pacific Treefrog	*Pseudacris regilla*	**Northern Pacific Treefrog**	*Pseudacris regilla*
American Bullfrog	*Rana catesbeiana*	American Bullfrog	***Lithobates catesbeianus***
Northern Leopard Frog	*Rana pipiens*	Northern Leopard Frog	***Lithobates pipiens***
Columbia Spotted Frog	*Rana luteiventris*	Columbia Spotted Frog	*Rana luteiventris*
Reptiles			
Snapping Turtle	*Chelydra serpentina*	Snapping Turtle	*Chelydra serpentina*
Painted Turtle	*Chrysemys picta*	Painted Turtle	*Chrysemys picta*
Spiny Softshell	*Apalone spinifera*	Spiny Softshell	*Apalone spinifera*
Northern Alligator Lizard	*Elgaria coerulea*	Northern Alligator Lizard	*Elgaria coerulea*
Greater Short-horned Lizard	*Phrynosoma hernandesi*	Greater Short-horned Lizard	*Phrynosoma hernandesi*
Common Sagebrush Lizard	*Sceloporus graciosus*	Common Sagebrush Lizard	*Sceloporus graciosus*
Western Skink	*Eumeces skiltonianus*	Western Skink	***Plestiodon skiltonianus***
Rubber Boa	*Charina bottae*	**Northern Rubber Boa**	*Charina bottae*
Eastern Racer	*Coluber constrictor*	**North American Racer**	*Coluber constrictor*
Western Hog-nosed Snake	*Heterodon nasicus*	**Plains Hog-nosed Snake**	*Heterodon nasicus*
Smooth Greensnake	*Opheodrys vernalis*	Smooth Greensnake	*Opheodrys vernalis*
Milksnake	*Lampropeltis triangulum*	Milksnake	*Lampropeltis triangulum*
Gophersnake	*Pituophis catenifer*	Gophersnake	*Pituophis catenifer*
Terrestrial Gartersnake	*Thamnophis elegans*	Terrestrial Gartersnake	*Thamnophis elegans*
Plains Gartersnake	*Thamnophis radix*	Plains Gartersnake	*Thamnophis radix*
Common Gartersnake	*Thamnophis sirtalis*	Common Gartersnake	*Thamnophis sirtalis*
Prairie Rattlesnake	*Crotalus viridis*	Prairie Rattlesnake	*Crotalus viridis*

CHAPTER 2

STATUS OF LENTIC BREEDING AMPHIBIANS AND AQUATIC REPTILES IN MONTANA: A BASE-LEVEL ASSESSMENT UNDER THE U.S. GEOLOGICAL SURVEY'S AMPHIBIAN RESEARCH AND MONITORING INITIATIVE

Abstract

We developed a state-wide inventory and monitoring scheme for 10 lentic breeding amphibians and 4 aquatic reptiles in Montana and conducted surveys at 6,741 potential lentic sites within 429 randomly selected watersheds between 2000 and 2008. We used classification trees to examine patterns in rates resulting from major habitat features able to be affected by management actions. Watershed and site detection rates for breeding added additional evidence for declines in the Western Toad (*Bufo boreas*) and Northern Leopard Frog (*Rana pipiens*). *R. pipiens* was only detected at 1 site in western Montana. *B. boreas* was not detected in portions of the eastern edge of its range and was detected breeding in only 17% of watersheds and 2% of sites. However, *B. boreas* bred at 29% of permanent sites associated with recent forest fires and 6% of ephemeral sites associated with recent timber harvest. Seven of the 10 amphibian species and the Common Gartersnake (*Thamnophis sirtalis*) were detected at significantly fewer sites when fish were detected. The presence of emergent vegetation was positively associated with breeding or occupancy rates for all but one of the species examined and appeared to partially mitigate fish impacts. Resource managers could enhance habitats for wetland herpetofauna by (1) creating new lentic sites on the landscape either directly or through protection or reestablishment of natural disturbances such as beaver, floods, and bison, (2) creating emergent vegetation at portions of existing sites that currently lack it via rotational fencing to temporarily exclude grazing, and (3) eliminating introduced fish populations. This collaborative study demonstrates how national efforts such as the U.S. Geological Survey's Amphibian Research and Monitoring Initiative can expand research and monitoring efforts beyond national park and wildlife refuge boundaries to develop a research and monitoring program that has broader inference on the status of lentic breeding amphibians and aquatic reptiles.

Introduction

Responding to Amphibian Declines

At the First World Congress of Herpetology in 1989 herpetologists began to realize that amphibian declines were a global phenomenon (Blaustein and Wake 1990, Stebbins and Cohen 1995, Gibbons 2003). There is now consensus that declines have occurred and hundreds of species are on the brink of extinction (Houlahan et al. 2000, Wake 2003, Stuart et al. 2004, Lannoo 2005), but there was initial debate as to whether declines really had occurred or whether they were just part of the natural variability of populations (Pechmann et al. 1991, Blaustein 1994, Pechmann and Wilbur 1994). Documenting declines and their underlying causes was, and continues to be, hampered by a lack of regional, national, and international integration of broad-scale inventory data with more intensive local studies of natural history and population dynamics and experimental research on causal mechanisms; in part due to competing resources for metrics that balance scope of inference against spatial inference (Table 2.1, Figure 2.1).

The U.S. Geological Survey's Amphibian Research and Monitoring Initiative (ARMI) has articulated a multi-tiered program to address this issue in the United States (Hall and Langtimm 2001, Corn et al. 2005a-c, Muths et al. 2005). The program is organized as a hierarchical pyramid, consisting of: (1) base-level periodic regional assessments of distribution and status from probabilistic sampling schemes to determine site occupancy rates across a variety of public and private lands; (2) mid-level regular monitoring of site occupancy rates on non-random focal portions of Department of Interior lands (e.g., national parks and national wildlife refuges); and (3) apex-level intensive population studies to document demographic responses to relevant covariates and conduct experimental cause-effect research on putative causes of declines. Modeling efforts would then integrate information gathered under each of these levels. Of the three levels of this pyramid, the base-level regional assessments may offer the greatest challenge to implement because they are logistically difficult and necessarily involve multiple federal, state, tribal, non-governmental organizations, and private partners with potentially conflicting information needs, mandates, and fiscal limitations.

Characteristics of a Successful Assessment Program

These challenges to implementing an effective base-level regional assessment require investigators to keep characteristics of a successful assessment program clearly in mind at all stages of the project (Table 2.2, Thompson et al. 1998). Involvement of all relevant stakeholders is important to ensure that the project is adequately funded, that goals are clearly defined and limitations are clearly stated, that sampling units are meaningful, and that target populations to which inferences will be drawn will meet everyone's needs. Sampling units should be randomly selected to ensure proper inference to the target population and are ideally naturally defined uniform portions of the environment that contain the response variables of primary interest as well as meaningful covariates (e.g., standing water bodies, springs, seeps, stream reaches). Stratification of sampling units into strata and substrata based on bioregion and land ownership may be necessary to define smaller target populations and sampling frames where response variables are likely to have higher precision (Thompson et al. 1998). Investigators should also consider the potential degree of bias of estimates. Ideally a response variable will have both high precision and low bias. Stakeholders should define biologically meaningful effect sizes that will trigger management actions before the project begins, and as the project proceeds periodically evaluate these effect sizes in the context of the precision with which variables are able to be measured (Steidl et al. 1997, Thompson et al. 1998, Johnson 1999). The better coordinated (locally, regionally, and nationally) and more flexible an assessment program is, the more likely it is to be successful over the long run. Much of the flexibility required revolves around the program's goals and the ability to meet those goals in the face of periodic funding shortfalls. Given that periodic shortfalls in funding are likely, regional assessment program objectives are probably best kept to periodic assessments of status for a given target population based on a fresh random sample on each sampling occasion rather than an unreasonable commitment to monitor trends in a population of the same sampling units on an annual or other regular interval (Skalski 1990). This has the added benefit of continually adding to the number of sites for which there is some baseline information. The ideal situation is to simultaneously add new baseline information, assess the status of the target population, and evaluate trends in previously sampled sites through some type of rotational sampling scheme with replacement (Skalski 1990, Urquhart et al. 1998).

Assessment Goals for Montana

Where possible, we applied these characteristics to the problem of a state-wide base-level assessment of the status of lentic breeding amphibians and aquatic reptiles across Montana, where almost 60% of the amphibians and over 40% of the aquatic reptiles are state Species of Concern and where very few standardized baseline surveys had been undertaken (Maxell et al. 2003, Werner et al. 2004, Maxell et al. 2009, MNHP and MFWP 2009). In collaboration with federal, state, tribal, and private stakeholders we established the following mutual goals: (1) develop a common state-wide inventory and monitoring scheme for periodically assessing the status and distribution of amphibians and aquatic reptiles; (2) establish standard survey protocols and a state-wide database that can be used by all partners to assess changes in site and watershed rates of breeding (amphibians) and occupancy (aquatic reptiles) over time with reference to a variety of local and landscape variables; (3) conduct baseline surveys using watersheds as the basic sampling unit, focusing on watersheds dominated by public lands as an initial assessment goal; (4) evaluate survey methodology to assess the precision with which habitat variables are measured and the degree to which detection rates for various life history stages vary by habitat; and (5) make survey information easily available to biologists and resource managers so that it can be easily used for management decisions on individual sites as well as for planning at the level of administrative units.

In this paper we describe the state-wide inventory and monitoring scheme we have developed for Montana to assess distribution and status of lentic breeding amphibians and aquatic reptiles and present results for our surveys of watersheds on public lands, and discuss implications for future surveys and management actions. This effort complemented mid and apex-level efforts of ARMI in Glacier, Theodore Roosevelt, and Yellowstone National Parks and a number of U.S. Fish and Wildlife Refuges in the Northern Rocky Mountains (e.g., Adams et al. 2005, Brooks et al. 2005, Hossack et al. 2005, Muths et al. 2005, Guscio et al. 2008, Hossack and Corn 2007, 2008).

Methods

Sampling Scheme

We stratified Montana into 11 geographic strata (Figure 2.1) defined by level 3 ecoregions (Nesser et al. 1997) and 8-digit hydrologic unit code (HUC) watersheds (Seaber et al. 1984). We then sub-stratified geographic strata by 12-digit watersheds that differed by land ownership makeup (> 40% public, >40% tribal, and <40% public or tribal). This created 28 different target populations and sampling frames for which results of site occupancy rate surveys can be more meaningfully inferred and interpreted into management actions (Table 2.3). Within each of these target populations and sampling frames we randomly selected 12-digit HUC watersheds in numbers approximately proportional to the total area and number of watersheds in the sampling frame (Table 2.3, Figure 2.1). This established a random selection of up to one third of the watersheds within the >40% public and >40% tribal land ownership strata and approximately 10% of the watersheds within the <40% public or tribal land ownership stratum.

Field Inventory

Prior to field work, we mapped all potential standing water bodies identified on the most recent 7.5-minute (1:24,000 scale) U.S. Geological Survey quadrangle maps or aerial photographs within each of the randomly selected watersheds; aerial photographs were not used to detect potential lentic sites in the drier non-mountainous portions of eastern Montana where standing waters capable of supporting amphibian reproduction are more reliably mapped. When we did not detect lentic sites in a watershed on topographic maps or aerial photographs, we briefly ground truthed the watershed by driving roads or hiking major trails to examine areas of low topographic relief or backwaters of streams that might provide lentic breeding habitat. Unless sites were inaccessible (e.g., some required crossing unsafe terrain or private lands where access was not granted), field crews navigated to all mapped potential lentic sites on public lands in each watershed and used timed visual encounter and dipnet surveys in all portions of the water bodies that were less than 50 cm in depth (Heyer et al. 1994, Olson et al. 1997). In addition, field crews surveyed standing water bodies that were encountered incidentally while navigating to mapped sites and areas within a 200 m radius of mapped sites were searched for additional water bodies potentially lumped under a single map feature by cartographers. Digital photographs of each site were taken and species and habitat information was recorded on a standardized datasheet (Appendix A, Table 2.4a).

Field surveys were conducted each year between 2000 and 2008 between the time when oviposition had been completed at most sites and when large numbers of metamorphosing animals were observed. Surveys were conducted typically between late May and early July in eastern Montana and between late May and the end of August in western Montana where crews could survey at progressively higher elevations as the summer proceeded. Presence/non-detection data may underestimate occupancy if sites are surveyed only once because probability of detection is often less than 1 (MacKenzie et al. 2002, 2006). Unfortunately, multiple visits to sites were not feasible in this study due to the goal of assessing species' distributions in regions that, for the most part, lacked any baseline information. However, sites in a small number of nonrandomly selected watersheds were surveyed by multiple crew members as part of training or evaluation sessions (Appendix B) and multiple site surveys were performed within Glacier and Yellowstone National Parks for mid-level monitoring efforts (Corn et al. 2005b, Muths et al. 2005, Hossack and Corn 2007). These sessions were used to examine detection probabilities for some species to identify potential biases resulting from surveys in different habitats and to inform future monitoring efforts (Appendix B). Presence/non-detection data may also be biased when species are misidentified (MacKenzie et al. 2002, 2006). We believe that misidentification of species was not a problem for the vast majority of our observations because eggs, larvae, juveniles, and adults of most species are easily distinguishable (see keys in Maxell et al. 2003 and Werner et al. 2004). However, larval Great Plains Toads (*Bufo cognatus*) and Woodhouse's Toads (*Bufo woodhousii*) are very similar in appearance to one another; small differences in arch of the tail fin, ventral coloration of tail musculature, and widths of labial tooth rows are the only distinguishing features and their use in species identification can be problematic in smaller larvae (Maxell et al. 2003, Werner et al. 2004). We, therefore, tried to rely on additional information when making a species designation for bufonid larvae on the plains of eastern Montana. This included the presence of adults, eggs, or newly metamorphosed juveniles, major habitat type (*B. woodhousii* was more broadly distributed on the landscape and used a wider variety of habitat types while *B. cognatus* was limited to breeding in ephemeral sites in grasslands or riparian flood plains). At sites where species identification was uncertain in the field, we preserved several larvae in 10% neutral buffered formalin and examined their tail fin arch, basal tail musculature pigmentation, and relative width of labial tooth rows under a dissecting microscope (Maxell et al. 2003, Werner et al. 2004).

To prevent the spread of fungal and viral pathogens, field equipment and clothing that contacted water or mud was washed and then decontaminated with 10% bleach between each watershed surveyed (Johnson et al. 2003, Johnson and Speare 2003, Johnson and Speare 2005). Voucher specimens were collected for each amphibian or reptile species encountered in each watershed and we intend to deposit the majority of these in the U.S. National Museum at the Smithsonian.

<u>Data Analysis</u>

We only used lentic sites that contained water at the time of the survey and were deemed capable of holding water long enough to support amphibian reproduction in analyses of the percent of watersheds or sites occupied (reptiles) or with breeding (amphibians). Similarly, we only used watersheds containing lentic sites in these analyses. We calculated confidence intervals for watershed rates using a standard error formula with a finite population correction factor given that the number of public dominated watersheds in each target population was known (Krebs 1999). However, for confidence intervals of site occupancy rates we used a standard error formula without a finite population correction factor given that we do not know the total number of lentic sites in each target population. To allow all stake holders to readily access survey information we created a dynamic internet mapping application housed at the Montana Natural Heritage Program that uses ASP.net, C#, and AJAX coding to display tabular data from surveys stored in a SQL Server database and mapped using ArcIMS.

We used classification trees in both S-PLUS 6.2 and CART 6.0 to group independent environmental variables associated with lentic habitats and identify patterns associated with detection of breeding by amphibians or the presence of aquatic reptiles (Table 2.4a-b). Classification and regression tree (CART) models are a non-parametric method that partitions a dataset recursively into binary subsets that are increasingly homogeneous. The result is a tree model that classifies different combinations of environmental variables as more or less suitable as habitat while simultaneously providing a dichotomous classification tree. CART has advantages over traditional linear models because independent variables can appear on multiple branches of a tree (e.g., emergent vegetation at ephemeral water bodies versus emergent vegetation at permanent water bodies with or without fish) and, therefore, may more realistically represent ecological rules determining a species presence (Breiman et al. 1984, Iverson and Prasad 1998, De'ath and Fabricius 2000, Urban 2002, Cutler et al. 2007).

Our main criteria for selection of independent variables for our CART analysis were that they were biologically relevant to the species and could be easily interpretable by resource managers in a variety of ecological settings across Montana. Thus, environmental variables were chosen from those available to have representation across the species' range and address major habitat features that could be affected by management actions (Table 2.4a). Continuous independent variables were converted to categorical present or absent values to simplify resulting trees. Analysis was limited to lentic sites randomly selected for survey within the known range of each species in Montana. Models were constructed using a splitting rule that minimizes deviance (node heterogeneity) in S-Plus 6.2 and using the Gini index in CART 6.0 (Breiman et al. 1984; De'ath and Fabricius 2000; Urban 2002); terminal node size was limited to a minimum of 10 observations. This yielded classification trees where the vertical depth of each split is proportional to the amount of variation explained by a given independent variable or group of variables. We used a 10-fold cross validation of the data in CART 6.0 to estimate the relative prediction error associated with trees of different sizes and typically pruned trees back to the number of terminal nodes resulting in the minimum relative error rate (Breiman et al. 1984; De'ath and Fabricius 2000). However, trees were sometimes pruned to smaller, more parsimonious, sizes than those resulting in the minimum relative error rate when the difference in error rate was minor (much less than 1 SE of the minimum tree) and the resulting nodes at the minimum relative error rate could not be meaningfully interpreted. We used S-Plus to create trees for display.

Results

Watershed and Site Characteristics

Of the 455 12-digit HUC watersheds randomly selected for survey in the 10 target populations containing greater than or equal to 40% public land ownership, 429 (94%) have been surveyed through the 2008 field season (Table 2.3, Figure 2.2). Of the 26 watersheds in this target population that have not been surveyed, 13 have been the focus of intensive assessments by ARMI in Glacier National Park (e.g., Corn et al. 2005b, Muths et al. 2005, Hossack and Corn 2007) and are not worth additional survey. We still intend to survey the remaining 13 (1 each in strata 1, 7, and 11 and 5 each in strata 2 and 10), but naïve site occupancy rate estimates in these strata are unlikely to be altered significantly as a result.

Of the 429 watersheds surveyed, we failed to detect any lentic sites on topographic maps, aerial photos, or with ground truthing in 22 and although 26 others had lentic sites detected on private or tribal lands within the watershed, we did not find any water bodies on public lands. Thus, 381 (84%) of the randomly selected watersheds had some form of a lentic site on public lands. Within these watersheds there was an average of 25.0 (SD = 28.7, range = 1 – 211) lentic sites detected on both public and private lands. However, on the public lands we surveyed, there was an average of 18.5 lentic sites (SD = 23.4, range = 1 – 211) per watershed.

We surveyed 6,741 potential lentic sites and 1,398 of these were evaluated as not capable of supporting amphibian reproduction or worth future survey because they were misidentified as a potential lentic site (e.g., a shadow on an aerial photograph or dry well on a 1: 24,000 quadrangle map), were lotic, or were lentic, but judged incapable of holding water long enough to support amphibian reproduction. Of the sites surveyed, 5,640 (84%) were detected on quadrangle maps, 679 (10%) were detected on aerial photographs, and 422 (6%) were detected incidentally while in the field.

Seventy-nine percent (5,343) of sites were evaluated as capable of supporting amphibian reproduction and worth future survey. Sites identified on aerial photographs had the highest error rate of a priori site identification for supporting amphibian reproduction; 64% (438) as compared to 80% (4,502) of those detected on quadrangle maps. Ninety-five percent (403) of sites detected incidentally while in the field were identified as capable of supporting reproduction.

Of the 5,343 sites deemed capable of supporting amphibian reproduction and worth future survey, 405 (8%) were dry at the time of survey, 2,726 (51%), were evaluated as ephemeral, and 2,206 (41%) were evaluated as permanent. Watersheds that had any sites evaluated as capable of supporting amphibian reproduction on public lands averaged 1.1 (SD = 2.5) dry sites, 7.6 (SD = 10.3) ephemeral sites with water, 4.3 (SD = 5.5) permanent sites with emergent vegetation, and 1.9 (SD = 7.4) permanent sites without emergent vegetation.

Most sites in strata 1-7 were on lands administered by the U.S. Forest Service, while in strata 10-12, the majority of sites are administered by the Bureau of Land Management (Figure 2.3a). Lands administered by the state of Montana typically made up a small portion of the sites surveyed, but they composed a relatively higher percentage of the sites in eastern Montana. Other administrators of sites surveyed include Plum Creek Timber Company and The Nature Conservancy in western Montana and the U.S. Fish and Wildlife Service in both western and eastern Montana. Elevations of sites surveyed are mostly normally distributed across strata 1-6 with the exceptions of modal classes from 1,400-1,600 m in stratum 3, 1,000-1,200 m in stratum 4, and 1,800-2,200 m in stratum 5 (Figure 2.3b). Sites were predominantly in the 2,400-2,800 m elevation classes in stratum 7, while sites in strata 10-12 were mostly below 1,000 m.

Lentic sites in strata 1-7 were most often formed in natural depressions, but beaver created 10-15% of sites in most strata and in stratum 3 they created 42% of sites (Figures 2.3c-d). In strata 10-12 more than 70% of sites surveyed were human made reservoirs for livestock watering with fewer than 20% of sites of a depressional nature and beaver created less than 2% of sites. Lentic oxbow or spring habitats created by hydraulic action composed 7-20% of sites across all strata. Lake, pond, and wetland habitats in stratum 7 and wetland habitats in stratum 10 were predominately created through the action of glaciers (Alt and Hyndman 1995). The relative proportion of ephemeral sites, permanent sites with emergent vegetation, and permanent sites without emergent vegetation was fairly consistent across all strata with ephemeral sites dominating other categories (Figure 2.3e).

Fish were detected in 13-25% of lentic sites in strata 1-7 and 6-9% of sites in strata 10-12 (Figure 2.3f). Fish were detected in approximately the same proportion of permanent sites with emergent vegetation as permanent sites without emergent vegetation in most strata, but a higher proportion of permanent sites with emergent vegetation had fish in strata 5 and 11 (Figure 2.3f). Fish were detected in a small proportion of sites classified as ephemeral because these sites were ephemerally connected to permanent waters supporting fish (Figure 2.3f). Structural impacts to shorelines, shoreline vegetation, and water quality from grazing were absent or light throughout most of the sites in strata 1-7 (Figure 2.3g). One exception to this was some of the nonforested areas in strata 6 south of Butte where a number of sites had received heavy structural or water quality impacts from cattle grazing. Heavy structural or water quality impacts from grazing were also present in strata 10-12, with an especially high percentage of sites impacted in strata 11 and 12 (68 and 51%, respectively). Waters were dammed (mostly earthen dams of drainages) or diverted (often capping and piping of springs to watering tanks) in 1-16% of sites in strata 1-7 and in 72-84% of sites in strata 10-12 (Figure 2.3h).

Species Distributions

Surveys and associated incidental observations during the course of this project resulted in 11,423 species observation records across Montana. The 9,499 amphibian observation records and 1,924 reptile observation records are respectively equivalent to 52 and 25% of all observations that have been gathered for these vertebrate classes in Montana since the time of the Lewis and Clark expedition (Maxell et al. 2003). The surveys and associated incidental observations filled in many gaps in the known distribution of these species and resulted in numerous extensions of the known range and elevation limits of species (Figures 2.4a-n). Most notably, these include: a 28 km southwest extension into the upper reaches of the West Fork of the Bitterroot River, a 32 km southeast extension to the Big Hole Divide, and a 20 km southeast extension in the Elkhorn Mountains for *A. macrodactylum* (Figure 2.4a); a 28 km extension into the upper reaches of the Blacktail Deer Creek drainage in southeastern Beaverhead County for *A. tigrinum* (Figure 2.4b); 53 observations on 11 different streams within 10 km of the first record of Idaho Giant Salamander (*Dicamptodon aterrimus*) in Montana in Mineral County; an extension 45 km south and 60 km east to southern Ravalli County for *P. idahoensis* (Maxell 2002a); a 40 km westward extension into north central Stillwater County and a 35 km northwest extension into Golden Valley County for *B. cognatus* (Figure 2.4e); a 16 km southward extension in Ravalli County for Pacific Treefrog (*Pseudacris regilla*) (Figure 2.4h); a 185 km eastward extension within Montana and a 56 km extension northward from Wyoming to the headwaters of Rosebud Creek in southeastern Bighorn County for Rubber Boa (*Charina bottae*); a 64 km eastward extension into west central Custer County for Terrestrial Gartersnake (*Thamnophis elegans*) (Figure 2.4l); and a 55 km westward extension into Stillwater County for Plains Gartersnake (*Thamnophis radix*) (Figure 2.4m).

Known elevation limits were extended during the course of these surveys up to or beyond those reported in Maxell et al. (2003) and Werner et al. (2004) for *A. macrodactylum* (2,774 m) near Homer Youngs Peak in Beaverhead County, *D. atterimus* (1,737 m) near the headwaters of the West Fork of Big Creek in Mineral County, *P. idahoensis* (1,585 m) along Little Rock Creek just south of Lake Como in Ravalli County, *B. cognatus*

(1,300 m) southeast of Wallop Butte in southwest Powder River County, *P. maculata* (2,841 m) just east southeast of Black Butte in southern Madison County, and *P. regilla* (1,810 m) between Lydia Mountain and Sutton Mountain in northeast Lincoln County.

The distribution of elevations used for breeding by Montana's amphibians and aquatic reptiles has received little attention in the past because most previous survey efforts have been focused on valley bottoms. Our recent efforts have been focused on public lands with a broader distribution of elevations. Given that temperatures and, therefore activity periods for ectotherms, are dependent on elevation a summary of breeding and occupancy rates by elevation class may prove useful to managers (Figures 2.4a-n). *A. macrodactylum* were detected breeding in 22-49% of sites below elevations of 2,600 m, but breeding rates dropped off quickly above this elevation (Figure 2.4a). *A. tigrinum* had a bimodal breeding distribution with peaks of higher detection between 1,000 and 1,600 m corresponding to sites in eastern Montana and between 2,000 and 2,400 m corresponding to sites in southwest Montana (Figure 2.4b). Plains Spadefoot (*Spea bombifrons*), *B. cognatus*, *B. woodhousii* and *R. pipiens* were all limited to breeding at sites below 1,228 m corresponding to non mountainous areas across eastern Montana, but *B. cognatus* was restricted to sites below 942 m, essentially corresponding to major river valleys and adjacent uplands (Figures 2.4c, 2.4e, 2.4f, and 2.4j). *B. boreas* had low (less than or equal to 2%) breeding rates across a wide range of elevations, but breeding rates were significantly higher (4-10%) at mid elevation mountainous areas between 1,600 and 2,000 m (Figure 2.4d). *P. maculata* was documented breeding in alpine meadows of southwestern Montana at elevations of just above 2,800 m. However, breeding rates were much lower (less than 11% of sites) at elevations above 1,400 m as compared to elevations below 1,200 m (greater than 42% of sites) (Figure 2.4g). Breeding rates were highest for *P. regilla* at elevations below 800 m and steadily decreased up to their maximum documented elevation of 1,810 m (Figure 2.4h). *R. luteiventris* bred at 17-32% of sites below 2,800 m, corresponding roughly to tree line across their range, with breeding rates dropping off drastically above this (Figure 2.4j). Painted Turtle (*Chrysemys picta*) was detected at 12% of sites below 800 m with rates of detection dropping off steadily above this to 2% of sites below 1,400 m (Figure 2.4k). *T. elegans* had a bimodal distribution of site occupancy with relatively high rates of detection (greater than 5%) below 800 m and rates of 6-9% between 1,400 and 2,200 m (Figure 2.4l). *T. radix* was detected at 10-19% of sites below 1,157 m, but was not detected at higher elevations (Figure 2.4m). Common Gartersnake (*Thamnophis sirtalis*) was detected at 1-10% of sites below 2,600 m, with a modal elevation class at 1,200-1,400 m (Figure 2.4n).

Naïve Site Breeding and Occupancy Rates and Habitat Associations

A. macrodactylum was detected breeding in 66-77% of watersheds west of the Continental Divide (strata 1, 2, and 4) and 24-58% of watersheds east of the Continental Divide (strata 3, 5, and 6) within its known range (Table 2.5). Similarly, it was detected breeding in 31-44% of lentic sites in strata west of the Continental Divide as compared to only 10-18% of sites east of the Continental Divide. With the exception of strata 3 and 5 which had broad confidence intervals for both watershed and site breeding rate estimates due to small sample sizes, all confidence intervals were reasonably precise. Overall, *A. macrodactylum* was detected breeding in 34% of the 2,119 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5a). The species bred at a lower percentage of sites where fish were detected (15%) than where fish were not detected (38%). When fish were present, the percentage of sites with breeding detected was over 4 times greater when emergent vegetation was present (18%) compared to when emergent vegetation was not present (4%). *A. macrodactylum* still favored breeding in sites with emergent vegetation even when fish were not present (42% versus 24%).

A. tigrinum was not detected breeding at any lentic sites in the randomly selected watersheds within their limited range west of the Continental Divide (Table 2.5, Figure 2.4b). However, it was detected breeding in 30% and 50% of watersheds and 3% and 20% of sites in strata 7 and 6, respectively, in southwest Montana. In strata 10, 11, and 12 on the Great Plains, *A. tigrinum* watershed breeding rates were consistently high at 73-79%, but site breeding rates ranged from 11-40%. Confidence intervals were reasonably precise for both watershed and site estimates. Overall, *A. tigrinum* was detected breeding in 14% of the 2,536 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5b). The species was twice as likely to be detected breeding at sites with emergent vegetation (16%) than sites without emergent vegetation (8%) and *A. tigrinum* was not detected breeding at any sites when fish were present and emergent vegetation was absent. In the presence of emergent vegetation, the percentage of sites where *A. tigrinum* was detected breeding was 143% higher when no fish were detected (17%) as compared to when fish were detected (7%). When fish were not detected, the percentage of sites where *A. tigrinum* was detected breeding was almost halved when no emergent vegetation was present (9%) as compared to when it was present (17%).

S. bombifrons was detected breeding in 14-24% of watersheds and only 2-4% of sites across strata 10, 11, and 12. Confidence intervals were reasonably precise for site breeding rates, but were fairly broad for watershed breeding rates (Table 2.5). Only a simple two forked classification tree was supported by the *S. bombifrons* breeding detection data (Table 2.4, Figure 2.5c). The species was detected breeding in only 3% of the 1,578 lentic sites that were surveyed across all strata within its known range (Table 2.4, Figure 2.5c). However, they were 5 times more likely to breed in ephemeral sites (5% of sites) than permanent sites (1% of sites).

B. boreas was not detected breeding within the western margins of strata 10, 11, and 12 that lie within its known geographic range and was also not detected breeding in randomly selected watersheds within strata 7 which is well within its known range (Table 2.5, Figure 2.4d, Maxell et al. 2003). Outside of these strata, watershed and site breeding rates did not appear to differ across the Continental Divide for *B. boreas* with watershed breeding rates ranging from 11-50% and site breeding rates ranging from 1-5% across strata 1-6. Confidence intervals for watersheds were broad with the interval for stratum 5 overlapping zero. Most of the confidence intervals for site breeding rates were also broad, given the small point estimates, and intervals for strata 2, 3 and 5 all overlapped zero. Overall, *B. boreas* was detected breeding at 2% of the 3,357 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5d). The species was more likely to breed at sites with emergent vegetation than those without emergent vegetation (3% versus 0.5%) and they were more likely to breed in permanent than ephemeral waters (4% versus 2%). At ephemeral sites *B. boreas* was more likely to breed at sites that had recently been logged (6% versus 1%) and at permanent sites they were more likely to breed at sites that had recently been burned (29% versus 4%).

B. cognatus was detected breeding in 3-22% of watersheds, but only 1-3% of sites across strata 10, 11, and 12. Confidence intervals were imprecise for both watershed and site breeding rates and overlapped zero in several cases (Table 2.5). Overall, *B. cognatus* was detected breeding at 2% of the 1,552 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5e). The species was not detected breeding at any sites with fish and at sites without fish it bred at 2% of sites with emergent vegetation as compared to only 1% of sites without emergent vegetation. At fishless sites with emergent vegetation it bred at 2% of sites that were ephemeral as compared to only 1% of sites that were permanent.

B. woodhousii was detected breeding in 42% of watersheds in stratum 12, but only 19% of watersheds in strata 10 and 11 which are more on the northern margin of its range (Table 2.5, Figure 2.4f, Stebbins 2003). Similarly, it was detected at 16 and 26% of sites in strata 12 and 11, respectively, but only at 2% of sites in strata 10 on the margin of its range. Confidence intervals for both watershed and site breeding rates were reasonably precise. Overall, *B. woodhousii* was detected breeding at 8% of the 1,543 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5f). The species was over twice as likely to breed in fishless sites (9%) than sites with fish (4%). At fishless sites they bred in a higher percentage of sites with emergent vegetation (9%) than without emergent vegetation (6%).

P. maculata was only detected breeding at 40% of sites within 1 watershed at the margin of its range in strata 3 and was not detected in strata 5 and 7 where it has only been sparsely reported in a few valley bottoms dominated by private lands that were not surveyed (Table 2.5, Figure 2.4g, Maxell et al. 2003). The species was detected breeding in 36% of watersheds and 13% of sites in stratum 6 which is on the margins of its range and elevation limits (Figure 2.4g, Stebbins 2003). In the non-glaciated areas of strata 11 and 12 breeding was detected in 61 and 59% of watersheds, respectively, and in the glaciated stratum 10, breeding was detected in 97% of watersheds (Table 2.5, Alt and Hyndman 1995). Breeding was detected in 35-49% of sites in strata 10, 11, and 12. With the exception of stratum 3, confidence intervals were reasonably precise for all estimates of watershed and site breeding rates. Overall, *P. maculata* was detected breeding at 26% of the 2,667 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5g). The species was detected breeding at very few sites without emergent vegetation (3%) as compared to sites with emergent vegetation (35%). At sites with emergent vegetation, it was detected at a higher percentage of fishless sites (36%) as compared to sites with fish (24%) and it seemed to only marginally favor permanent sites (39%) over ephemeral sites (34%) at fishless sites with emergent vegetation.

P. regilla was detected breeding in 43% of the watersheds and 18% of the lentic sites surveyed within its core range in stratum 1 (Table 2.5, Figure 2.4h). However, in strata 2 and 4, at the margins of its range, it was detected breeding in only 4% of watersheds and 0.2-5% of sites. Confidence intervals for watershed breeding rate estimates in strata 2 and 4 overlapped zero, but confidence intervals for the stratum 1 watershed rate estimate and all site rate estimates were reasonably precise. Overall, *P. regilla* was detected breeding at 4% of the 1,370 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5h). The species was detected breeding in very few sites without emergent vegetation (0.3%) as compared to sites with emergent vegetation (5%). At sites with emergent vegetation, it was detected breeding at a higher percentage of fishless sites (6%) as compared to sites with fish (1%).

R. luteiventris was not detected within any of the watersheds within its potential range in stratum 11 and was only detected in one watershed (33%) within its range in stratum 12 and 2 watersheds (50%) within its range in stratum 3 (Table 2.5, Figure 2.4j). Thus, confidence intervals for watershed and site breeding rate estimates in both strata 3 and 12 are very broad. In all other strata, watershed breeding rate estimates varied from 50-77%

and site breeding rate estimates varied from 13-45% with reasonably precise confidence intervals for all. Overall, *R. luteiventris* was detected breeding at 29% of the 2,781 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5j). The species was detected breeding in very few sites without emergent vegetation (2%) as compared to sites with emergent vegetation (43%). At sites with emergent vegetation, it was detected breeding at a higher percentage of permanent sites (52%) as compared to ephemeral sites (36%). At permanent sites with emergent vegetation it was detected breeding at a marginally higher percentage of fishless sites (54%) as compared to sites with fish (46%).

R. pipiens was only detected at one randomly selected lentic site in western Montana and was detected breeding in only 25% of watersheds and 6% of sites in strata 11 between the Missouri and Yellowstone Rivers. However, the species was detected breeding in 62% and 82% of watersheds, and 14% and 18% of sites, in strata 10 and 12, respectively. Confidence intervals were reasonably precise for all watershed and site breeding rate estimates in strata 10, 11, and 12 (Table 2.5, Figure 2.4i). Overall, *R. pipiens* was detected breeding at 14% of the 1,435 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5i). We detected breeding at a rate 3 times higher in permanent sites (25%) as compared to ephemeral sites (8%). At permanent sites *R. pipiens* was detected breeding in only sites with emergent vegetation (28%) and in these sites it was detected breeding at a higher percentage of sites when fish were present (45%) as compared to fishless sites (25%). Similarly at ephemeral sites the species was detected breeding at a higher percentage of sites that were connected to waters with fish (44%) as compared to ephemeral sites not connected to waters with fish (7%).

C. picta was not detected in any of the randomly selected watersheds in strata 5, 6, and 7 and was only detected in 2-25% of watersheds and 0.1-2% of sites in strata 1-4 in western Montana and stratum 11 in eastern Montana where confidence intervals were too imprecise to make estimates valuable for future comparisons (Table 2.5, Figure 2.4k). In strata 10 and 12 *C. picta* was detected in 53-55% of watersheds and 9% of sites, with reasonably precise confidence intervals for all estimates. Overall, *C. picta* was detected at 3% of the 4,961 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5k). The species was less likely to be detected at ephemeral sites (1%) than permanent sites (5%). At permanent sites, they were more often detected at sites with emergent vegetation (6%) than sites without emergent vegetation (2%).

T. elegans was detected in 19-42% of watersheds and 2-8% of sites in strata 1-7 in western Montana. With the exception of stratum 4 where confidence intervals overlapped zero, confidence intervals were reasonably precise (Table 2.5, Figure 2.4l). *T. elegans* was not detected in stratum 10 in eastern Montana and was only detected in 4% and 6% of watersheds and 1% and 0.4% of sites in strata 11 and 12, respectively. Confidence intervals for watershed and site occupancy rates overlapped zero in both strata 11 and 12 and would be of little use for future comparison. Overall, *T. elegans* was detected at 3% of the 4,182 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5l). The species was twice as likely to be detected at sites where juvenile or adult amphibians were detected (6%) than sites where juvenile or adult amphibians were not detected (3%). At sites where juvenile or adult amphibians were detected, *T. elegans* was detected at a higher percentage of sites where fish were detected (14%) than where fish were not detected (5%). At sites where juvenile or adult amphibians were not detected, *T. elegans* was detected at a higher percentage of sites with emergent vegetation (4%) as compared to sites without emergent vegetation (1%) and at these sites, the species was detected at a higher percentage of sites that are permanent (5%) than ephemeral (3%).

T. radix was detected in 42-86% of watersheds and 11-18% of sites in strata 10, 11, and 12 and all confidence intervals were reasonably precise (Table 2.5, Figure 2.4m). Overall, the species was detected at 16% of the 1,522 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5m). It was more than twice as likely to be detected at permanent sites (24%) than ephemeral sites (11%). At permanent sites it was more likely to be detected at sites where juvenile or adult amphibians were detected (31%) than sites where juvenile or adult amphibians were not detected (21%) and at the sites were juvenile or adult amphibians were not detected, it was twice as likely to be detected at sites with emergent vegetation (22%) than sites without emergent vegetation (11%). At ephemeral sites *T. radix* was more likely to be detected at sites where fish were detected (26%) than sites where fish were not detected (11%) and at the sites were fish were not detected, it was more likely to be detected at sites where larval amphibians were detected (13%) than where larval amphibians were not detected (8%).

T. sirtalis was not detected in any of the randomly selected watersheds in strata 5, 7, 10, or 11 (Table 2.5, Figure 2.4n). In the mountainous portion of western Montana in strata 1-4 and 6, detection rates in watersheds ranged from 8-32% and detection rates at sites ranged from 1-8%. On the plains of southeast Montana in stratum 12, the species was detected in 6% of watersheds and 1% of sites. Confidence intervals overlapped zero in strata 3, 6, and 12 and will be of little use for future status comparisons. Overall, *T. sirtalis* was detected at 3% of the 3,993 sites that were surveyed across all strata within the species' known range (Table 2.4, Figure 2.5n). It was three times more likely to be detected at sites where juvenile or adult amphibians were detected (6%) than sites where juvenile or adult amphibians were not detected (2%). At sites where juvenile and adult amphibians were detected, *T. sirtalis* was most likely to be detected at sites where neither larval amphibians or fish were detected and emergent vegetation was present (15%) (Figure 2.5n). At sites were juvenile or adult amphibians were not detected, *T. sirtalis* was most likely to be detected at sites with emergent vegetation, where larval amphibians were detected, and where waters were ephemeral (5%).

Discussion

Watersheds as Sampling Units

Similar to many areas in the western United States outside of National Parks and other protected areas, surveys assessing the status of amphibians and aquatic reptiles in a systematic manner across the vast majority of Montana were rare prior to this effort (Corn 1994, Maxell et al. 2003; see Werner et al. 1998 for a rare exception in Montana).

Data available prior to these surveys were mostly limited to positive findings (i.e., negative data were not published or recorded in a central database) and non-random so that inferences could not be properly drawn to a variety of administrative boundaries. Furthermore, existing data failed to consider animals detected at individual water bodies in the context of how the spatial structure and composition of the surrounding landscape might provide critical seasonal resources or govern source-sink population dynamics (Pulliam 1988, Dunning et al. 1992). Periodic assessments of the status of species, and the habitats on which they depend, to assess changes in status over time is essential for government agencies and nongovernmental organizations alike in the fulfillment of their mandates to ensure the continued persistence of healthy amphibian and reptile populations (Koch and Peterson 2005).

Our cooperative assessment attempted to bring characteristics of a successful assessment program (Table 2.2) to bear on this problem. One of our first hurdles was to define a biologically meaningful sampling unit. A meaningful sampling unit is important to integrating different levels of a multi-tiered strategy for assessment of, and research on, lentic breeding amphibians (Corn et al. 2005a, b, Figure 2.1). We believe local watersheds are an ideal sampling unit not only because they encompass networks of habitat patches and local breeding populations that may function as metapopulations (e.g., Gill 1978, Berven and Grudzien 1990, Sinsch 1992, Sjögren-Gulve 1994), but because they encompass natural disturbance regimes such as flooding and beaver which create new habitat patches (Sousa 1984, Pickett and White 1985, Johnston and Naiman 1990, Lind et al. 1996, Wright et al. 2002). Watersheds are also commonly used as management units by federal and state agencies and tribal governments so often encompass anthropogenic disturbance regimes as well. The boundaries of watersheds may be particularly useful in areas of high topographic relief where they are likely to act as barriers to dispersal (e.g., Funk et al. 2005). The U.S. Geological Survey has defined an integrated series of watersheds for much of the United States (Seaber et al. 1984) and the smallest watershed unit defined, a 12-digit HUC watershed, represents a relatively uniform naturally bounded portion of the environment. Because these watersheds have been defined for large areas of the western U.S. they can be used to define target populations and sampling frames. Furthermore, because they are available as GIS layers, stratification of sampling by bioregion, major hydrologic unit, and degree of public ownership can be easily accomplished to define smaller, more meaningful, target populations and sampling frames to make surveying and application of results more straightforward while increasing the precision of response variables (Peterson et al. 2005).

We defined 28 target populations for Montana using 12-digit HUC watersheds as the basic sampling unit and have nearly completed surveys in the 10 target populations dominated by public lands (Table 2.3, Figure 2.2). Overall, we feel that simultaneous assessment of site and watershed breeding and occupancy rates was a success and offers great advantages over single site assessments because it is more informative to managers at the scale at which they plan and undertake management actions, allows for assessment of the status of occupancy or breeding rates at different scales, and allows for integration of broad scale assessments with research on local population dynamics and landscape processes. However, because we limited surveys to only public lands within these watersheds and private lands fragment public lands in many watersheds, our surveys were effectively carried out on partial watersheds in many cases. While fiscal and logistical constraints precluded us from carrying out surveys on adjacent private lands within these watersheds in this initial effort, we suggest this be undertaken where landowner permissions are granted under future assessments because of the advantages of having baseline information for entire watersheds as noted above.

Need for Multiple Site Surveys

Our second major suggestion for improving future assessments is to survey individual lentic sites on multiple occasions to estimate true occupancy rates by correcting for incomplete detectability (MacKenzie et al. 2002, 2006). Multiple visits to sites were not feasible in this study due to budgetary and logistical constraints associated with the large number of sites selected to meet the objective of assessing species' distributions in regions that, for the most part, lacked any baseline information. Multiple surveys conducted while training field crews and as evaluation sessions to determine the precision with which habitat and species abundances were recorded (Appendix B) were relatively limited and estimates of probability are suspect. To date, detection probability estimates for various species and survey efforts in Glacier and Yellowstone National Parks have ranged from 0.51 to 0.94 and models that best fit the data have supported either constant detection probabilities across habitat types or detection probabilities that are dependent on the amount of emergent vegetation (Corn et al. 2005b, Muths et al. 2005, Hossack and Corn 2007). However, multiple site surveys conducted in Glacier and Yellowstone National Parks in 2006 resulted in detection probabilities for *A. macrodactylum*, *B. boreas*, *P. maculata*, and *R. luteiventris* that ranged from 0.84-0.93 and true estimates of occupancy rates differed very little from naïve estimates (Corn et al. 2008). Given these relatively high detection probabilities, small differences between naïve and true occupancy rate estimates, and some evidence that detection probabilities are only significantly reduced in complex habitats such as beaver dam complexes (pers. obs.), biases of naïve occupancy estimates reported here are probably not large in magnitude. However, we have no way of assessing the degree to which they are biased low. We intend to use our naïve occupancy estimates in conjunction with detection probability estimates from ARMI surveys in Glacier and Yellowstone National Parks in the simulation module of Program PRESENCE (MacKenzie and Royle 2005, MacKenzie et al. 2006) to plan future sample sizes and numbers of surveys per site needed to estimate occupancy rates with a high degree of precision.

<u>Species Status</u>

Classification tree models have been used in a broad variety of ecological and non-ecological applications involving complex categorical and continuous data, nonlinear relationships, and higher order interactions to yield results that are easily interpretable by both researchers and non-experts (Breiman et al. 1984, Iverson and Prasad 1998, De'ath and Fabricius 2000, Urban 2002, Cutler et al. 2007). Our classification tree analyses helped place naïve site breeding and occupancy rates for various geographic strata in the context of major habitat characteristics capable of being effected through management actions (Tables 2.5-2.6, Figures 2.5a-n). Below we discuss the status of individual species relative to naïve occupancy estimates in the various sampling strata and for the overall classification tree analysis.

A. macrodactylum is widespread at the watershed scale and occupies a high percentage of sites west of the Continental Divide, but the species occupies a smaller percentage of watersheds and lentic sites east of the Continental Divide (Table 2.5, Figure 2.4a). This likely results from a combination of reduced precipitation, forest cover, and forest litter east of the Continental Divide due to the rain shadow formed by adiabatic cooling of Pacific air masses. This interpretation of the drivers of regional patterns in occupancy is supported by observations at the scale of a local forest stand in western Montana where areas subject to timber harvest had less ground cover, higher soil temperatures and 75% fewer salamanders than control plots (McGraw 1997). Confidence intervals for the species in strata 3 and 5 east of the Continental Divide are too broad to be very useful for future comparisons of watershed or site occupancy rates as a result of small sample sizes and managers may want to consider boosting sample sizes in these regions to allow meaningful comparisons in the future (Table 2.5). The negative impacts of the presence of fish on site breeding rates in *A. macrodactylum* is dramatic with a 153% decrease at sites with fish regardless of other habitat characteristics, a 133% decrease when comparing sites with emergent vegetation, and a 500% decrease when comparing sites without emergent vegetation cover (Figure 2.5a). These findings agree with a number of studies showing strong negative impacts of fish across the species' range (Tyler 1996, Tyler et al. 1998, Funk and Dunlap 1999, Monello and Wright 1999, 2001).

A. tigrinum was not detected breeding at any of the randomly selected sites in northwest Montana (Table 2.5, Figure 2.4b). However, populations isolated on the Tobacco Plains between Rexford and Fortine are regularly reported and surveys on private lands between Eureka and the port of Roosville will likely identify additional populations. Watershed and site breeding rates were lower in strata 7 of southwestern Montana likely as a result of the high elevations dominating this region (Table 2.5, Figure 2.4b). Elsewhere the species bred in a fairly high percentage of watersheds (50-79%), but not always a high percentage of sites (11-40%). Confidence intervals in all strata throughout the species' main range are all precise enough to allow for comparisons of watershed or site occupancy rates (Table 2.5). The negative effects of the presence of fish on site breeding rates in *A. tigrinum* are dramatic. The species was never detected with fish when emergent vegetation was absent and suffered a 143% decrease at sites with fish when comparing sites with emergent vegetation (Figure 2.5b). These findings are consistent with a number of studies reporting almost complete exclusion of *A. tigrinum* from waters where predatory fishes have been introduced (Carpenter 1953, USFWS 1964-1982; Collins et al. 1988, Corn et al. 1997).

S. bombifrons had fairly consistent watershed (14-24%) and site (2-4%) breeding detection rates across Montana's eastern plains (Table 2.5). We believe these naïve estimates are essentially the correct magnitude and indicate the species is widespread, but limited in numbers across much of its range on public lands in eastern Montana. Nocturnal calling surveys from roads after spring rain events suggest that *S. bombifrons* is more common than these estimates in or near riparian areas which are dominated by private lands that were not the focus of our lentic site survey efforts. The majority of locality records we gathered for this species came from these roadside nocturnal calling surveys which were often in or near riparian areas (evident in distribution patterns in Figure 2.4c). We most commonly detected *S. bombifrons* on sandy soils in or near riparian areas, but also detected them on other soil types great distances from riparian areas. Only a simple two forked classification tree was supported by the *S. bombifrons* breeding detection data which showed a 400% increase in site breeding rates for this species at ephemeral sites as compared to permanent sites (Figure 2.5c). The associations we noted with ephemeral breeding sites and sandy soils is consistent with other reports of habitat use by this species on the Great Plains (Klassen 1998, Lauzon and Balagus 1998) and fits its life history which is highly adapted to rapid egg and larval development (Bragg 1937, 1940, 1964, 1966, Pfenning 1990). Given that many riparian areas are the focus of agricultural activities we encourage research on the status of habitats used by *S. bombifrons* for breeding on private lands.

B. boreas was not detected within the western margins of strata 10, 11, and 12 or within any of the randomly selected watersheds within strata 7 within the species' historic range (Table 2.5, Figure 2.4d). In all of our survey work throughout the Highwood, Beartooth, and Absaroka Mountain ranges, we detected only a single road killed adult on 15 August 2005, 8.1 km northwest of Cooke City in the Beartooth Mountains of Park County (Figure 2.4d). The only other recent observation from this region was an *B. boreas* adult reported 17.8 km west northwest of Cooke City in the Beartooth Mountains of Park County on 28 July 2001. There are only 4 other observations of *B. boreas* from the Absaroka or Beartooth Mountains and they are all over 41 years old (Montana Natural Heritage Program Point Observation Database, Helena, MT). The only reported observations of *B. boreas* in the Highwood Mountains that we are aware of are voucher specimens of an adult, tadpoles, and eggs collected 24 June 1962 on upper Highwood Creek near the pass to Arrow Creek (Montana State University, Bozeman catalogue numbers 4431, 4436, 4440, and 4444; species identification confirmed by BAM). Without more historic observations it is difficult to assess what the lack of current observations in these regions means, but it is suggestive of declines or near extirpations on the southeast and northeast edges of its historic range. *B. boreas* is still widespread throughout the rest of its historic range in western Montana, but breeding seems to be clustered in groups with distances between groups being greater (often >20 km) in the south and east and becoming progressively smaller toward the north and west (Figure 2.4d). Overall watershed and site breeding rates (17% and 2%, respectively) across strata 1-6 approach the lowest of any lentic breeding amphibian we assessed with 95% confidence intervals for site occupancy overlapping 0 in 3 of 6 strata (Table 2.5, Figure 2.4d). Historic information on the distribution and status of *B. boreas* is limited mostly to anecdotal qualitative accounts that note them as "common throughout western Montana" (Black and Timken 1976), "a very common and obvious amphibian that is found almost everywhere" in southwest Montana (Timken, no date, but from the 1970s), "the most widespread amphibian in the Jackson Hole region" (Carpenter 1953), "commonly found throughout" Yellowstone National Park "in humid situations" (Turner 1951), "very common" around the lakes and streams of the Front Range just east of Glacier National Park (Coues and Yarrow 1878), "common" west of the Continental Divide and in the central mountain ranges (Black 1970), have as an extensive a range as *A. macrodactylum* (Brunson and Demaree 1951, see Table 2.5 and Figure 2.4a for current distribution and status of *A. macrodactylum*), “abundant in the forested areas especially near lakes and ponds” of Flathead County (Franz 1971), and “common” in the Bitterroot Valley (Rodgers and Jellison 1942). Cooper (1869) reported *B. boreas* as “not very common” along the Clark Fork and Bitterroot Rivers. Breeding populations appear to be relatively stable over the last decade, but populations have likely declined from historic levels. Small breeding populations (<10 females) at most current breeding sites and the large distances between local groups of breeding sites are cause for concern, especially in light of declines in other portions of the species’ range (Muths and Nanjappa 2005).

B. cognatus larvae are very similar in appearance to larval *B. woodhousii*, so there was a potential for errors in identification. We attempted to minimize these errors, but a study that examines species’ determinations made from external morphological features as compared to assignment via genetic methods would be useful. The naïve estimates for watershed (3-22%) and site (1-3%) breeding detection rates for *B. cognatus* (Table 2.5) are consistent with our experience with the species being more broadly distributed in grasslands on the glaciated plains and along major riparian floodplains (Figure 2.4e). As with *S. bombifrons*, our nocturnal calling survey experiences make us believe that *B. cognatus* is more common than these estimates in or near riparian areas which are dominated by private lands that were not the focus of our lentic site survey efforts. The majority of locality records we gathered for this species came from these roadside nocturnal calling surveys which were often in or near riparian areas (evident in distribution patterns in Figure 2.4e). We most commonly detected *B. cognatus* on sandier soils in or near major riparian floodplain areas, but also detected them on other soil types in upland grasslands quite distant from riparian areas. The classification tree shows preferred breeding habitat to be fishless, ephemeral, and with emergent vegetation (Figure 2.5e). This is consistent with the temporary rain-filled buffalo wallow sites that Bragg (1937) described as *B. cognatus* breeding habitat in high prairie areas around Norman, Oklahoma. Confidence intervals for site breeding rate estimates overlapped zero in two of the three strata examined and watershed breeding rate estimates overlapped zero in one stratum as well (Table 2.5). It may therefore be more appropriate to monitor the status of this species through nocturnal calling surveys given our past successes under appropriate weather conditions and its widespread use of private lands.

B. woodhousii is rarer and more patchily distributed on the glaciated plains north of the Missouri River than in the area between the Missouri and Yellowstone Rivers or in the Tongue and Powder River basins south of the Yellowstone River where they were commonly encountered on the landscape (Table 2.5, Figures 2.2 and 2.4f). Confidence intervals for both watershed and site breeding rates in all 3 of the strata in the species' range are precise enough to allow for meaningful future comparisons (Table 2.5). The classification tree (Figure 2.5f) certainly supports our field experience that the species breeds in a wide variety of lentic sites across the landscape, including sites with fish, even though breeding rates at sites with fish are 125% lower than those at sites without fish (Kruse and Stone 1984).

P. maculata is widespread across the plains of eastern Montana and occupies a high percentage of watersheds and sites across this region (Table 2.5, Figure 2.4g). The species is also distributed across southwest Montana with another center of distribution in southern Beaverhead, Madison, and Park Counties. However, in this area, it occupies a smaller percentage of watersheds and sites. There are a few scattered records between the species' centers of distribution, but they do not seem to fully connect these two regions. With the exception of strata 3 where the species was only detected in a single watershed, confidence intervals for both watershed and site breeding rates are precise enough to allow for meaningful future comparisons (Table 2.5). Site breeding rates for the species were most dependent on the presence of emergent vegetation, but fish did have a small negative impact (Figure 2.5g). Corn et al. (1997) noted that occurrence of *P. maculata* was unrelated to the presence or absence of introduced or native trout in Rocky Mountain National Park because the species' tadpoles, as herbivorous filter feeders, inhabit heavily vegetated shallow waters where they are unlikely to be exposed to predation risk. We feel that these foraging strategies are also responsible for the small impact we observed for fish at sites with emergent vegetation in Montana.

P. regilla bred in 43% of watersheds and 18% of sites within its core range in stratum 1 and confidence intervals for both watershed and site breeding rates are precise enough to allow for meaningful future comparisons (Table 2.5, Figure 2.4h). Outside of this core range, *P. regilla* occupied a very small percentage of watersheds (4%) and sites (<5%) and many confidence intervals overlapped zero. In these strata we have likely documented most of the breeding populations present on the landscape and these relatively isolated populations may benefit from special management consideration. Similar to *P. maculata*, *P. regilla* also showed a very strong preference for breeding sites with emergent vegetation (Figure 2.5h). However, *P. regilla* showed a 500% decrease in site breeding rate when fish were present. This finding is consistent with other studies in the Pacific Northwest and the Sierra Nevada showing strong negative impacts of fish on *P. regilla* (Licht 1969, Yoon 1977, Bradford 1989, Monello and Wright 1999).

R. pipiens declines west of the Continental Divide in western Montana were reviewed by Werner (2003). The causes of declines are unknown, but only 3 populations are known to be extant west of the Continental Divide (Montana Natural Heritage Program Point Observation Database, Helena, MT). *R. pipiens* has been reported in the intermountain valleys of the upper Missouri watershed in southwestern Montana since the mid 1990s (Maxell et al. 2003), but there have been no formal surveys in this region which is dominated by private lands. *R. pipiens* is still widespread and relatively common in eastern Montana and confidence intervals for both watershed and site breeding rates are suitable for comparison with future assessments (Table 2.5, Figure 2.4i). In this region, watershed and site breeding rates appear to be positively correlated with availability of surface waters; higher rates in stratum 10 the glaciated plains north of the Missouri River, and stratum 12 south of the Yellowstone River that receives relatively higher amounts of rainfall and the lowest rates in stratum 11 which is dominated by a relatively dry upland landscape with low rainfall. This interpretation is supported by the classification tree which indicates a 213% increase in site breeding rates in permanent over ephemeral waters (Figure 2.5i). The classification tree also indicates that breeding is dependent on emergent vegetation, but that the species breeds at sites with and without fish.

R. luteiventris is widespread and relatively common across most of its range in the mountainous portion of western Montana with high watershed and site breeding rates throughout most of this area (Table 2.5, Figures 2.2 and 2.4i). However, we failed to detect the species in our randomly selected watersheds in the Highwood Mountains. We detected the species in adjacent sections of the Little Belt Mountains approximately 40 km to the south, but the last time they were reported in the Highwood Mountains was in 1970 (Black 1970). Systematic surveys of all lentic sites in this mountain range are probably warranted given its heavy use for livestock grazing (pers. obs.), relative isolation, and declines that have been reported for the species in arid range lands in the southern portions of its range as a result of livestock grazing and watering and changes in hydrology (Cuellar 1994, Reaser 1996, 1997, 2000, Hovingh 1997). The species was also not detected breeding in any of the high elevation watersheds on the eastern edge of the Beartooth Plateau in stratum 11. This is likely a result of the short growing season on these northerly aspect watersheds which are mostly above tree line and lack emergent vegetation in the majority of the water bodies they contain. Short growing season and relatively little emergent vegetation in the large number of sites above tree line is also the likely driver behind the relatively low site breeding rates observed in stratum 7 (Table 2.5). As is indicated by the classification tree for breeding site characteristics, the species is dependent on emergent vegetation for oviposition and larval rearing (Figure 2.5j). The classification tree also indicates that *R. luteiventris* was equally likely to breed at, or in satellite pools adjacent to, sites with and without fish (46% with fish versus 54% without fish) as long as emergent vegetation was also present. The majority of sites that we sampled with fish contained salmonid fishes and the co-occurrence pattern we observed is consistent with other studies that have reported co-occurrence with salmonid fishes (Marnell 1997, Bull and Hayes 2000, Pilliod et al. 2002, Dunham et al. 2004). However, these studies also warned of the potential for longer term negative effects of fish and fish stocking that are too subtle to be observed by simple patterns of co-occurrence from single visit surveys or even short-term monitoring. In contrast, Monello and Wright (1999) found that *R. luteiventris* never bred at sites with warm water fishes in the Palouse region of northern Idaho.

C. picta is restricted to lower elevation permanent water sites that, more often than not, have emergent vegetation (Figures 2.4k and 2.5k). This complicates interpretation of watershed and site occupancy rates within regions where watersheds include higher elevations. This is mostly a problem for western Montana where the primary low elevation habitats of the species are dominated by private lands that have not yet been systematically evaluated. Still, the relatively low watershed and site occupancy rates rates for strata 1-7 (Table 2.5) are an accurate representation of the species distribution and status on public lands within this region. The species generally had high watershed and site occupancy rates on the plains of eastern Montana. However, similar to the pattern observed for *R. pipiens*, another species dependent on permanent waters, *C. picta* occupied a much lower percentage of watersheds and sites in stratum 11 which is dominated by dry upland landscapes with low rainfall. We note that *C. picta* is much more broadly distributed on arid landscapes across Montana than it was likely to have been prior to the widespread development of reservoirs and stock ponds with permanent surface waters starting in the later half of the 19^{th} century (Figures 2.3c, d, e, g, and h).

T. elegans, *T. radix*, and *T. sirtalis* all had relatively low site occupancy rates as measured by our single visit surveys and with the exception of *T. radix* many of the 95% confidence intervals for site occupancy approached or overlapped zero (Table 2.5). We have not yet analyzed detection probability estimates for any of these species from multiple surveys of sites, but a mark recapture study we conducted in the Bitterroot Mountains for *T. elegans* and *T. sirtalis* resulted in very low capture probabilities (unpublished data) and we believe this is indicative of low detectability for all three of these species during site surveys. For these species, and others with low detection probabilities or low occupancy rates (e.g. *B. boreas*), it may be more appropriate to use watershed breeding or occupancy rates as a metric for tracking status. The three gartersnake species had site occupancy rates that were 50-200% higher when juvenile or adult amphibians or amphibian larvae were present as compared to when they were absent (Figures 2.5l-n). Of the three species, it is clear that *T. radix* is more common at both the watershed and site scales within its range and there is evidence that *T. sirtalis* is more patchily distributed than *T. elegans* (i.e., relatively lower watershed occupancy rates at the same overall site occupancy rate) across their largely overlapping geographic ranges (Table 2.5). This patchy distribution for *T. sirtalis* may be due to its greater dependency on amphibians as prey items as compared to *T. elegans* and *T. radix* which make use of a wide variety of non-amphibian prey items (Gregory 1978, Kephart and Arnold 1982, Rossman et al. 1996) and not only preferentially associate with amphibians, but also preferentially associate with sites with fish (Figures 2.5l-m).

<u>Implications for Lentic Site Management</u>

Several patterns are evident in our survey data that have important implications for management of lentic wetlands. There has been a large overall shift in the distribution of habitats available in prairie landscapes away from naturally created ephemeral depressional wetlands disturbed intensely, but infrequently, by bison (Knapp et al. 1999) toward human made reservoirs with permanent hydroperiods disturbed intensely and frequently by cattle (Figures 2.3c-h). Waters were dammed or diverted at 72-84% of potential lentic sites surveyed in strata 10-12 in eastern Montana (Figures 2.2 and 2.3h). This has been coupled with a dramatic loss of ephemeral wetland habitats in this region through draining of wetlands for agriculture (Dahl 1990, Samson and Knopf 1996). Loss of ephemeral breeding sites such as bison wallows and shallow glacial pothole wetlands has likely negatively impacted local populations of species like *S. bombifrons* and *B. cognatus* that are highly adapted to, and dependent on, ephemeral habitats (Figures 2.5c and 2.5e) (Bragg 1937, 1940, 1966, Gerlanc and Kaufman 2003). Other species like *A. tigrinum*, *B. woodhousii* and *P. maculata* have likely been negatively impacted by the loss of ephemeral wetlands in some areas, but have been able to more than make up for those losses by widespread use of reservoirs and stock ponds (Figures 2.5b, 2.5f, and 2.5g). Species like *R. pipiens*, *C. picta*, and *T. radix* that are either dependent on permanent hydroperiods, or the prey items at these sites (Figures 2.5i, 2.5k, and 2.5m), have likely benefited a great deal by this shift in hydroperiods and are almost certainly more widespread on prairie landscapes as a result. These patterns and other studies on hydroperiods (e.g., Skelly 1996) highlight the need for managers to consider the availability of the full range of hydroperiods on local portions of the landscape when creating new lentic sites. In many cases, the full spectrum of hydroperiods can be created at a single site by constructing portions of the site to maintain deep water habitats and portions of the site with shallow sloped benches as an attached or semi-attached arm (particularly useful if fish predators are present in the main site).

Dam building by beaver (*Castor canadensis*) and the hydraulic actions of water are the two natural disturbances that are still creating large amounts of potential lentic breeding habitat across Montana. Beaver created wetlands have been shown to be used by a variety of amphibian and reptile species in eastern North America (Russell et al. 1999, Metts et al. 2001). Beaver created 2-42% of lentic sites and lentic oxbow or spring habitats created by hydraulic action composed 7-20% of sites within the various strata (Figures 2.3c-d). Where fur trapping has eliminated beaver from some mountain ranges in the central portion of the state many water bodies are approaching their final successional stages as they fill in with sediments (pers. obs.). In southwestern Montana, lentic sites engineered by beaver increased the number of *R. luteiventris* breeding populations and improved connectivity among breeding sites at the scale of a 12-digit HUC watershed (Amish 2006). Management actions that protect and restore natural hydrological regimes, flood pulses, and beaver populations are likely to benefit amphibian populations (Zedler 2000, Smith and Tyers 2008, Moore 2006).

Emergent vegetation plays a valuable role in providing places of attachment for egg masses, cover from predators, and foraging habitat for larvae and post metamorphic animals (Werner et al. 2004; Maxell et al. 2009). Thirteen of the 14 species we examined showed increases in naïve estimates of site occupancy ranging from 100-2,050% when emergent vegetation was present as compared to when it was absent (Figures 2.5a-n and 2.6). One land use that stands out as a potential threat to wetland vegetation structure and water quality at lentic sites across western North America is grazing (Fleischner 1994, Belsky et al. 1999). We noted heavy structural impacts to wetland vegetation or water quality from livestock grazing at a notable percentage of sites in strata 6, 10, 11, and 12, but an especially high percentage (51-68%) of sites were heavily impacted in strata 11 and 12 on the non-glaciated plains of central and southeastern Montana (Figures 2.2 and 2.3g). Wetlands and riparian areas in these landscapes were historically disturbed intensely, but infrequently, by bison and likely benefit from some level of disturbance through nutrient inputs and maintenance of open waters (Knapp et al. 1999). However, disturbances of wetland vegetation by cattle are now both intense and frequent and may be leading to continued deterioration of these habitats (Belsky et al. 1999). We encourage evaluation of the effects of grazing on type and amount of emergent vegetation present at wetlands and its value to the population dynamics of individual species along the lines of Adams et al. (2009). In the meantime, our data indicate that resource managers may benefit a number of amphibian and aquatic reptile species by establishing emergent vegetation at sites were it currently is not present (Figures 2.5a-n and 2.6). Thousands of recently created reservoirs across eastern Montana currently lack emergent vegetation and managers could facilitate the establishment of emergent vegetation at these sites through selected plantings and temporarily fencing out cattle from portions a site.

Strong negative responses to the presence of fish were evident in classification trees for 7 of the 14 species we examined, with *B. cognatus* and *T. sirtalis* never being found with fish and *A. tigrinum*, *P. maculata* and *P. regilla* only being found with fish when emergent vegetation was present (Figures 2.5a-n and 2.6). Of the 7 species negatively affected by fish, there was evidence that emergent vegetation at least partially mitigated these impacts for 2 species. Occupancy rates of *A. macrodactylum* at sites with fish were 350% higher when emergent vegetation was present while occupancy rates at sites without fish were only 150% higher when emergent vegetation was present (Figure 2.5a). *A. tigrinum* was not present at sites with fish when emergent vegetation was absent, but was present at 7% of sites with fish when emergent vegetation was present (Figure 2.5b). These dominant negative impacts of fish on 6 of the 10 amphibian species assessed have implications for the three gartersnake species we examined because all three of these species had site occupancy rates that were 50-200% higher when juvenile or adult amphibians or amphibian larvae were present as compared to when they were absent (Figures 2.5l-n). All of these patterns are consistent with the findings of other research on the interaction of fish, amphibians, and their gartersnake predators in the western United States where nonindigenous fishes have been widely introduced (Bradford 1989, Liss and Larson 1991, Bahls 1992, Bradford et al. 1993, Kiesecker and Blaustein 1998, Tyler et al. 1998, Fuller et al. 1999, Knapp et al. 2001, Matthews et al. 2001, 2002). The implications for resource managers are clear. Fisheries management actions have the potential to negatively impact native amphibian populations and their predators and should be carefully considered at both the scale of individual sites as well as the scale of local watersheds which are often composed of complementary seasonal habitat patches (Pilliod et al. 2002). Restrictions in stocking or eradication of nonnative populations (Knapp and Matthews 1998) should be considered as management actions in some watersheds in order to balance recreational opportunities with other ecosystem values (Dunham et al. 2004).

Some amphibian species benefit from forest disturbance by wildfire (see reviews by Pilliod et al. 2003, Bury 2004). In Glacier National Park, Montana, *B. boreas* adults have been documented using severely burned forested habitats much more extensively than partially burned or unburned forested habitats (Guscio et al. 2008). Furthermore, adults bred in a large number of forested wetlands that were unoccupied prior to the fire (Hossack and Corn 2007). *B. boreas* have been documented responding positively to disturbances varying from wetland creation to catastrophic volcanic eruptions that have reset entire landscapes to early successional states (Crisafulli et al. 2005, Pearl and Bowerman 2006). Our inclusion of recent timber harvest and fire disturbance variables in the CART analysis for *B. boreas* adds additional evidence for the species as a disturbance associate (Figure 2.5d). *B. boreas* had a breeding occupancy rate at recently burned permanent lentic sites that was 625% higher than it was at unburned permanent lentic sites. Similarly, it had a breeding occupancy rate at ephemeral sites where timber had recently been harvested that was 500% greater than at ephemeral sites where forest canopy was left intact. Better knowledge of the effects of wildfire and other factors potentially affecting the population dynamics of *B. boreas* is needed (Bury 2004), particularly in light of the low rates of watershed and site breeding occupancies (Table 2.5, Figure 2.4d). Future increases in the size of forest disturbances due to beetle kill (Gibson et al. 2008) and continued climate warming and subsequent increases in forest fires (Westerling et al. 2006) or salvage logging have the potential to benefit *B. boreas* if the breeding occupancy patterns we observed hold. We encourage resurveys of lentic sites with baseline information from this survey if they are disturbed by future fires and encourage managers to consider *B. boreas* in management plans as yet another species that utilizes patches of disturbed forest.

Availability of Survey Information

One of the most critical features of any inventory, monitoring, or research program is to ensure that resource managers can easily access the information gathered to inform on-the-ground management decisions in a timely manner. To provide all partners ready access to survey information we created a dynamic internet based mapping application housed at the Montana Natural Heritage Program. Agency partners can use a password protected version of this website to display locations of over 9,900 surveys of standing water bodies, locations for over 26,000 observation records, and digital photographs of all sites surveyed in the context of a variety of map layers, including high resolution aerial photographs, land ownership, public land survey sections, and topographic maps. For the public version of the application, we protect exact locations of species by limiting display of distribution information to grid cells covering one quarter of a degree of latitude by one quarter of a degree of longitude. In addition to displaying information, the application also allows users to add their own observations to the master database to engage all partners and the general public in adding baseline and monitoring data. Records added in this manner remain marked as pending final acceptance until they are evaluated by a species expert to ensure they are within the known range of the species, known date ranges of the species' active season, and are associated with appropriate habitats. In its first 18 months, this site has received over 5,000 hours of use by agency biologists and natural resource managers and over 6,100 additional hours of use by the general public.

Summary Recommendations

This study is a good example of how the needs of a diverse group of stakeholders can be met under the common goal of assessing the status of lentic breeding amphibians and aquatic reptiles at a state-wide level and in a manner complementary to national efforts such as the U.S. Geological Survey's ARMI program. Major recommendations from this study follow below.

We encourage state-wide or regional inventory and monitoring schemes for lentic breeding amphibians and aquatic reptiles in other areas through diverse partnerships of stakeholders. Baseline inventories that can be used for assessing future changes in distribution and status are still lacking for large portions of the United States and the methods used in this study are applicable to most of western North America. In tandem with ARMI, these efforts could result in a research and monitoring program that truly has national inference.

Where topography potentially creates dispersal barriers between watersheds (e.g., Funk et al. 2005) we advocate a watershed-based approach to sampling with an attempt to survey all water bodies in each randomly selected watershed regardless of ownership. This approach has the advantages of: (a) assessing occupancy rates at both the site and watershed scales; (b) assessing occupancy rates for local clusters of populations that likely act as metapopulations; (c) explicitly mapping the distribution of breeding, foraging, and aquatic overwintering habitats on the landscape to inform management decisions in the watersheds surveyed; (d) providing insights into processes that create, destroy, and modify water bodies species are dependent on; (e) reducing costs of fieldwork by accessing clusters of local sites instead of random sites spread out across a landscape; and (f) allowing results of studies with narrower spatial scopes of inference, but greater inference strength (e.g., annual monitoring and demographic and experimental studies at the mid- and apex-level of the ARMI program), to be more easily integrated.

This first probabilistic state-wide baseline inventory on public lands in Montana has greatly expanded our understanding of the general distributions and site occupancy rates of species. However, the next periodic assessment should take a more solid statistical approach by surveying all sites on multiple occasions, or with multiple surveyors on a single occasion, to estimate true occupancy rates in the context of various local and landscape covariates while accounting for changes in detectability across different habitats (MacKenzie et al. 2002, 2006). This will increase expense and reduce the number of sites that can be surveyed to provide managers with baseline information to make on-the-ground planning decisions, but it is much more defensible for assessing changes in status as assessed by site breeding or occupancy rates. An approach that both randomly selects watersheds from those previously surveyed and those not previously surveyed is likely to provide the strongest inference about changes in status (Skalski 1990, Urquhart et al. 1998).

It required 8 years to complete surveys for the target population on public lands due to the number of sites and watersheds involved and the degree of interest by resource managers in surveys of non-randomly selected watersheds to inform on-the-ground management decisions regarding issues such as fish stocking and restoration. An analysis of confidence intervals and biologically meaningful effect sizes for changes in site occupancy rates is needed to determine sample sizes required for future monitoring efforts (MacKenzie et al. 2006). In addition, surveys are still needed for private and tribal target populations, which probably contain the primary habitats for a number of species that are more rarely encountered on public lands (e.g., *S. bombifrons*, *B. cognatus*, and *C. picta*). Finally, between periodic assessments of lentic and watershed occupancy rates, we encourage establishment of annual systematic roadside calling surveys similar to those promoted by the North American Amphibian Monitoring Program (Weir and Mossman 2005) to better understand the status of species in riparian areas dominated by private lands.

This study supports earlier findings (Werner 2003) that *R. pipiens* populations in western Montana have been nearly extirpated. Reintroduction efforts have already been initiated by the Confederated Salish and Kootenai Tribes on the Flathead Indian Reservation (Janene Lichtenberg, Confederated Salish and Kootenai Tribal Wildlife Management Program, pers. comm.). Given that populations in eastern Montana are still widespread, relatively abundant, and appear to be reasonable source populations from a genetic standpoint (Hoffman and Blouin 2004), we recommend additional reintroduction efforts across the historic range of *R. pipiens* in western Montana. Furthermore, potentially isolated populations in the intermountain valleys of the upper Missouri River should be documented and conservation measures implemented where appropriate (Chapter 3).

This study adds to earlier evidence (Werner et al. 1998, Maxell 2000, Maxell et al. 2003) that *B. boreas* has undergone declines in western Montana. The failure to detect any reproductive effort in the Beartooth and Absaroka Mountain Ranges is of particular concern. We recommend annual monitoring of all known *B. boreas* breeding populations in western Montana to better document their annual reproductive effort and more quickly detect any future declines or causative agents of declines. Our findings corroborate other studies that have found *B. boreas* to be associated with disturbances to forest canopies. We recommend additional research on the correlation between *B. boreas* breeding sites and forest disturbances using the baseline data from this study in conjunction with newly burned or harvested forest stands.

Patterns in occupancy rates of the species we assessed relative to site origin, habitat type, and habitat threats identified a number of recurring recommendations for resource managers. These include:

(1) Promote natural disturbance regimes such as beaver, floods, and bison wallows in order to maintain networks of lentic habitat patches at the watershed scale.
(2) When creating new lentic sites, consider the full spectrum of hydroperiods species are dependent on and incorporate all hydroperiods into the design of the site. Consider site characteristics in a landscape context to maintain or improve connectivity of seasonal habitats at local patch and landscape scales.
(3) Consider fisheries management actions at both the scale of individual sites as well as the scale of local watersheds which are often composed of complementary seasonal

habitat patches for amphibians. Restrictions in stocking or eradication of nonnative populations should be considered as management actions in some watersheds in order to balance recreational opportunities with other ecosystem values.

(4) Protect emergent vegetation which provides important breeding, hiding, and foraging habitats for aquatic herpetofauna and potentially mitigates some of the impacts of fish populations. Temporary fencing on impaired sites or newer stock ponds can be used to establish or reestablish emergent vegetation and still allow for periodic disturbance through rotation of the excluded area.

Finally, one of the most critical features of any inventory, monitoring, or research program is to ensure that resource managers can easily access the resulting information to inform on-the-ground management decisions in a timely manner. We encourage resource managers to use our dynamic internet based mapping application housed at the Montana Natural Heritage Program to access survey information and report observations and we recommend creation of similar applications in other regions.

Table 2.1. Tradeoffs between strength of inference and spatial extent of inference in methods for monitoring wildlife populations (see Figure 2.1).

Monitoring Approach	Cost Per Species	Strength of Inference	Spatial Inference
Area of Suitable Habitat	Low	Low	High
Disturbance Regimes	Low	Low	High
Apparent and True Patch Occupancy Rates	Low	Low	High
Estimates of True Numbers or Densities	High	High	Low
Indices of Relative Abundance or Density	Low	Low	High
Survival Rates	High	High	Low

Table 2.2. Characteristics of a successful base-level assessment program

- Well coordinated (locally, regionally, nationally)
- Involvement of all stakeholders throughout program
- Program goals clearly defined and limitations clearly stated
- Well defined and biologically meaningful sampling unit
- Well defined target population(s)
- Sampling frames stratified by bioregion and land ownership
- Estimates of response variable have low bias and high precision
- High statistical power of detecting change in response variable
- Biologically meaningful management thresholds and responses determined *a priori*
- Flexible and inexpensive
- Response variables informative about status at any one time and trends over time
- Meaningful even in the face of periodic funding losses and shortfalls

Table 2.3. Summary of watershed sampling and surveys for monitoring lentic breeding amphibians in Montana through 2008.

Geographic Strata	Ownership Substrata	No. Watersheds	No. Watersheds Randomly Selected	Percent of Watersheds Randomly Selected	No. Watersheds Surveyed through 2008 (Random / Non Random)	Percent Random Surveyed through 2008
1	≥ 40% Public	188	61	32	60 / 5	98
	≥ 40% Tribal	36	13	36	0 / 0	0
	≤ 40% Public	76	7	9	0 / 8	0
	Total	300	81	27	60 / 13	74
2	≥ 40% Public	137	54	39	39 / 6	72
	≥ 40% Tribal	2	1	50	0 / 0	0
	≤ 40% Public	1	1	100	0 / 0	0
	Total	140	56	40	39 / 6	70
3	≥ 40% Public	62	10	16	7 / 0	70
	≥ 40% Tribal	9	3	33	0 / 0	0
	≤ 40% Public	7	3	43	0 / 0	0
	Total	78	16	21	7 / 0	44
4	≥ 40% Public	268	78	29	78 / 4	100
	≥ 40% Tribal	0	-	-	-	-
	≤ 40% Public	106	14	13	0 / 0	0
	Total	374	92	25	78 / 4	85
5	≥ 40% Public	101	30	30	30 / 5	100
	≥ 40% Tribal	0	-	-	-	-
	≤ 40% Public	121	12	10	0 / 0	0
	Total	222	42	19	30 / 5	71
6	≥ 40% Public	273	63	23	63 / 12	100
	≥ 40% Tribal	0	-	-	-	-
	≤ 40% Public	80	8	10	0 / 2	0
	Total	353	71	20	63 / 14	89
7	≥ 40% Public	111	37	33	36 / 4	97
	≥ 40% Tribal	0	-	-	-	-
	≤ 40% Public	75	8	11	0 / 14	0
	Total	186	45	24	36 / 18	80
10	≥ 40% Public	177	44	25	39 / 4	89
	≥ 40% Tribal	103	31	30	0 / 0	0
	≤ 40% Public	527	53	10	0 / 8	0
	Total	807	128	16	39 / 12	30
11	≥ 40% Public	174	39	22	38 / 0	97
	≥ 40% Tribal	2	1	50	0 / 0	0
	≤ 40% Public	767	77	10	0 / 1	0
	Total	943	117	12	38 / 1	32
12	≥ 40% Public	141	39	28	39 / 18	100
	≥ 40% Tribal	128	36	28	0 / 0	0
	≤ 40% Public	480	48	10	0 / 38	0
	Total	749	123	16	39 / 56	32
13	≥ 40% Public	0	-	-	-	-
	≥ 40% Tribal	10	4	40	0 / 0	0
	≤ 40% Public	66	7	11	0 / 0	0
	Total	76	11	14	0 / 0	0
Total ≥ 40% Public		1632	455	28	429 / 58	94
Total ≥ 40% Tribal		290	89	31	0 / 0	0
Total ≤ 40% Public		2306	238	10	0 / 71	0
Overall Total		4228	780	18	429 / 129	55

Table 2.4. Categorical variable descriptions (a) and independent variables used (b) in classification trees.

(a)

Independent Variable	Variable Description
Permanent/Ephemeral	Water permanent (1) or ephemeral (0)
Eveg/No Eveg	Emergent vegetation present (1) or absent (0)
Fish/No Fish	Fish detected (1) or not (0)
Timber Harvest/No Timber Harvest	Trees around site recently harvested (1) or not (0)
Burned/Not Burned	Forest around site recently burned (1) or not (0)
J or A Amphibians	Juvenile or adult amphibians detected (1) or not (0)
L Amphibians	Larval amphibians detected (1) or not (0)

(b)

Species	Independent Variables Used in Model
Ambystoma macrodactylum	Permanence + Eveg + Fish
Ambystoma tigrinum	Permanence + Eveg + Fish
Spea bombifrons	Permanence + Eveg + Fish
Bufo boreas	Permanence + Eveg + Fish + Burned + Timber Harvest
Bufo cognatus	Permanence + Eveg + Fish
Bufo woodhousii	Permanence + Eveg + Fish
Rana pipiens	Permanence + Eveg + Fish
Rana luteiventris	Permanence + Eveg + Fish
Chrysemys picta	Permanence + Eveg + Fish
Thamnophis elegans	Permanence + Eveg + Fish + J or A Amphibians + L Amphibians
Thamnophis radix	Permanence + Eveg + Fish + J or A Amphibians + L Amphibians
Thamnophis sirtalis	Permanence + Eveg + Fish + J or A Amphibians + L Amphibians

Table 2.5. Naïve watershed and lentic site occupancy rates for breeding by amphibians and occupancy by aquatic reptiles. Rates are rounded to the nearest percent. Only watersheds and sites within the known range of the species in each strata were used for calculations (Figures 2.2-2.3).

Strata	Total Number Watersheds / Sites	Percent Watershed Occupancy (95% CI[a])	Percent Site Occupancy (95% CI[b])
Ambystoma macrodactylum			
1	53 / 286	66 (55–77)	44 (39 – 50)
2	36 / 638	75 (63–87)	31 (28 – 35)
3	4 / 43	24 (2–98)	12 (2 – 21)
4	65 / 803	77 (68–86)	44 (41 – 48)
5	3 / 11	33 (0–87)	18 (0 – 41)
6	24 / 338	58 (39–78)	10 (7 – 13)
Overall	185 / 2119	70 (64–76)	34 (32 – 36)
Ambystoma tigrinum			
1	2 / 17	0 (-)	0 (-)
6	14 / 222	50 (13–24)	20 (14–25)
7	27 / 749	30 (14–45)	3 (2–4)
10	37 / 922	73 (60–86)	11 (9–13)
11	26 / 139	77 (62–92)	40 (31–48)
12	34 / 487	79 (67–91)	28 (24–32)
Overall	140 / 2536	64 (56–71)	14 (13–15)
Spea bombifrons			
10	37 / 848	14 (4–24)	2 (1–3)
11	29 / 1084	21 (7–34)	3 (2–4)
12	34 / 491	24 (11–36)	4 (2–6)
Overall	100 / 1578	19 (12–26)	3 (3–4)
Bufo boreas			
1	52 / 283	17 (8–26)	3 (1–5)
2	36 / 626	17 (6–27)	1 (0–2)
3	4 / 43	50 (2–98)	5 (0–11)
4	64 / 788	23 (14–33)	4 (3–5)
5	19 / 82	11 (0–23)	2 (0–6)
6	53 / 729	23 (12–33)	3 (2–4)
7	29 / 768	0 (-)	0 (-)
10	1 / 1	0 (-)	0 (-)
11	7 / 30	0 (-)	0 (-)
12	1 / 7	0 (-)	0 (-)
Overall	266 / 3357	17 (13–22)	2 (2–3)
Bufo cognatus			
10	37 / 929	22 (10–34)	2 (1–3)
11	26 / 139	15 (2–28)	3 (0–6)
12	33 / 484	3 (0–8)	1 (0–1)
Overall	96 / 1552	14 (7–20)	1 (1–2)
Bufo woodhousii			
10	37 / 928	19 (7–30)	2 (1–2)
11	63 / 137	19 (11–27)	26 (18–33)
12	33 / 478	42 (27–57)	16 (13–19)
Overall	133 / 1543	25 (18–31)	8 (7–9)

Pseudacris maculata			
3	1 / 10	1 (-)	40 (9–71)
5	16 / 74	0 (-)	0 (-)
6	22 / 327	36 (17–56)	13 (9–17)
7	29 / 769	0 (-)	0 (-)
10	37 / 855	97 (92–100)	47 (44–51)
11	29 / 160	59 (42–75)	49 (41–57)
12	33 / 472	61 (46–75)	35 (31–39)
Overall	167 / 2667	49 (42–56)	26 (24–28)
Pseudacris regilla			
1	53 / 282	43 (32–55)	18 (14–23)
2	24 / 539	4 (0–12)	0.2 (0–1)
4	55 / 2191	4 (0–8)	5 (4–6)
Overall	132 / 1370	20 (14–25)	4 (3–5)
Rana pipiens			
10	37 / 853	62 (48–76)	14 (11–16)
11	24 / 126	25 (9–41)	6 (2–11)
12	33 / 434	82 (70–94)	18 (14–22)
Overall	94 / 1435	60 (50–69)	14 (12–16)
Rana luteiventris			
1	49 / 233	61 (49–73)	30 (24–36)
2	34 / 521	67 (54–82)	22 (18–26)
3	4 / 29	50 (2–98)	7 (0–16)
4	61 / 611	77 (68–87)	45 (41–49)
5	18 / 69	56 (34–77)	32 (21–43)
6	51 / 571	73 (61–84)	40 (36–44)
7	28 / 708	57 (41–73)	13 (11–16)
11	7 / 28	0 (-)	0 (-)
12	3 / 10	33 (0–87)	50 (18–82)
Overall	256 / 2781	65 (59–70)	29 (27–31)
Chrysemys picta			
1	53 / 287	6 (0–11)	1 (0–3)
2	36 / 639	11 (2–20)	1 (0–1)
3	4 / 43	25 (0–67)	2 (0–7)
4	65 / 803	2 (0–4)	0.1 (0–0.3)
5	19 / 86	0 (-)	0 (-)
6	53 / 752	0 (-)	0 (-)
7	29 / 769	0 (-)	0 (-)
10	38 / 930	55 (41–70)	9 (8–11)
11	23 / 161	13 (0–26)	2 (0–5)
12	34 / 491	53 (38–68)	9 (6–11)
Overall	354 / 4961	14 (11–17)	3 (2–3)
Thamnophis elegans			
1	53 / 287	19 (10–28)	4 (2–6)
2	36 / 639	31 (17–44)	3 (2–4)
3	4 / 43	25 (0–67)	2 (0–7)
4	65 / 803	26 (17–36)	5 (3–6)
5	19 / 86	26 (8–45)	8 (2–14)
6	53 / 752	42 (29–54)	7 (5–8)
7	29 / 769	28 (13–42)	2 (1–3)
10	12 / 183	0 (-)	0 (-)
11	24 / 135	4 (0–12)	1 (0–2)
12	32 / 485	6 (0–14)	0.4 (0–1)
Overall	327 / 4182	24 (19–28)	3 (3–4)

Thamnophis radix			
10	37 / 929	86 (76–96)	18 (15–20)
11	24 / 133	42 (23–60)	20 (13–26)
12	25 / 460	52 (34–70)	11 (8–14)
Overall	86 / 1522	64 (54–73)	16 (14–18)
Thamnophis sirtalis			
1	53 / 287	13 (5–21)	3 (1–5)
2	36 / 639	22 (10–34)	5 (4–7)
3	4 / 43	25 (0–67)	5 (0–11)
4	65 / 803	32 (22–42)	8 (6–10)
5	19 / 86	0 (-)	0 (-)
6	53 / 752	8 (1–14)	1 (0–1)
7	29 / 769	0 (-)	0 (-)
10	1 / 1	0 (-)	0 (-)
11	21 / 122	0 (-)	0 (-)
12	34 / 491	6 (0–13)	1 (0–2)
Overall	315 / 3993	14 (10–17)	3 (2–4)

[a] Calculated using a standard error formula with a finite population correction factor since we know the number of public dominated watersheds in each target population. SE = (((occupancy rate * (1 – occupancy rate)) / n) * (1 – n / N)) ^ (1/2).

[b] Calculated using a standard error formula without a finite population correction factor since we do not know the total number of lentic sites in each target population. SE = ((occupancy rate * (1 – occupancy rate)) / n) ^ (1/2).

Table 2.6. Summary of responses to fish, emergent vegetation, and preferred hydroperiod in classification tree analyses. Positive (+), negative (-), neutral (●), (P = permanent, E = ephemeral). See Figures 2.5a-n for magnitude of responses.

Species	Fish Presence	Emergent Vegetation	Preferred Hydroperiod
Ambystoma macrodacytlum	-	+[1]	P / E
Ambystoma tigrinum	-	+[1]	P / E
Spea bombifrons	●	●	E
Bufo boreas	●	+	P / E
Bufo cognatus	-	+	E
Bufo woodhousii	-	+	P / E
Pseudacris maculata	-	+	P / E
Pseudacris regilla	-	+	P / E
Rana pipiens	+	+	P
Rana luteiventris	●	+	P
Chrysemys picta	●	+	P
Thamnophis elegans	+	+	P / E
Thamnophis radix	+	+	P
Thamnophis sirtalis	-	+	P / E

[1] Partial mitigation of negative response to fish when emergent vegetation present (Figures 2.5a-b).

Figure Legends

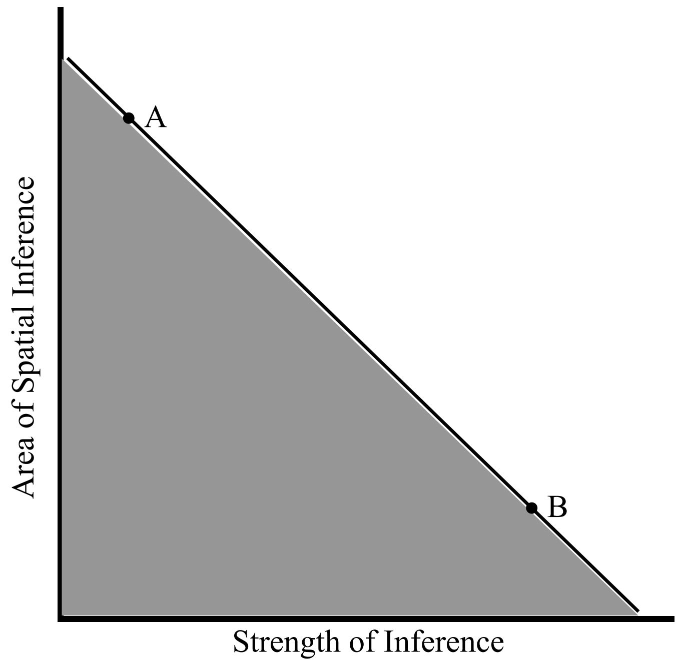
Figure 2.1. Tradeoff between spatial inference and inference strength of an estimator. This results from sample size limitations as a function of cost. The area shaded in gray represents combinations of spatial inference and inference strength where an estimator would not yield any useful information for the management of a species. Two or more estimators that individually either have broad spatial inference, but low strength of inference (A), or strong inference, but only over a small spatial scale (B), are needed in combination to maximize overall inference about the status of a species.

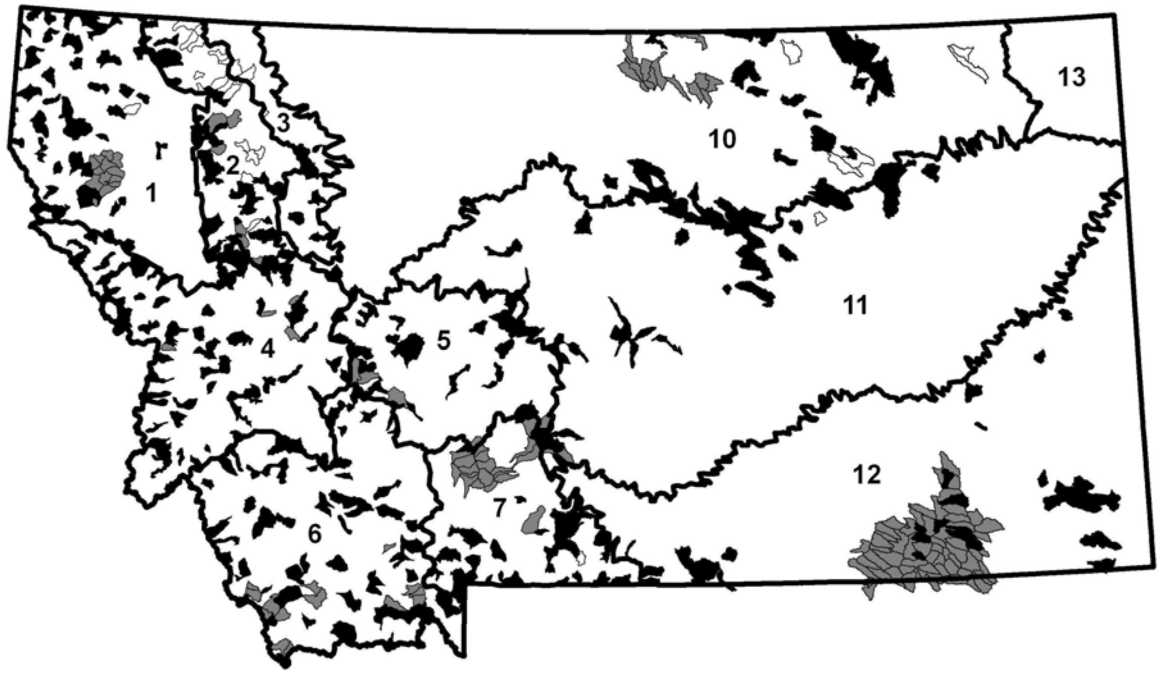
Figure 2.2. Sampling scheme for assessing status and trends in lentic breeding amphibians in Montana. Eleven geographic strata (dark lines) corresponding to those in Table 2.3 are based on a combination of level 3 ecoregions and 8-digit hydrologic unit code (HUC) watersheds. Within strata, 12-digit HUC watersheds containing >40% public lands were randomly selected in numbers proportional to the total area and number of watersheds. Within each watershed all lentic sites on public lands were surveyed to assess the proportion of sites and watersheds with breeding. As of the 2008 field season, watersheds completed and uncompleted are shown in black and outline, respectively; note that most of the uncompleted watersheds at the northern portions of strata 2 and 3 have received intensive surveys under the U.S. Geological Survey's Amphibian Research and Monitoring Initiative (ARMI). A number of additional watersheds were surveyed to address specific management questions (gray). Watersheds dominated by private and tribal lands were also randomly selected for survey (not shown, but summarized in Table 2.3) to define a total of 28 target populations. There are currently no geographic strata numbered 8 or 9 because these strata were part of an earlier sampling scheme to ensure that island mountain ranges in central Montana would be sampled; these are now combined with strata 10 and 11.

Figure 2.3. Characteristics of randomly selected sites surveyed across sample strata. (a) ownership, (b) elevation in meters, (c) site origin, (d) habitat type, (e) water permanence and emergent vegetation, (f) fish presence, (g) grazing, (h) water dammed or diverted.

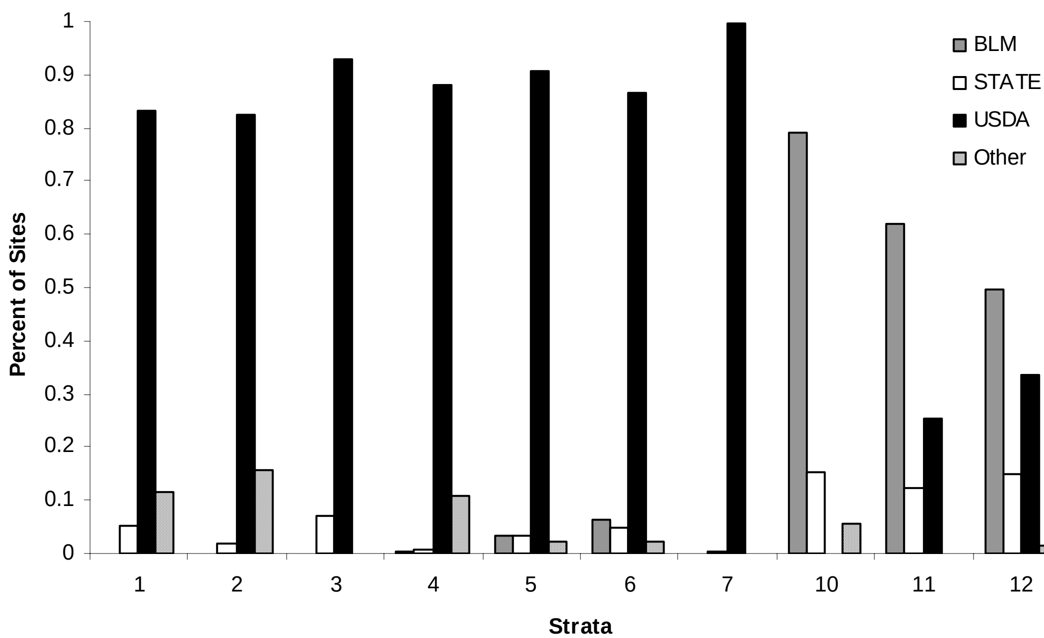
Figure 2.4. Spatial and elevation distributions of lentic breeding amphibians and aquatic reptiles. Black circles are observations of any life history stage from this project. Black crosses are other observations in the Montana Natural Heritage Program's database as of the fall of 2008. Elevation summaries are for detection of breeding (amphibians) or presence (reptiles) at lentic sites surveyed during this project within the known range of the species in Montana. Elevation numbers are lower margins of each elevation class.

Figure 2.5. Classification trees for detection of breeding by amphibians or occupancy of reptiles. Trees were constructed using CART 6.0 employing a splitting rule that uses the Gini index with a minimum of 10 observations per node and tested with a 10-fold cross validation of the same data. Vertical depth of each split is proportional to the amount of variation explained. Nodes indicate the percent of sites occupied (SE), and number of sites with those characteristics. Trees were generally pruned back to the number of terminal nodes resulting in a minimum relative error rate. However, when there was no clear minimum error rate the number of nodes within 1 SE of the optimal tree was minimized to reduce complexity and ensure interpretability of models. See Table 2.4 for variables used and variable descriptions.


A
B
Area of Spatial Inference
Strength of Inference

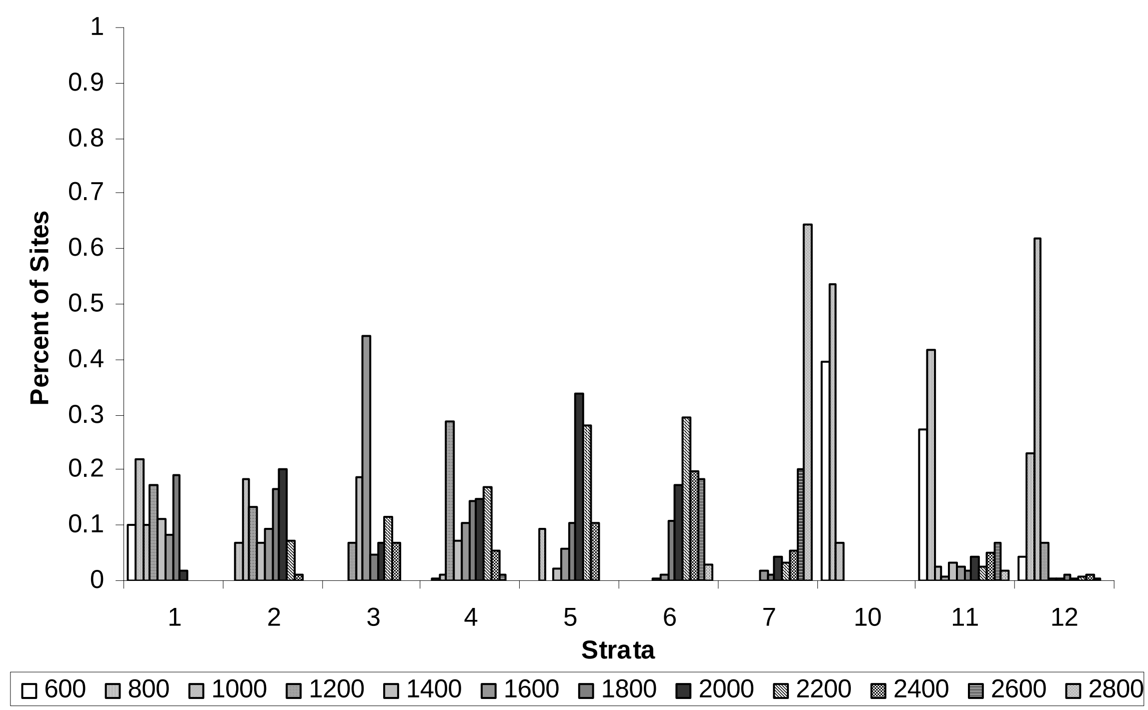

1
2
3
4
5
6
7
10
11
12
13

(a)



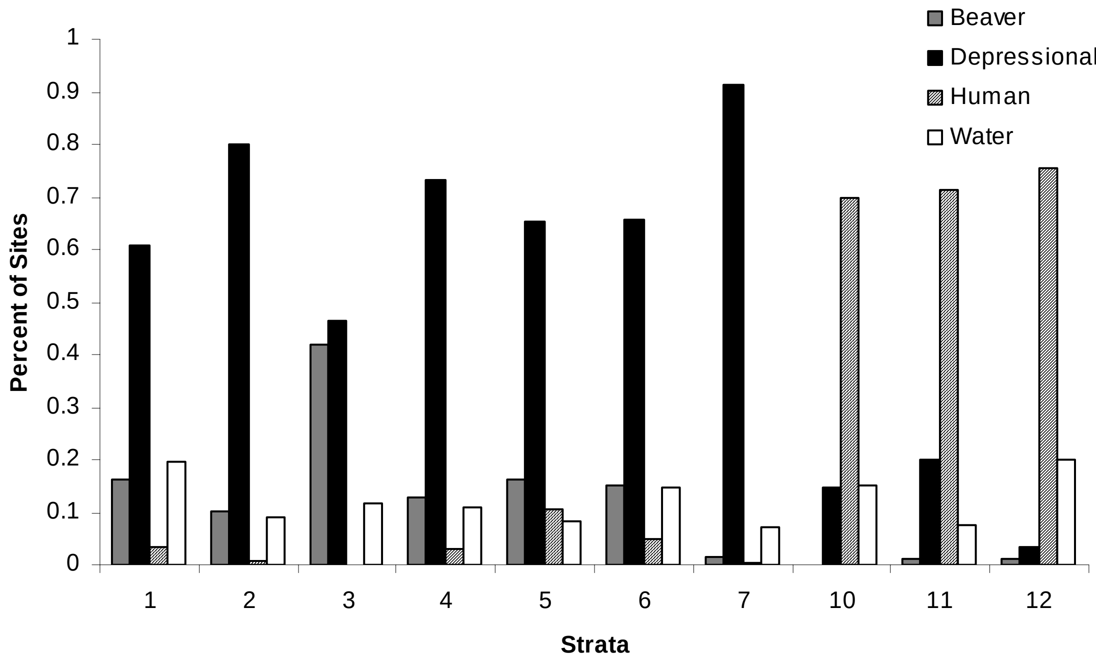
Percent of Sites
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
1 2 3 4 5 6 7 10 11 12
Strata
BLM
STATE
USDA
Other

(b)



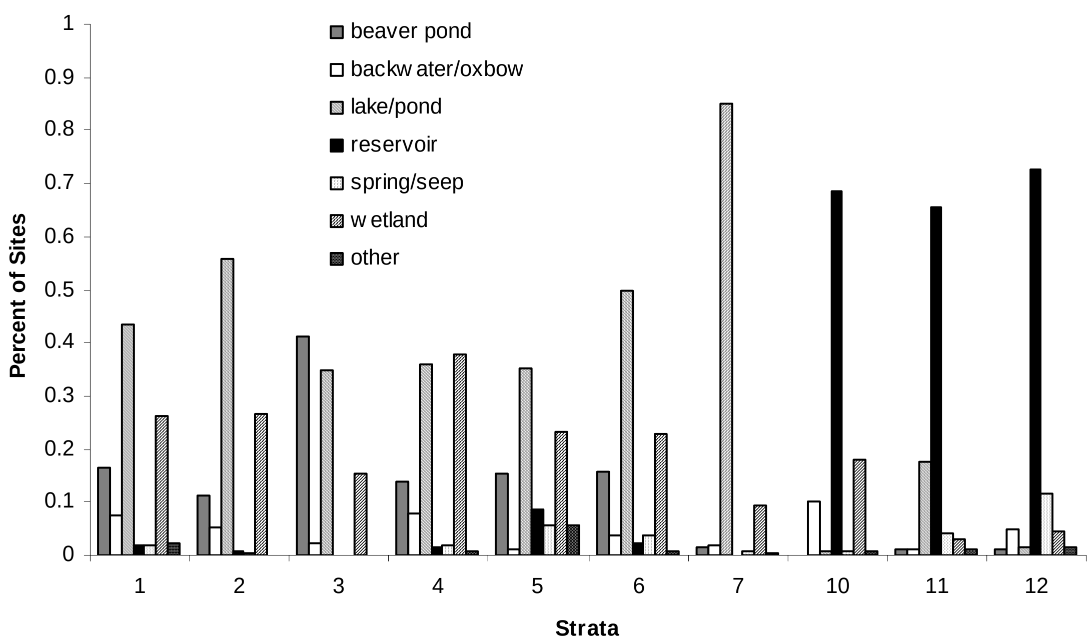
Percent of Sites
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
1 2 3 4 5 6 7 10 11 12
Strata
600 800 1000 1200 1400 1600 1800 2000 2200 2400 2600 2800

(c)



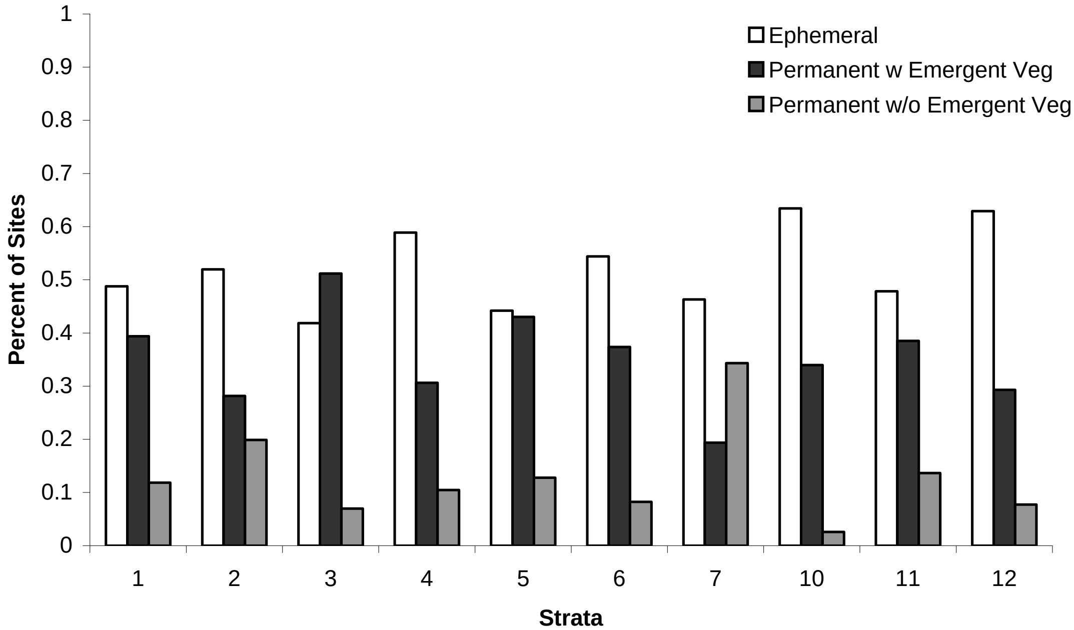
Beaver
Depressional
Human
Water
Percent of Sites
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
1
2
3
4
5
6
7
10
11
12
Strata

(d)

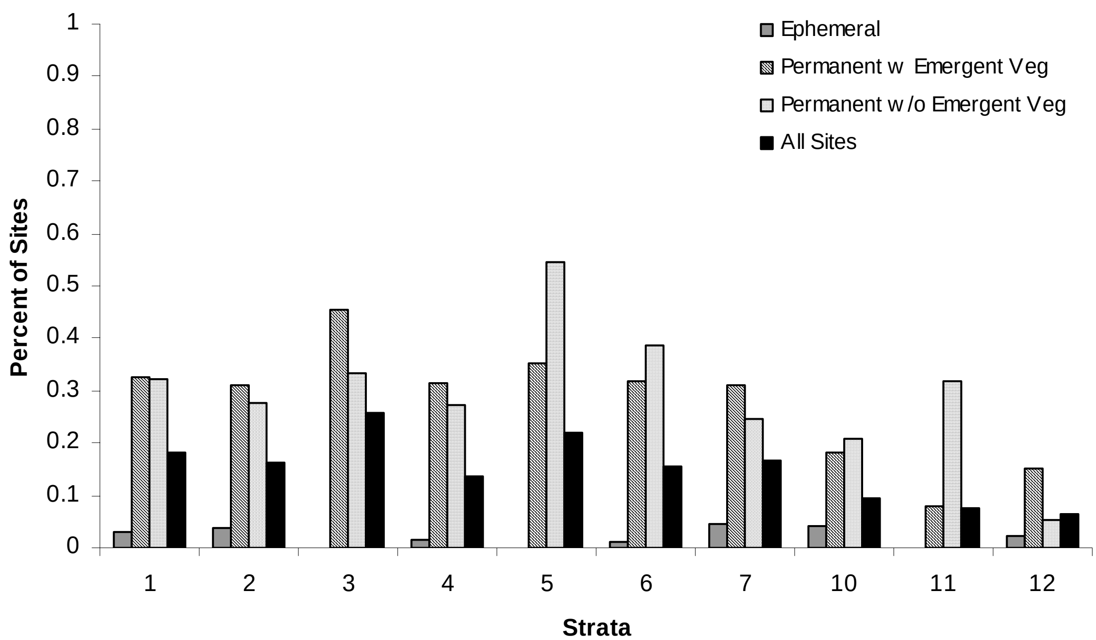


beaver pond
backw ater/oxbow
lake/pond
reservoir
spring/seep
w etland
other
Percent of Sites
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
1
2
3
4
5
6
7
10
11
12
Strata

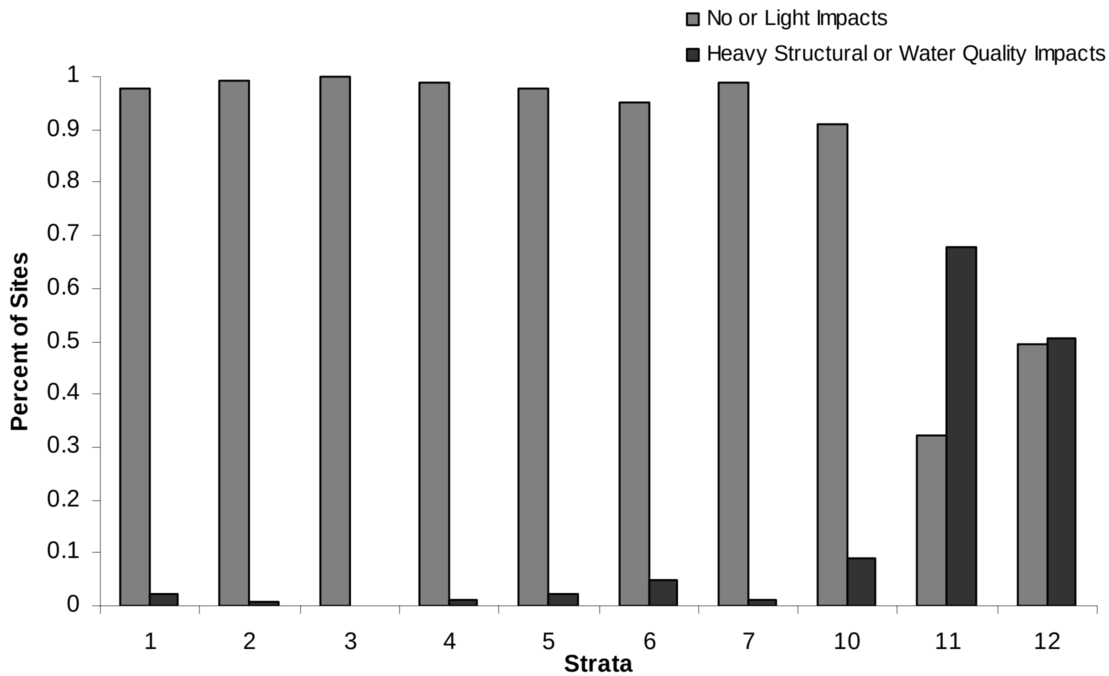
(e)


Ephemeral
Permanent w Emergent Veg
Permanent w/o Emergent Veg
Percent of Sites
Strata
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1
1
2
3
4
5
6
7
10
11
12

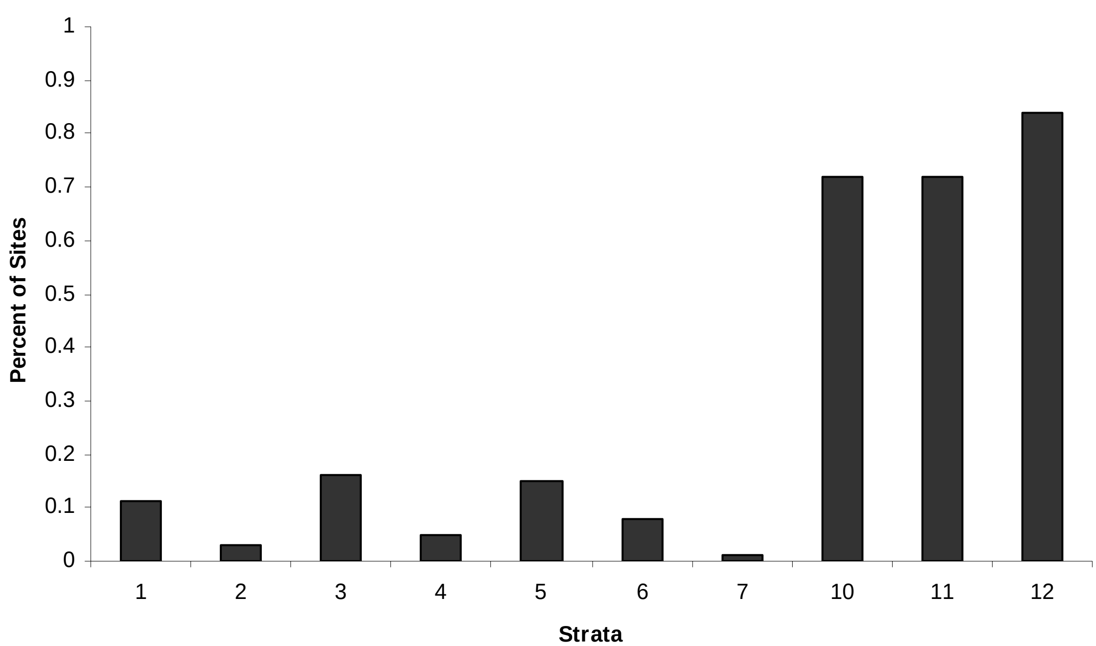
(f)


Ephemeral
Permanent w Emergent Veg
Permanent w /o Emergent Veg
All Sites
Percent of Sites
Strata
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1
1
2
3
4
5
6
7
10
11
12

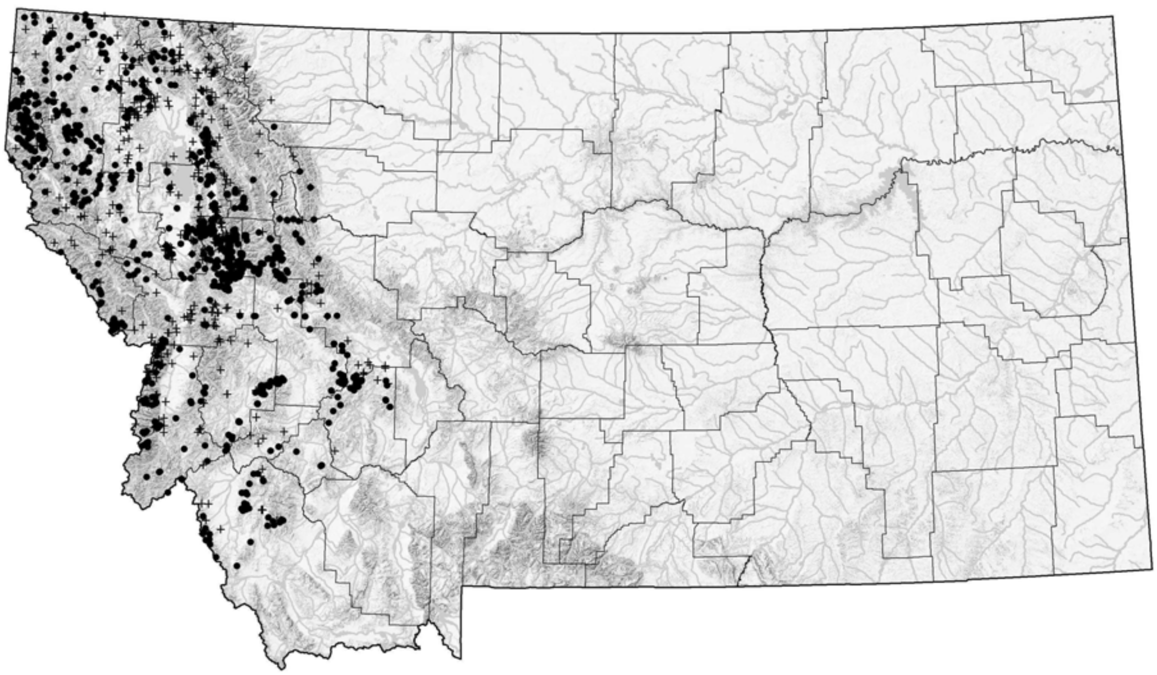
(g)

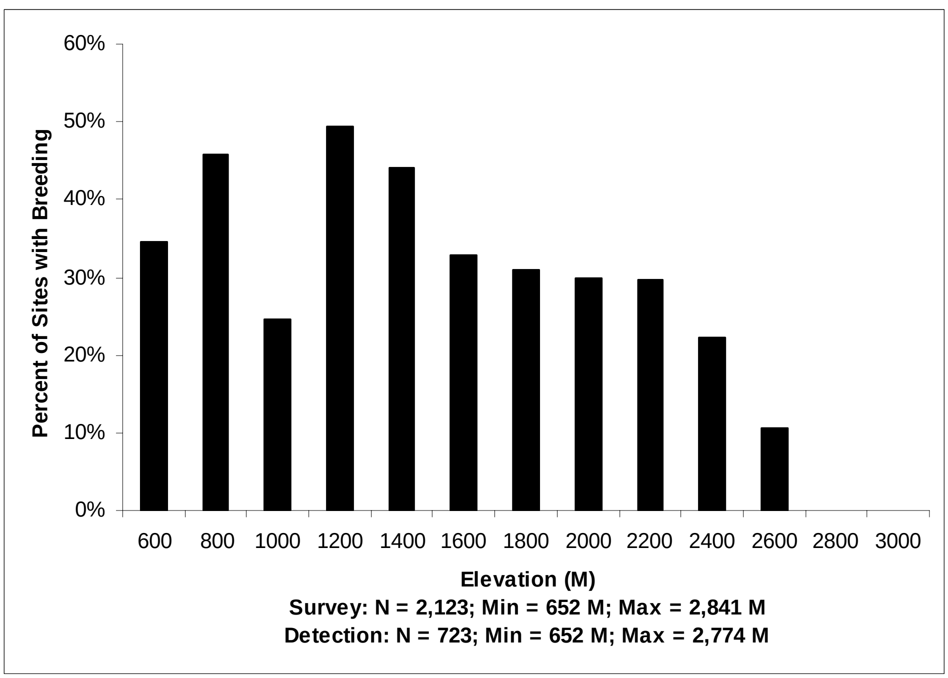

No or Light Impacts
Heavy Structural or Water Quality Impacts
Percent of Sites
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1
1
2
3
4
5
6
7
10
11
12
Strata

(h)

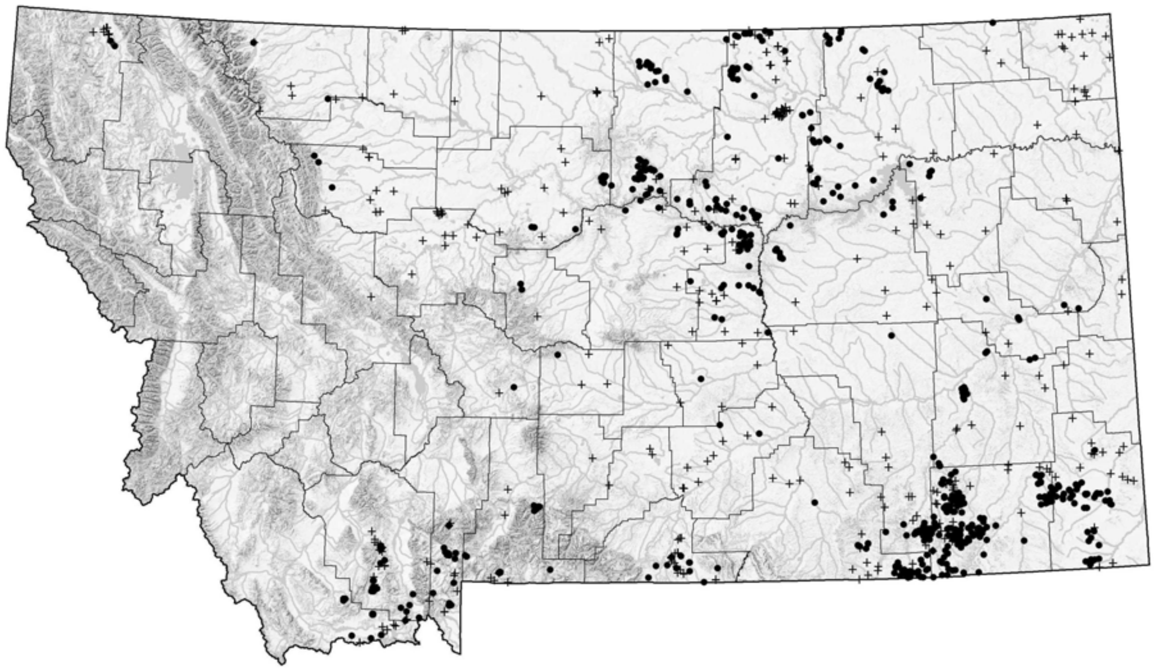

Percent of Sites
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1
1
2
3
4
5
6
7
10
11
12
Strata

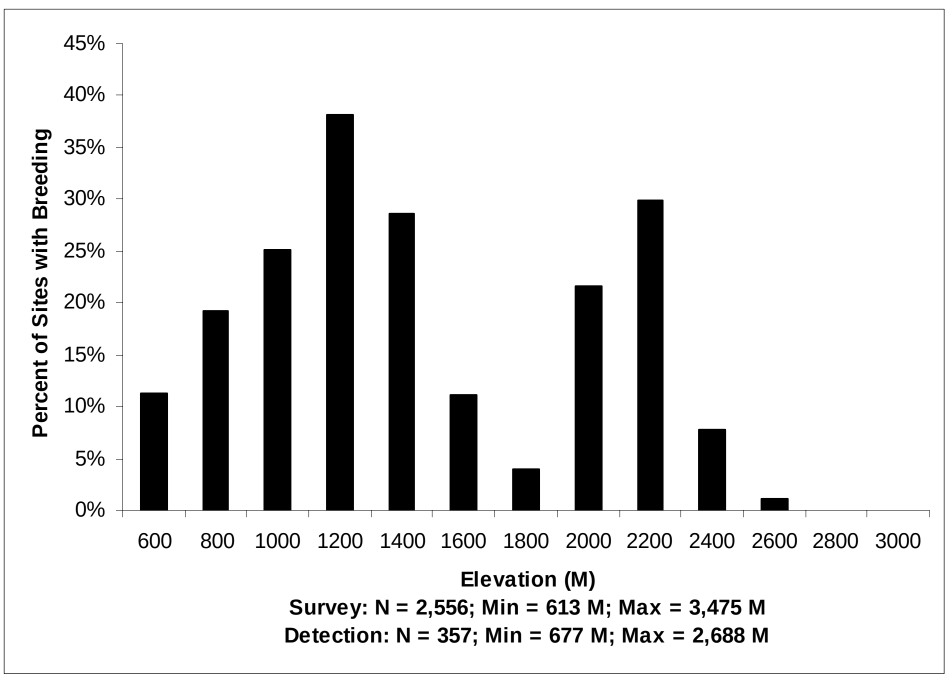
(a)




60%
50%
40%
30%
20%
10%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 2,123; Min = 652 M; Max = 2,841 M
Detection: N = 723; Min = 652 M; Max = 2,774 M

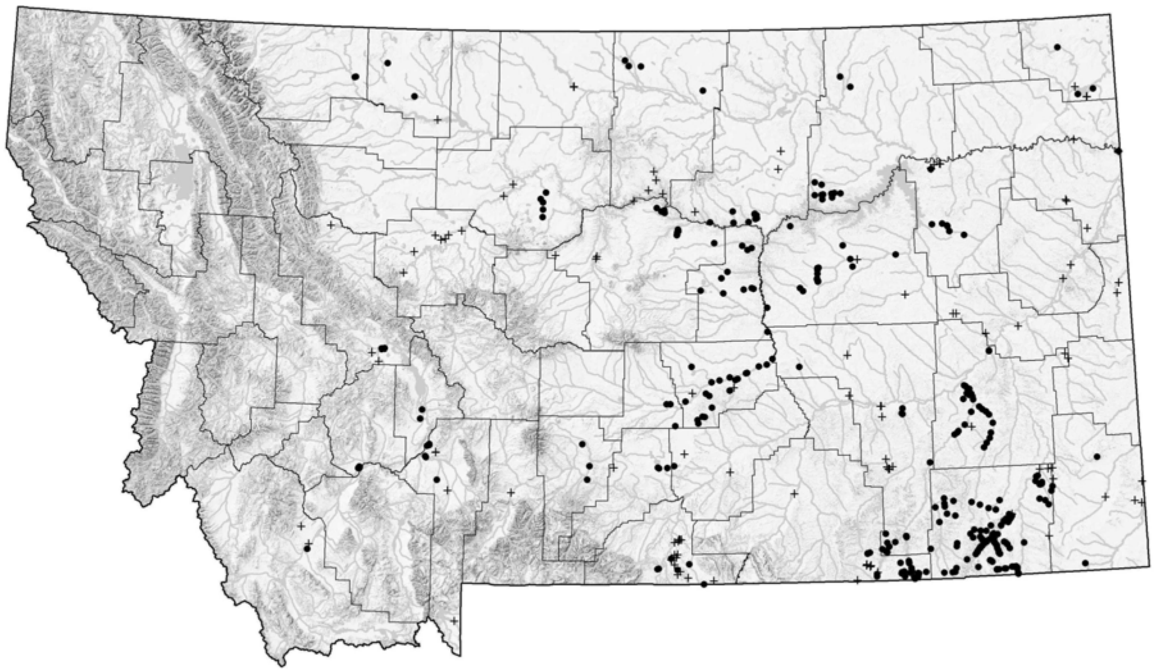
(b)

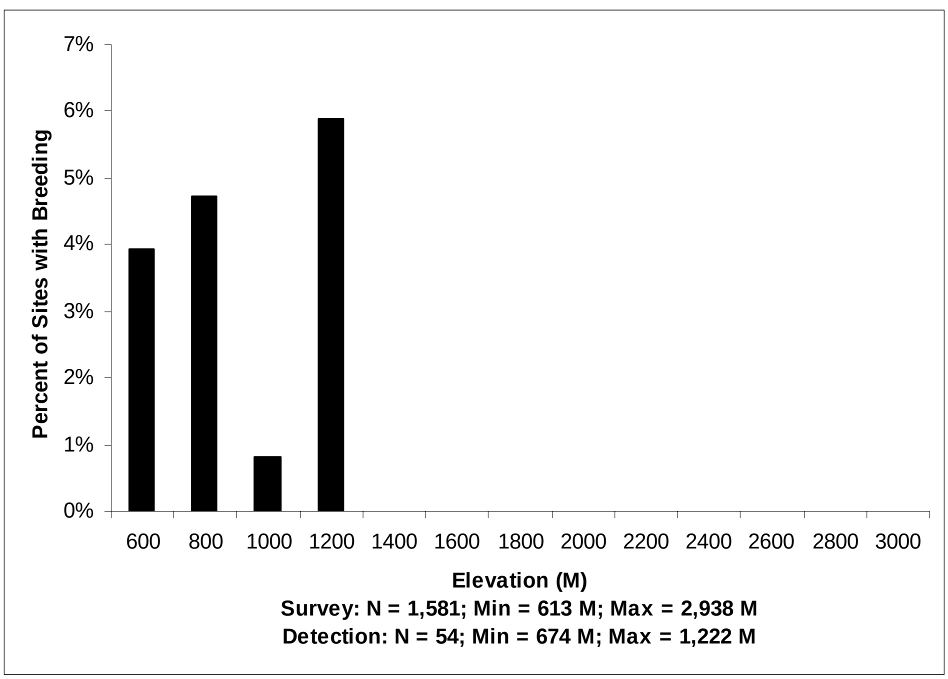




45%
40%
35%
30%
25%
20%
15%
10%
5%
0%
Percent of Sites with Breeding
600 800 1000 1200 1400 1600 1800 2000 2200 2400 2600 2800 3000
Elevation (M)
Survey: N = 2,556; Min = 613 M; Max = 3,475 M
Detection: N = 357; Min = 677 M; Max = 2,688 M

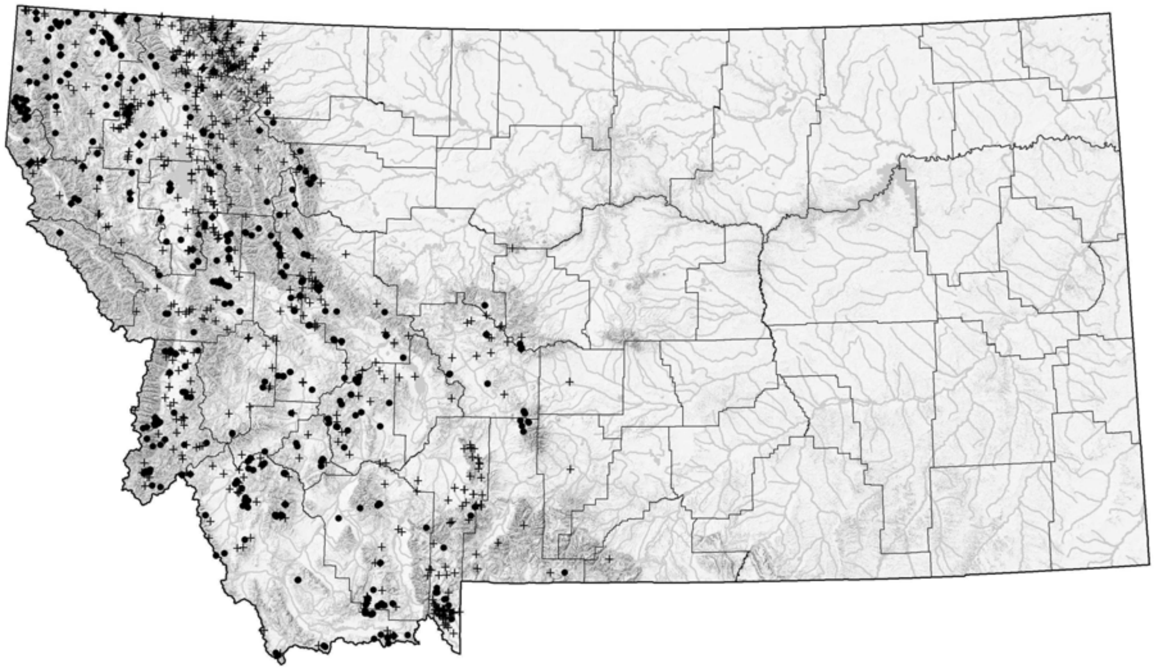
(c)

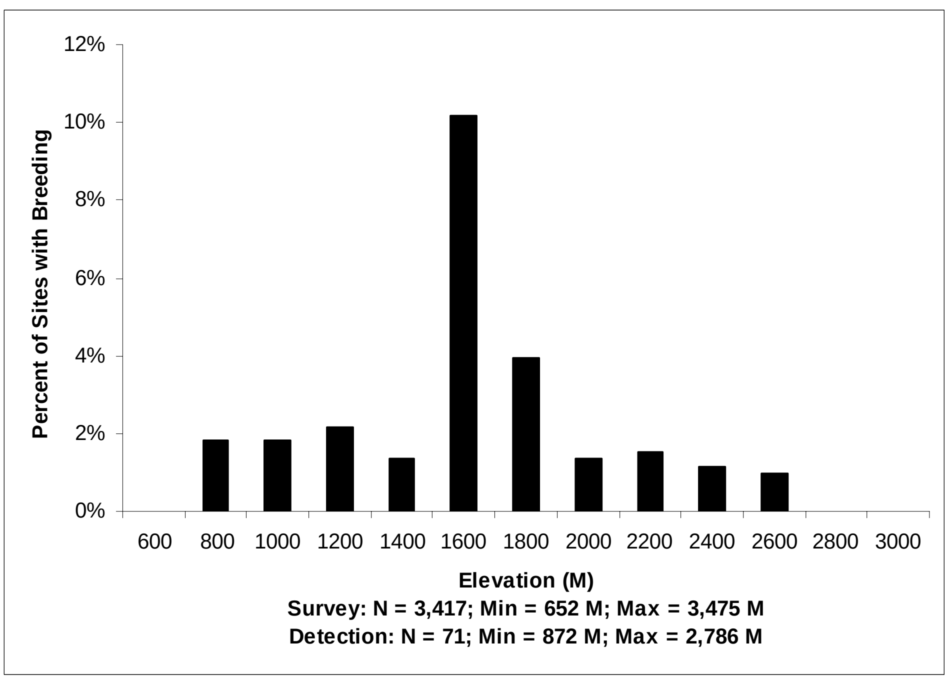




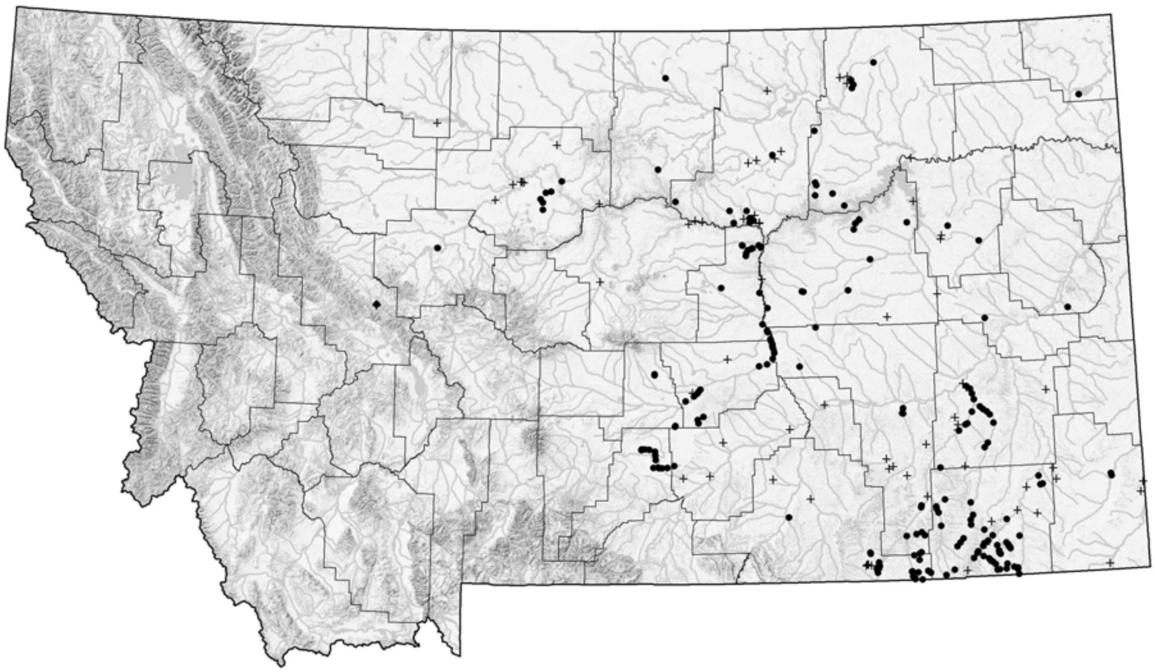
7%
6%
5%
4%
3%
2%
1%
0%
Percent of Sites with Breeding
600 800 1000 1200 1400 1600 1800 2000 2200 2400 2600 2800 3000
Elevation (M)
Survey: N = 1,581; Min = 613 M; Max = 2,938 M
Detection: N = 54; Min = 674 M; Max = 1,222 M

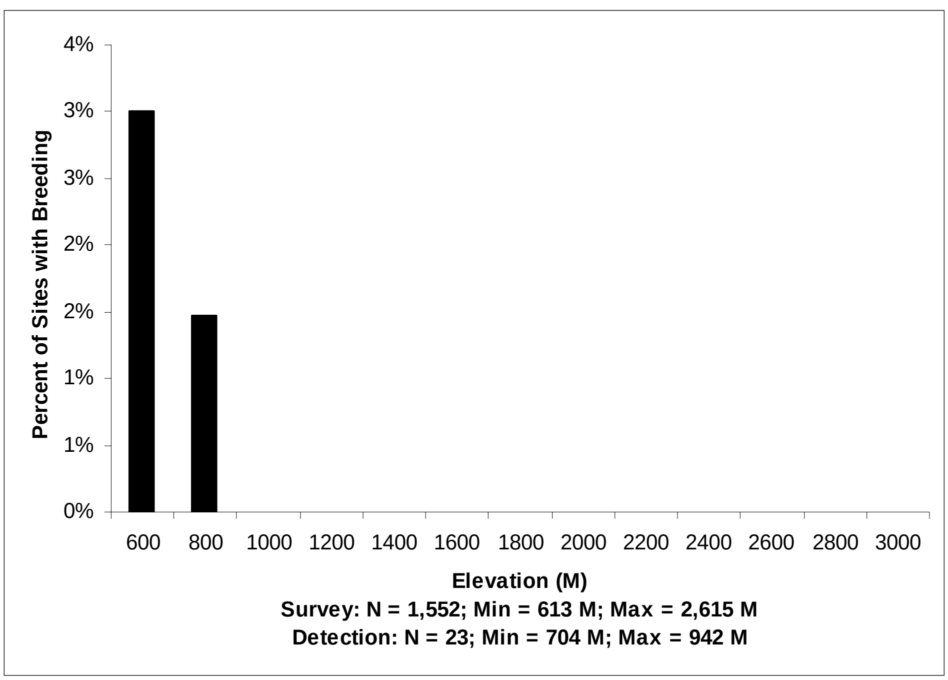
(d)



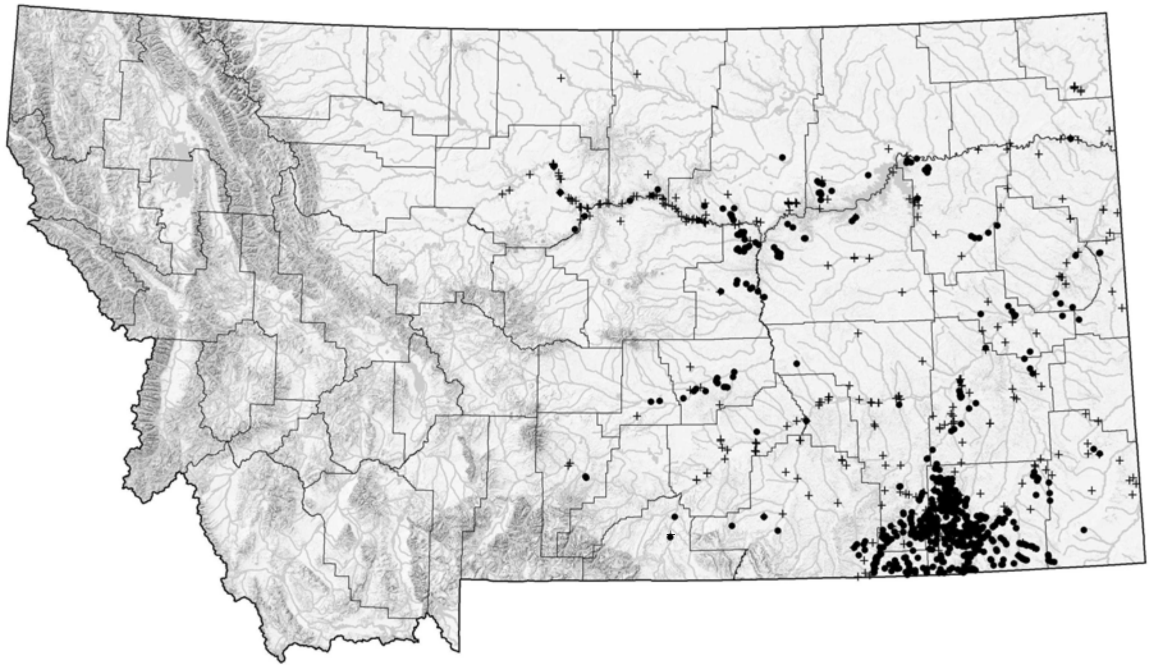

12%
10%
8%
6%
4%
2%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 3,417; Min = 652 M; Max = 3,475 M
Detection: N = 71; Min = 872 M; Max = 2,786 M

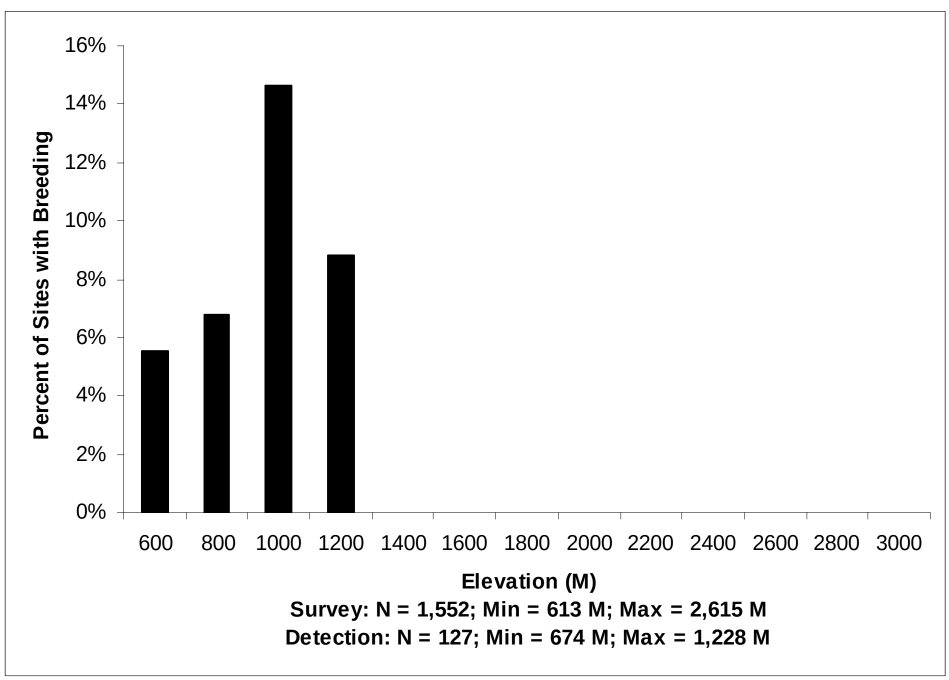
(e)



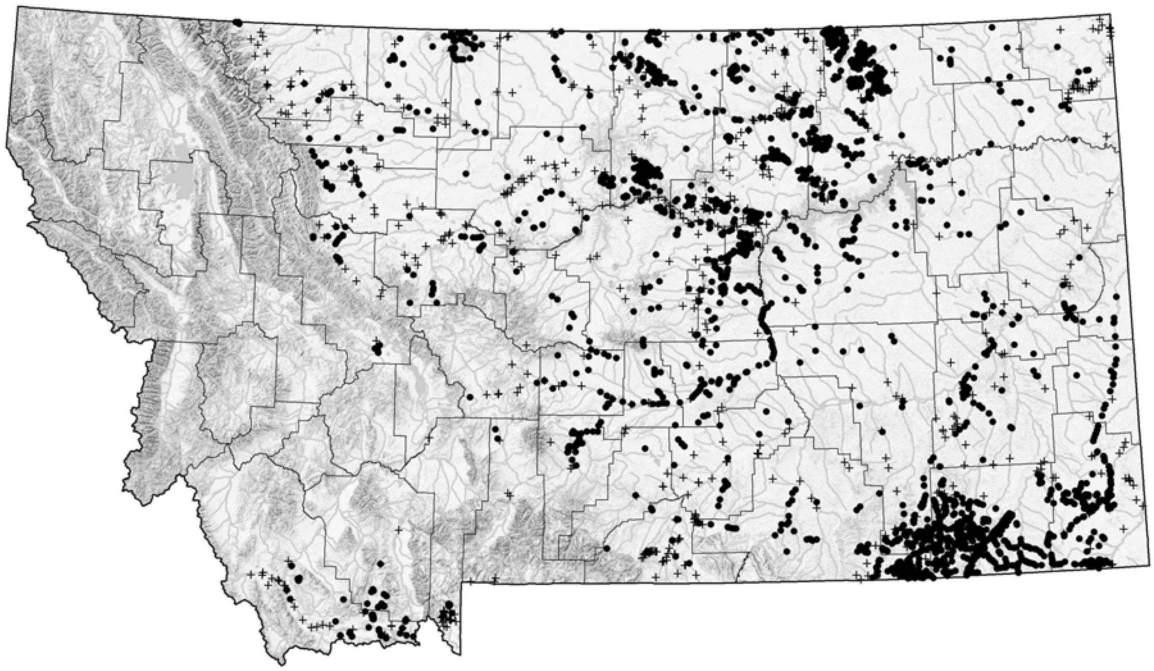

4%
3%
3%
2%
2%
1%
1%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 1,552; Min = 613 M; Max = 2,615 M
Detection: N = 23; Min = 704 M; Max = 942 M

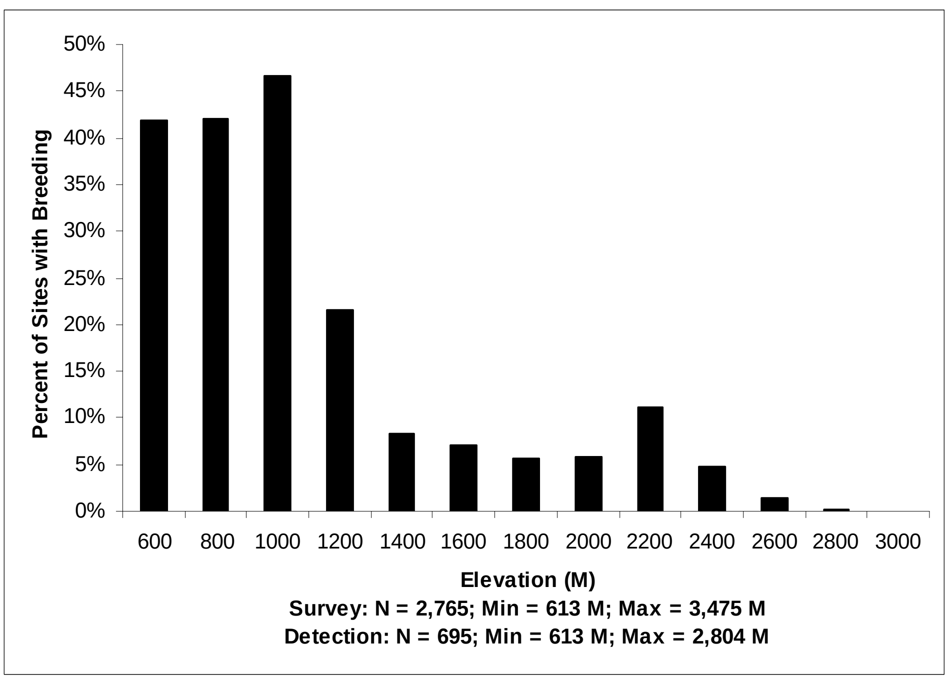
(f)




16%
14%
12%
10%
8%
6%
4%
2%
0%
Percent of Sites with Breeding
600 800 1000 1200 1400 1600 1800 2000 2200 2400 2600 2800 3000
Elevation (M)
Survey: N = 1,552; Min = 613 M; Max = 2,615 M
Detection: N = 127; Min = 674 M; Max = 1,228 M

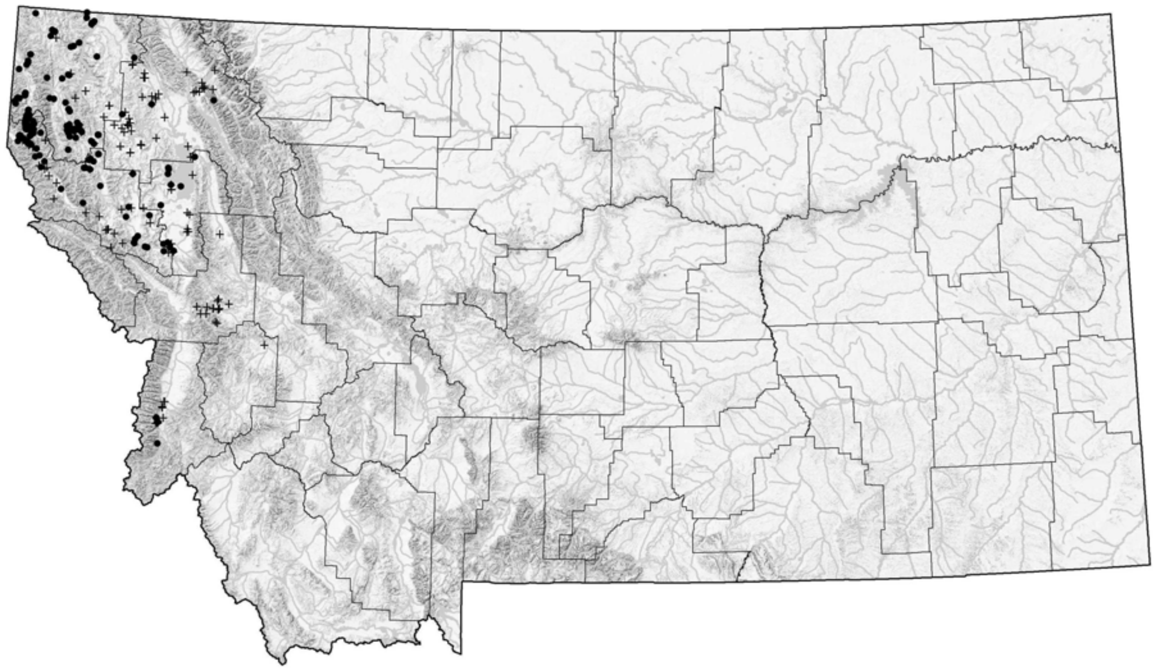
(g)

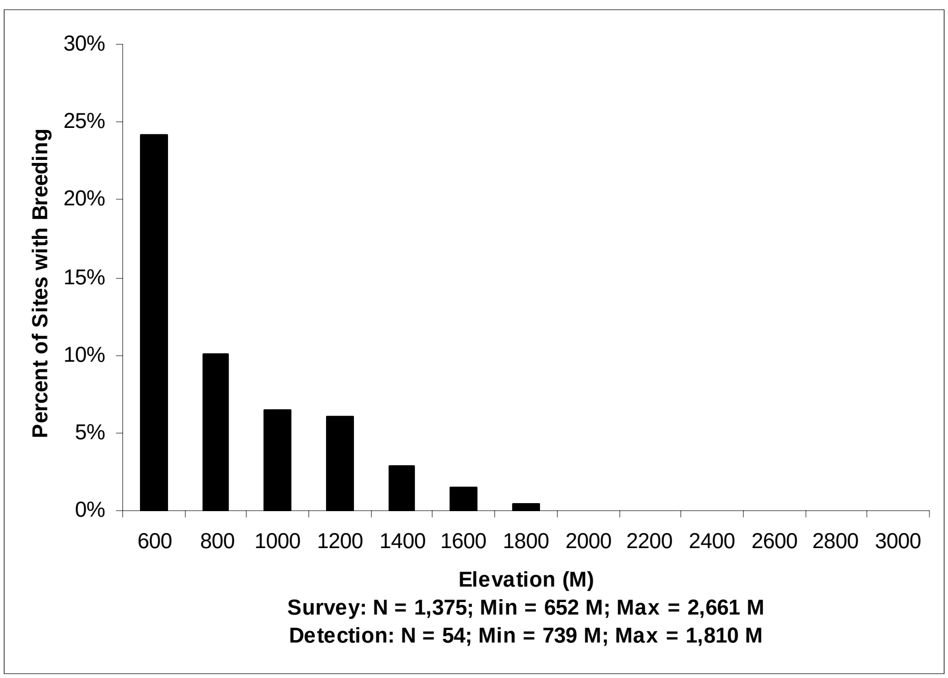




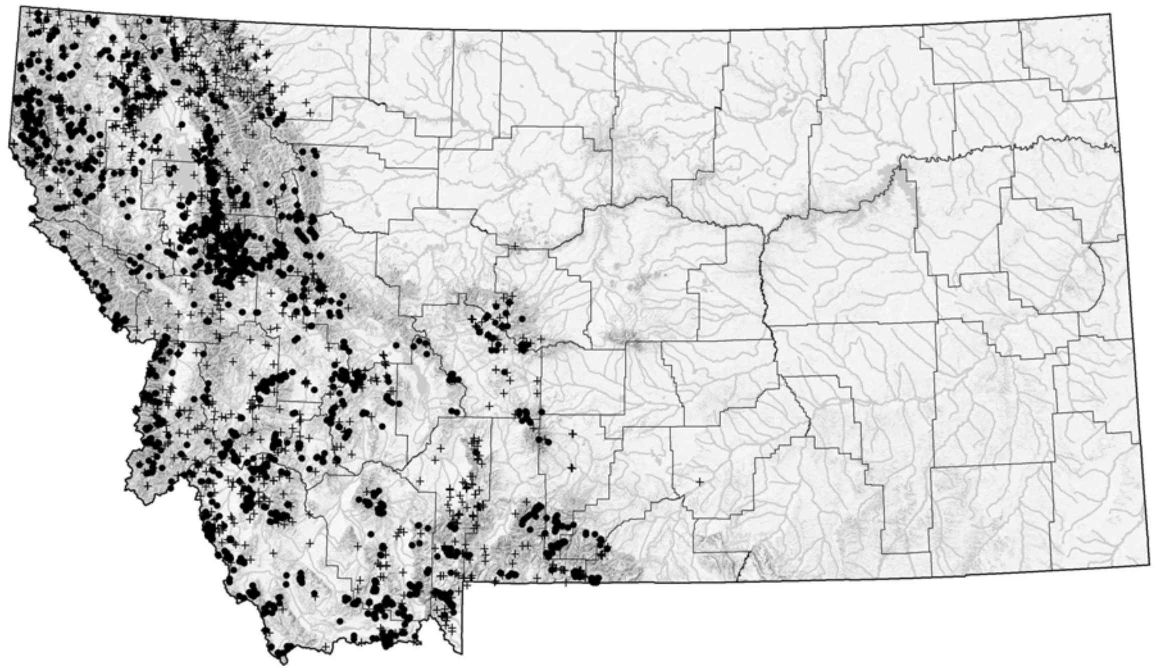
50%
45%
40%
35%
30%
25%
20%
15%
10%
5%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 2,765; Min = 613 M; Max = 3,475 M
Detection: N = 695; Min = 613 M; Max = 2,804 M

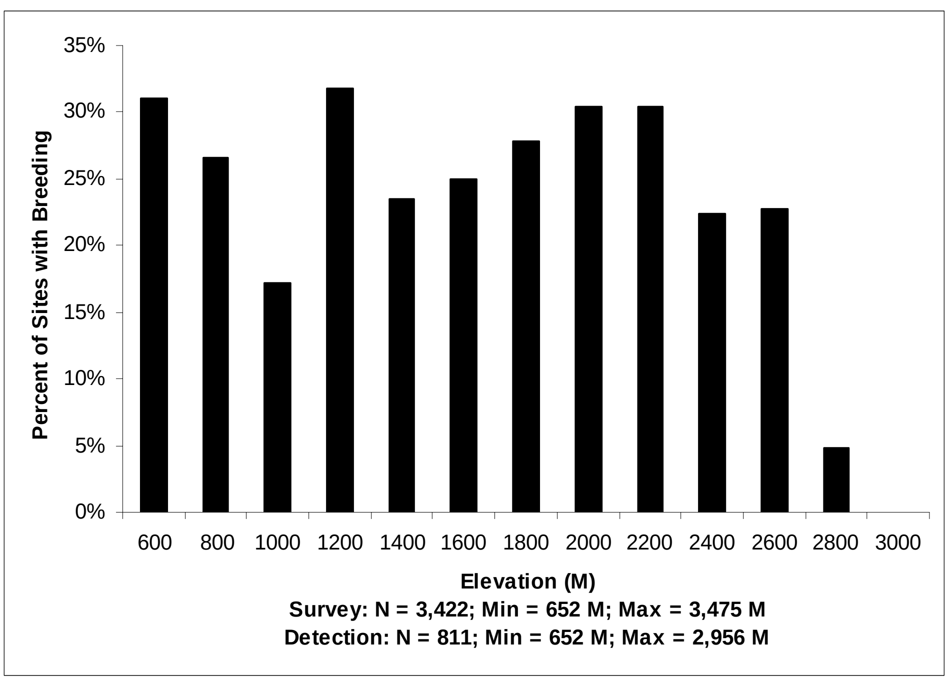
(h)



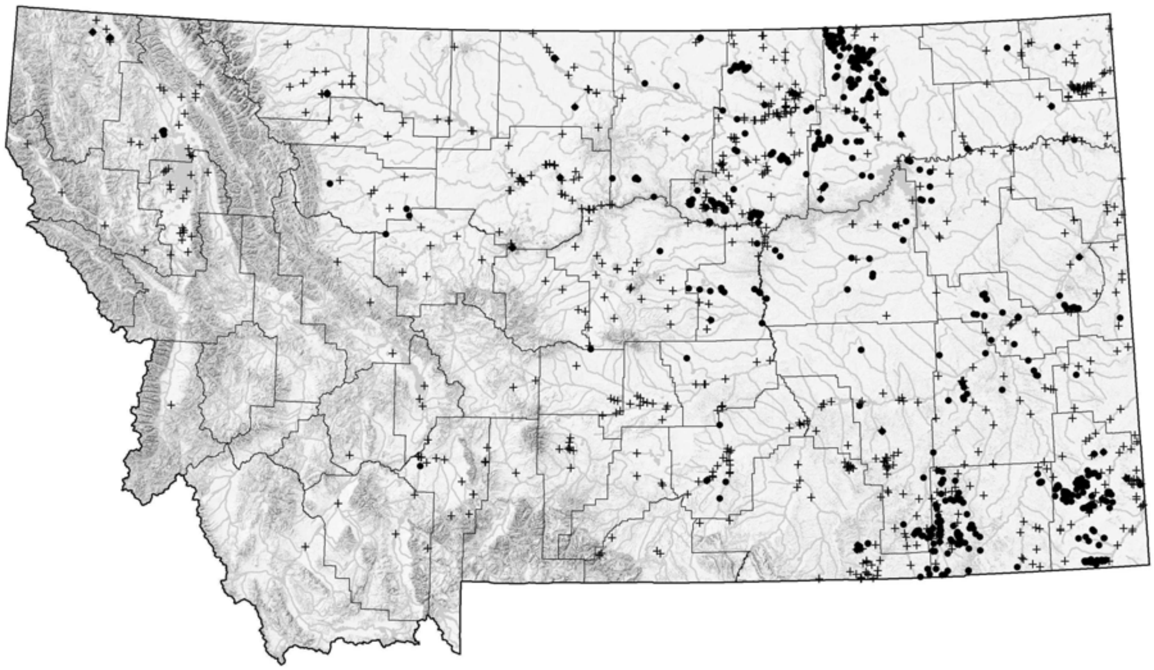

30%
25%
20%
15%
10%
5%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 1,375; Min = 652 M; Max = 2,661 M
Detection: N = 54; Min = 739 M; Max = 1,810 M

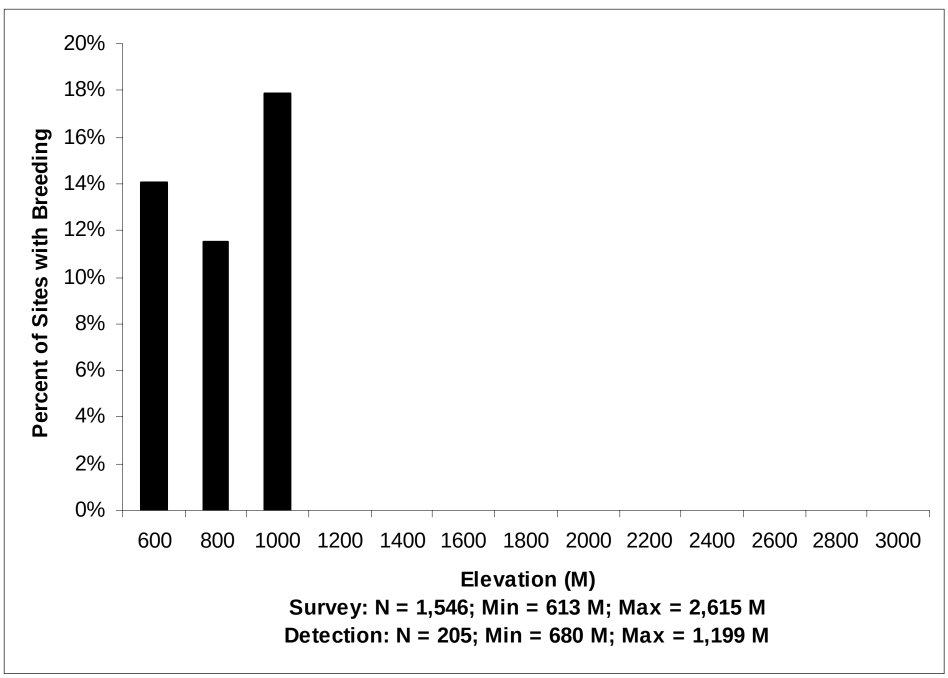
(i)



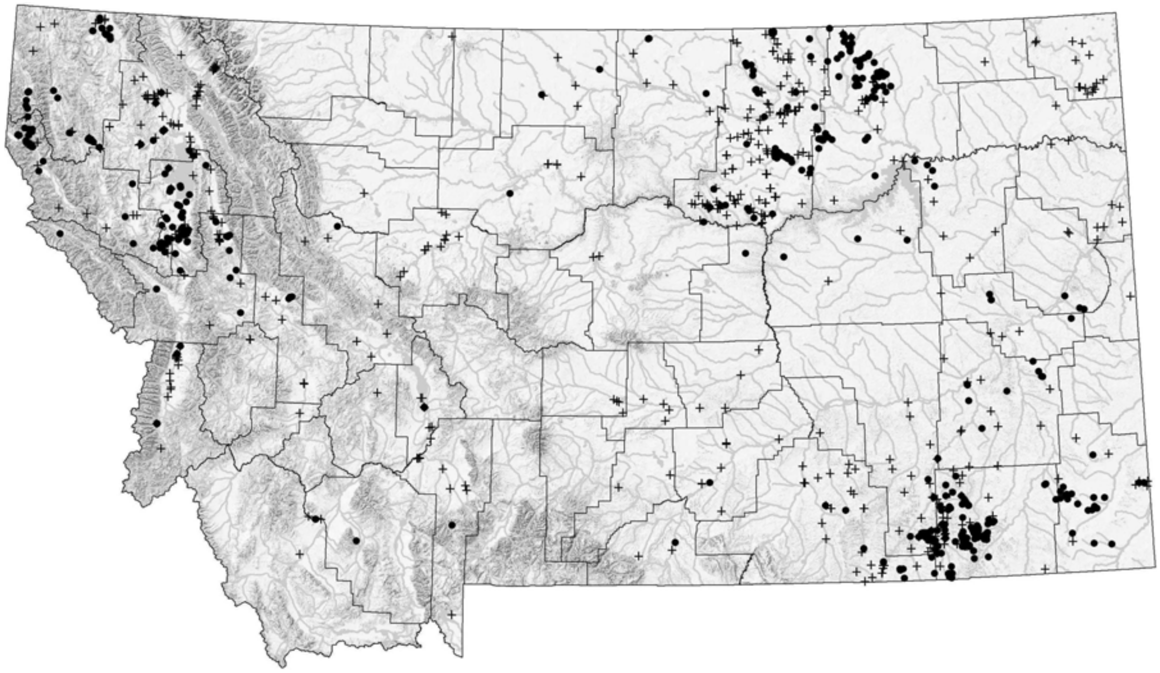

35%
30%
25%
20%
15%
10%
5%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 3,422; Min = 652 M; Max = 3,475 M
Detection: N = 811; Min = 652 M; Max = 2,956 M

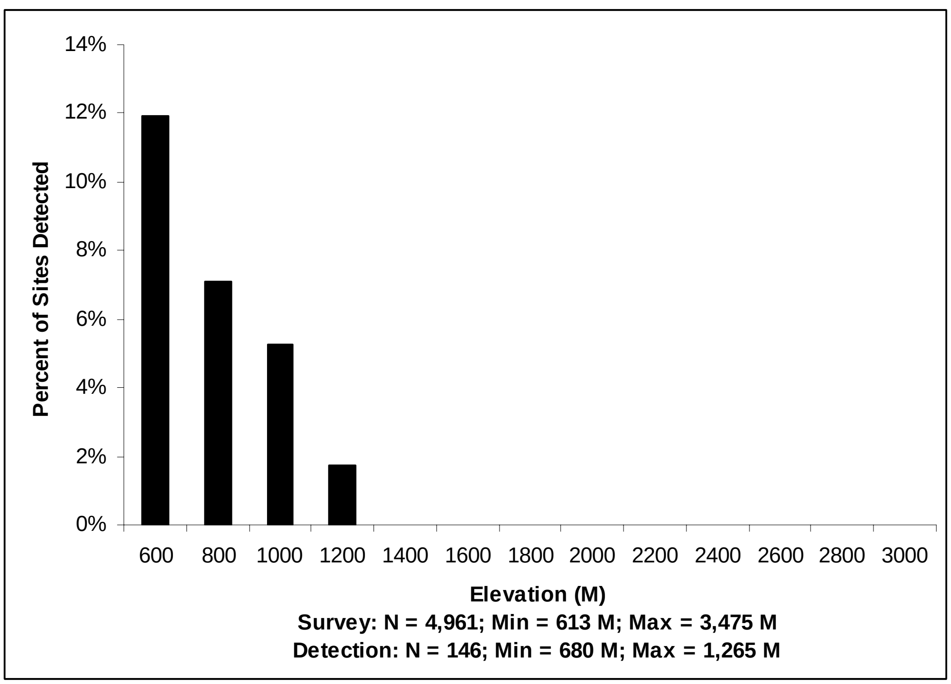
(j)



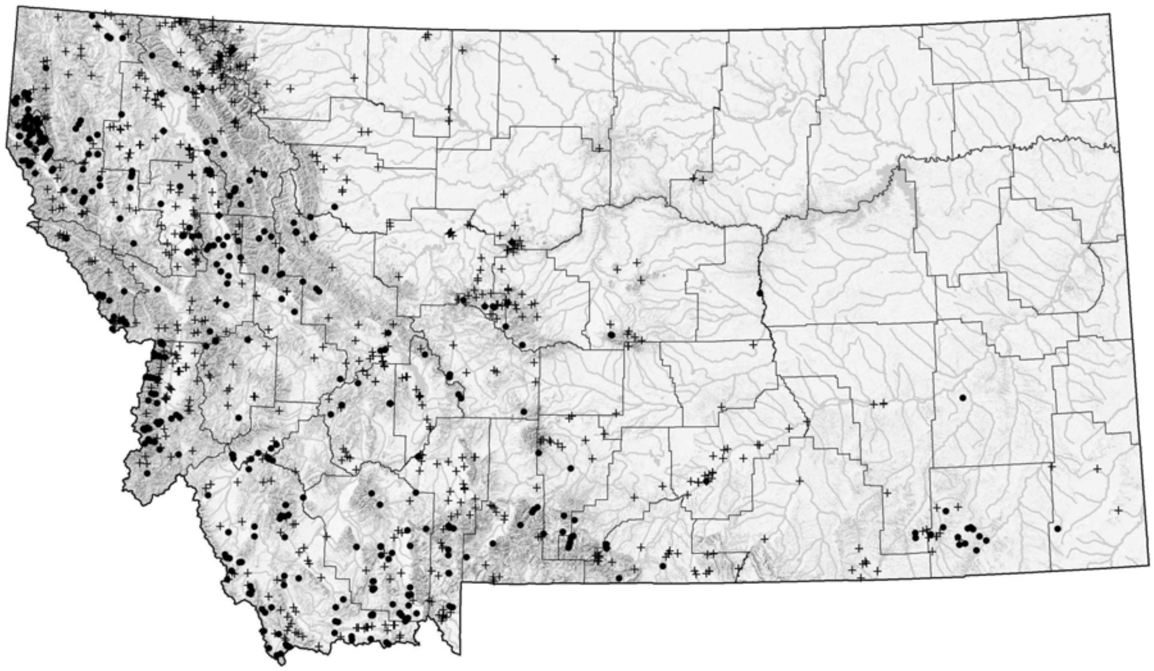

20%
18%
16%
14%
12%
10%
8%
6%
4%
2%
0%
Percent of Sites with Breeding
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 1,546; Min = 613 M; Max = 2,615 M
Detection: N = 205; Min = 680 M; Max = 1,199 M

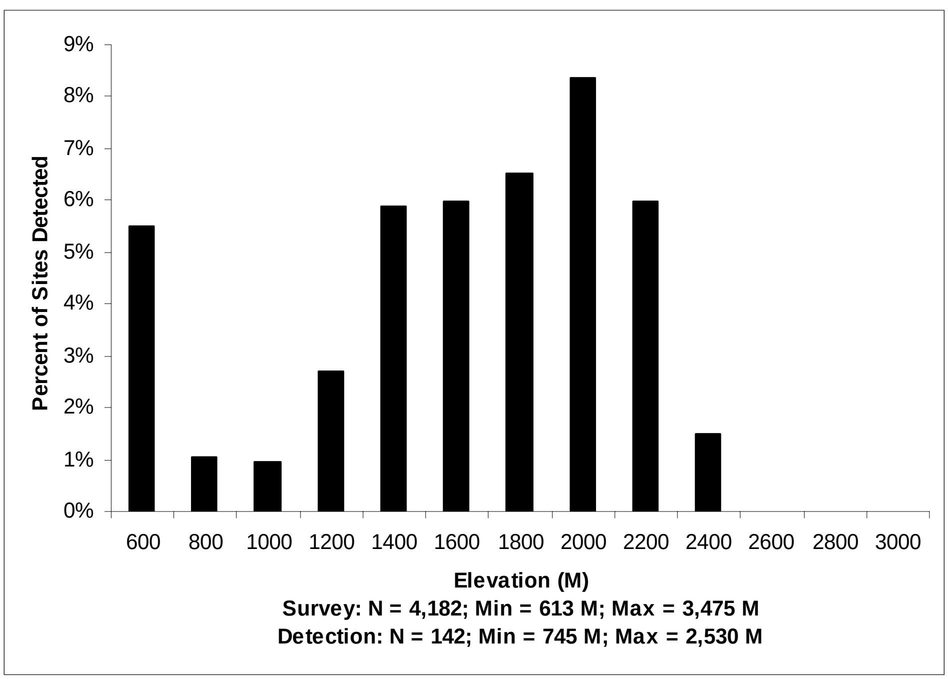
(k)




Percent of Sites Detected
14%
12%
10%
8%
6%
4%
2%
0%
600 800 1000 1200 1400 1600 1800 2000 2200 2400 2600 2800 3000
Elevation (M)
Survey: N = 4,961; Min = 613 M; Max = 3,475 M
Detection: N = 146; Min = 680 M; Max = 1,265 M

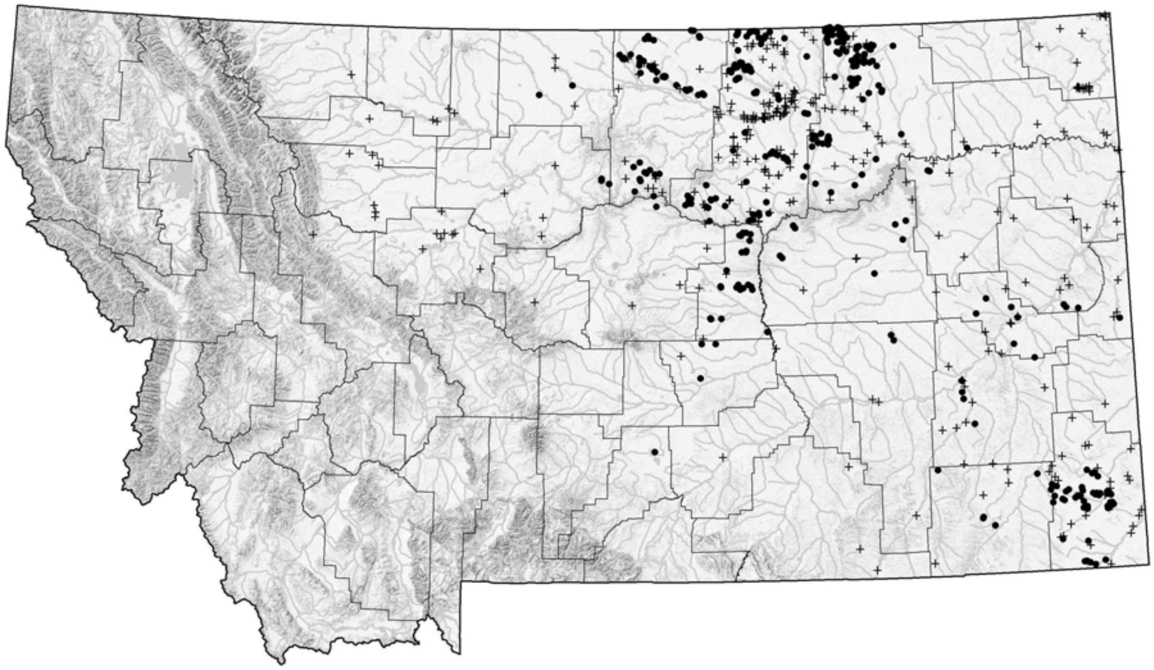
(l)

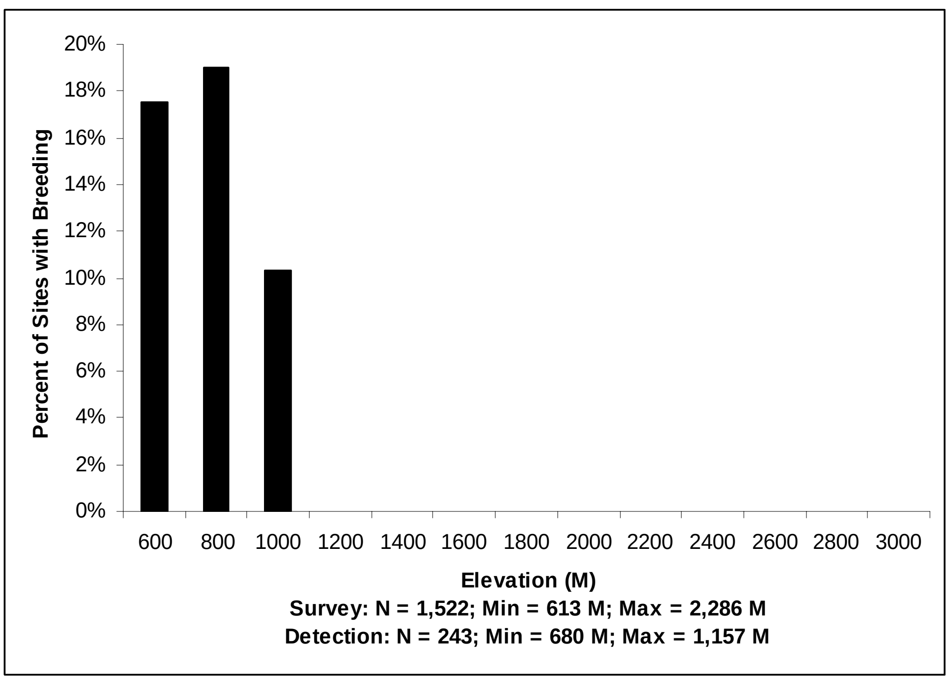




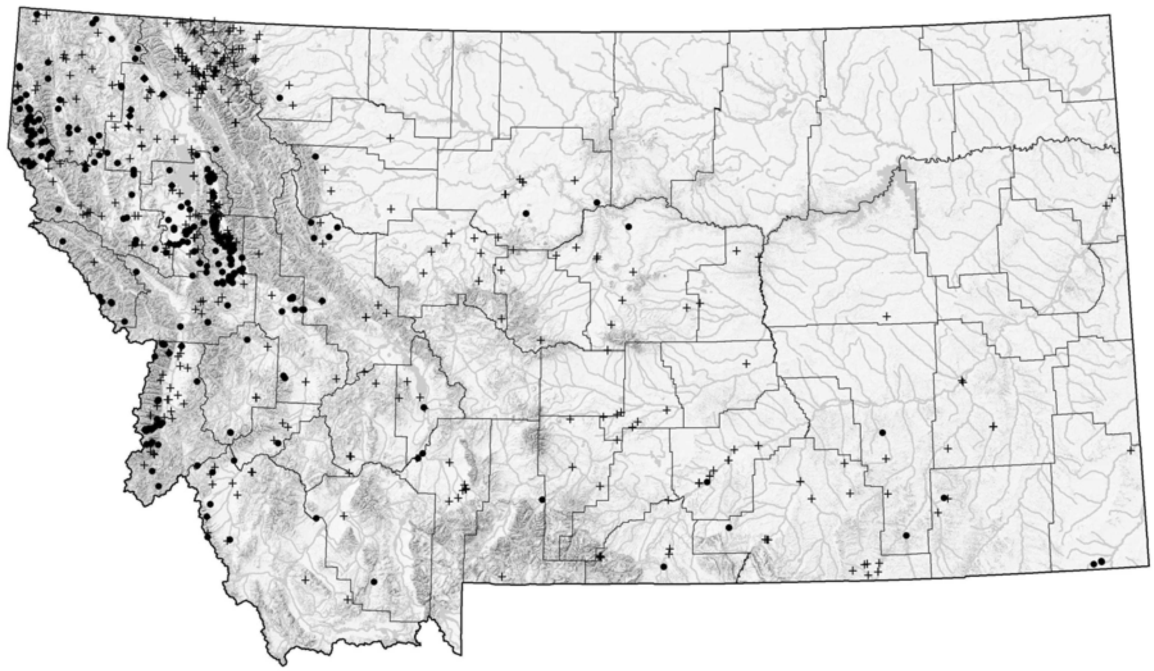
9%
8%
7%
6%
5%
4%
3%
2%
1%
0%
Percent of Sites Detected
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 4,182; Min = 613 M; Max = 3,475 M
Detection: N = 142; Min = 745 M; Max = 2,530 M

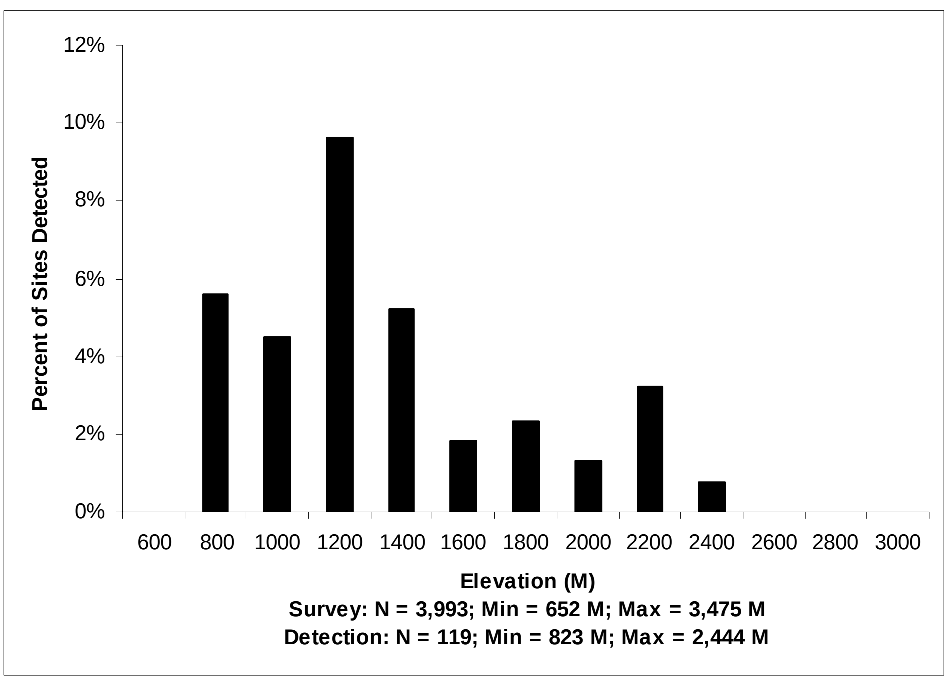
(m)




20%
18%
16%
14%
12%
10%
8%
6%
4%
2%
0%
Percent of Sites with Breeding
600 800 1000 1200 1400 1600 1800 2000 2200 2400 2600 2800 3000
Elevation (M)
Survey: N = 1,522; Min = 613 M; Max = 2,286 M
Detection: N = 243; Min = 680 M; Max = 1,157 M

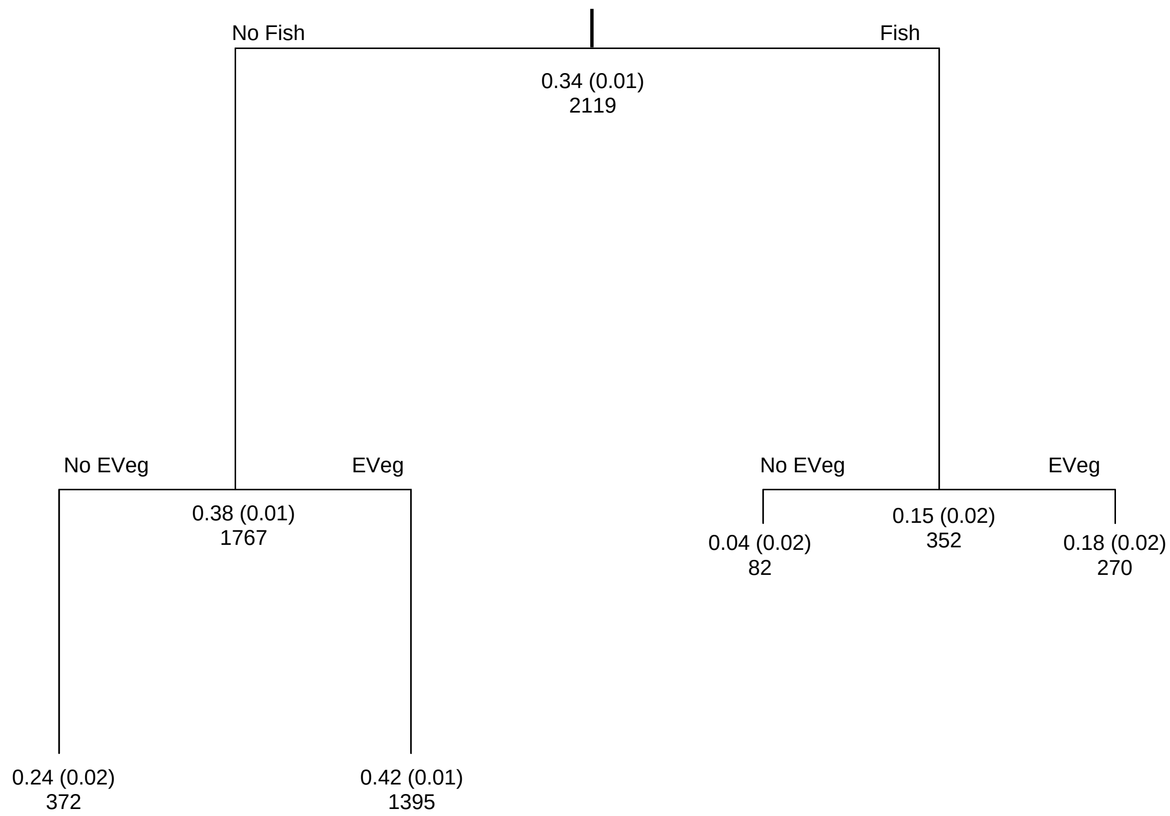
(n)





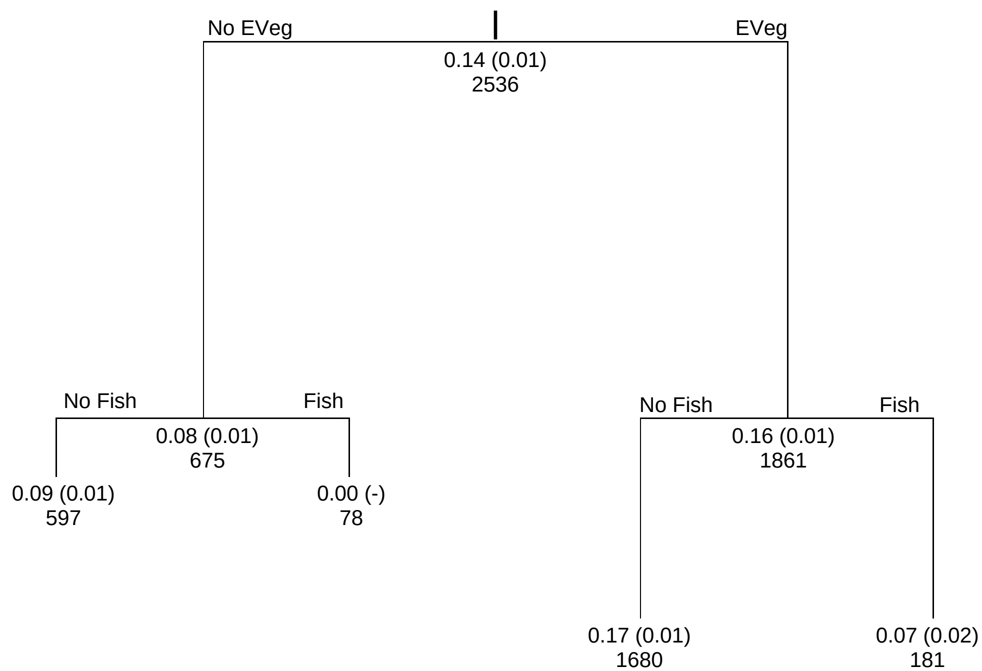
12%
10%
8%
6%
4%
2%
0%
Percent of Sites Detected
600
800
1000
1200
1400
1600
1800
2000
2200
2400
2600
2800
3000
Elevation (M)
Survey: N = 3,993; Min = 652 M; Max = 3,475 M
Detection: N = 119; Min = 823 M; Max = 2,444 M

(a)



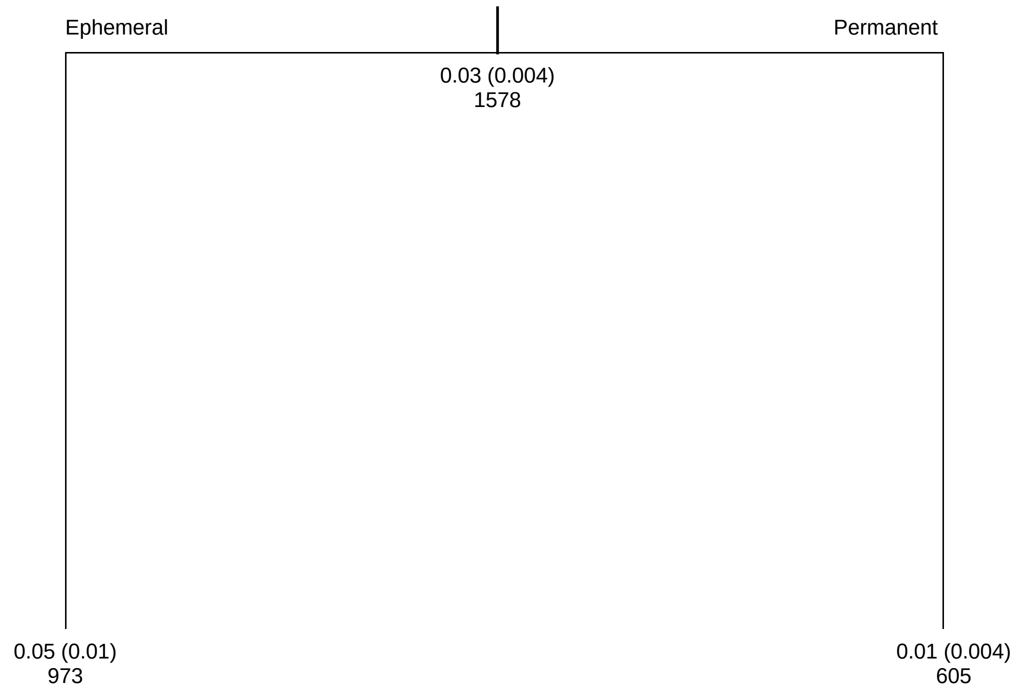
No Fish
Fish
0.34 (0.01)
2119
No EVeg
EVeg
0.38 (0.01)
1767
No EVeg
EVeg
0.15 (0.02)
352
0.04 (0.02)
82
0.18 (0.02)
270
0.24 (0.02)
372
0.42 (0.01)
1395

(b)

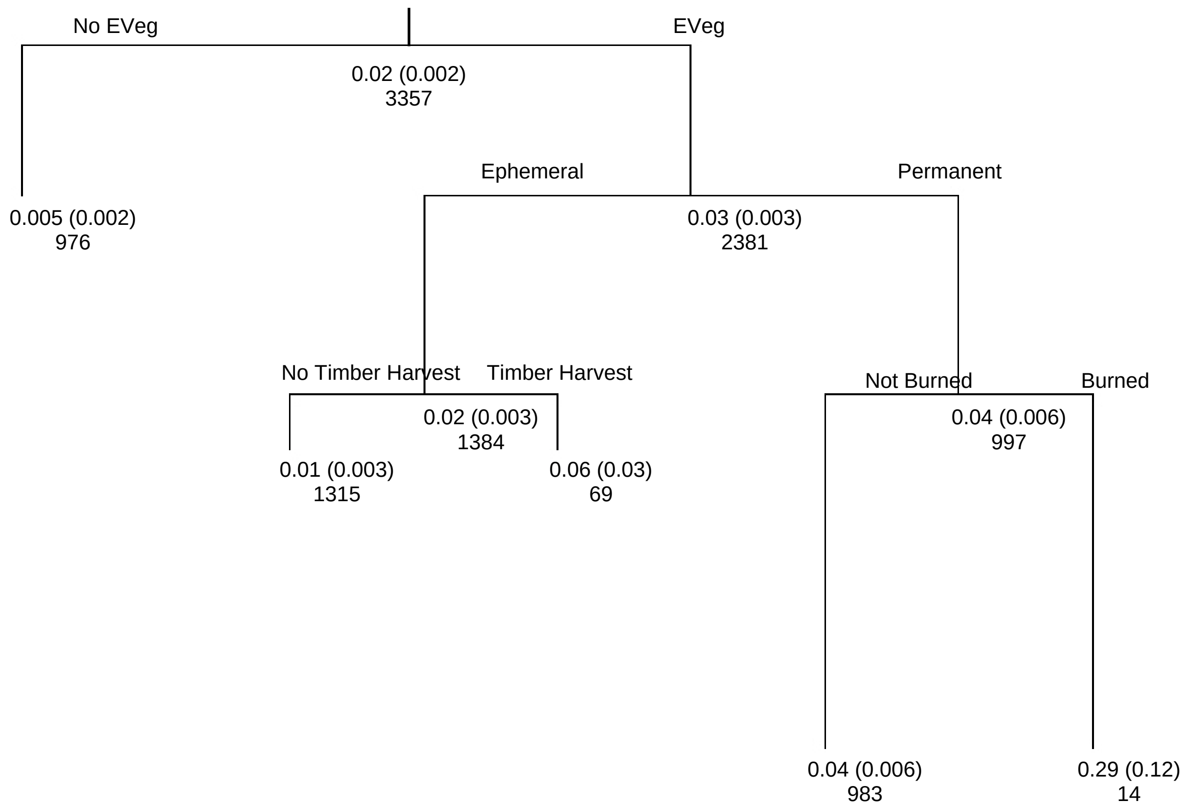


No EVeg
EVeg
0.14 (0.01)
2536
No Fish
Fish
0.08 (0.01)
675
0.09 (0.01)
597
0.00 (-)
78
No Fish
Fish
0.16 (0.01)
1861
0.17 (0.01)
1680
0.07 (0.02)
181

(c)

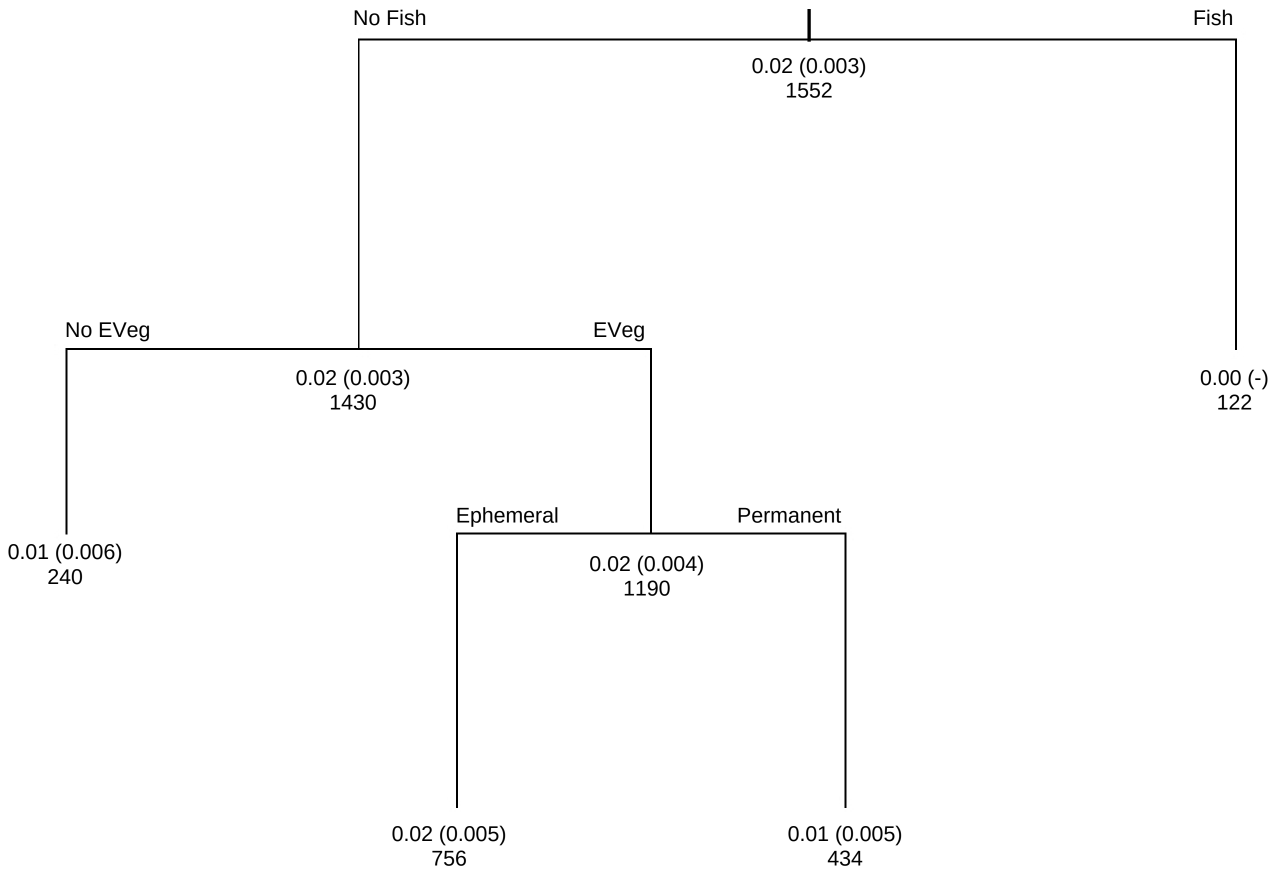


(d)



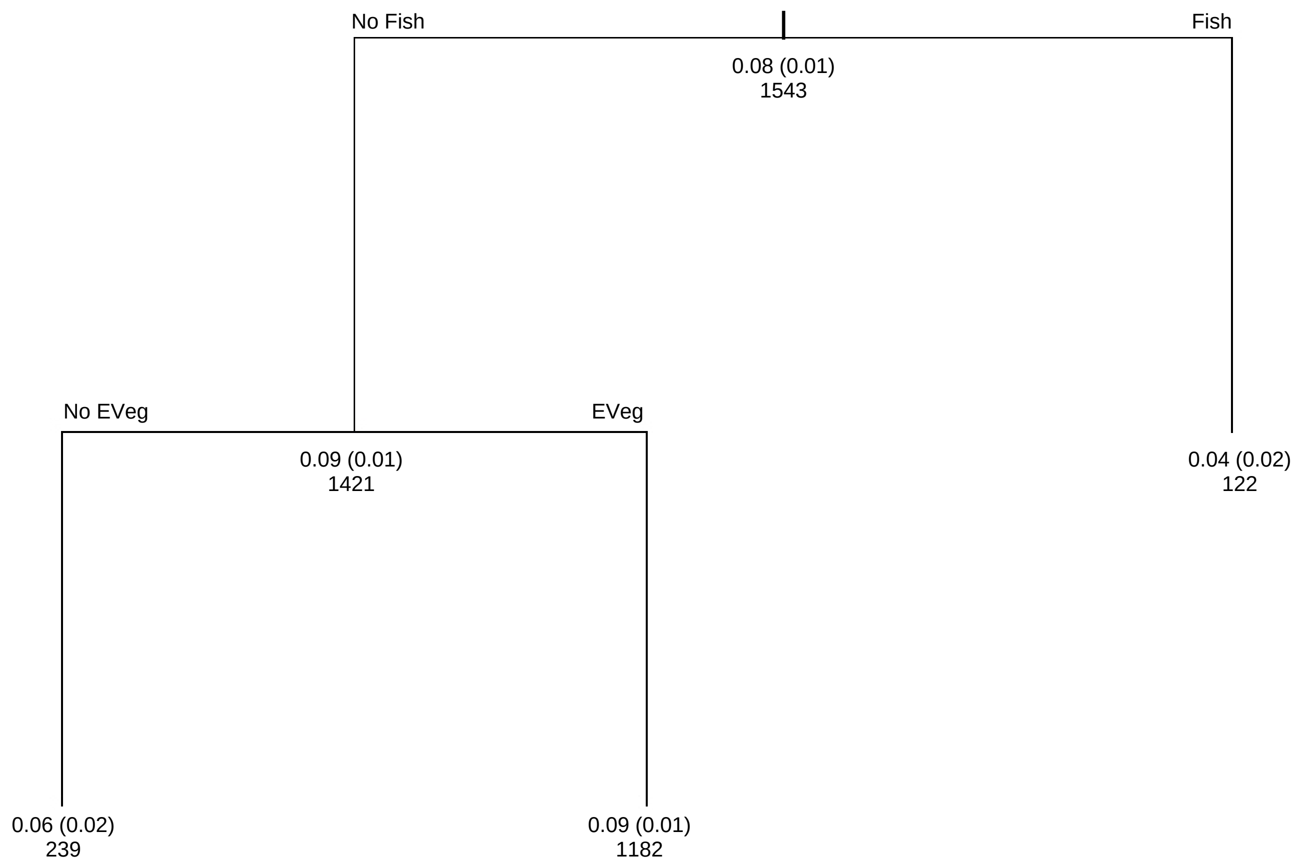
No EVeg
EVeg
0.02 (0.002)
3357
Ephemeral
Permanent
0.005 (0.002)
976
0.03 (0.003)
2381
No Timber Harvest
Timber Harvest
Not Burned
Burned
0.02 (0.003)
1384
0.04 (0.006)
997
0.01 (0.003)
1315
0.06 (0.03)
69
0.04 (0.006)
983
0.29 (0.12)
14

(e)

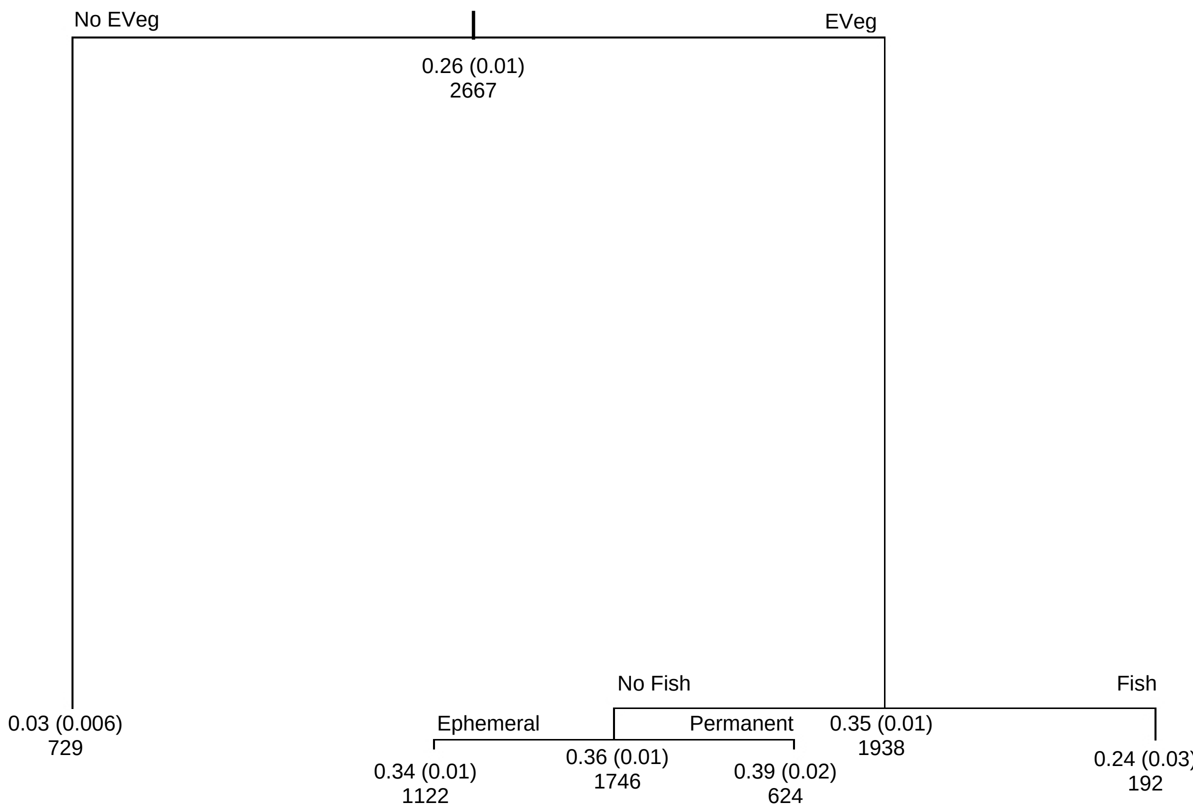


No Fish
Fish
0.02 (0.003)
1552
No EVeg
EVeg
0.02 (0.003)
1430
0.00 (-)
122
Ephemeral
Permanent
0.01 (0.006)
240
0.02 (0.004)
1190
0.02 (0.005)
756
0.01 (0.005)
434

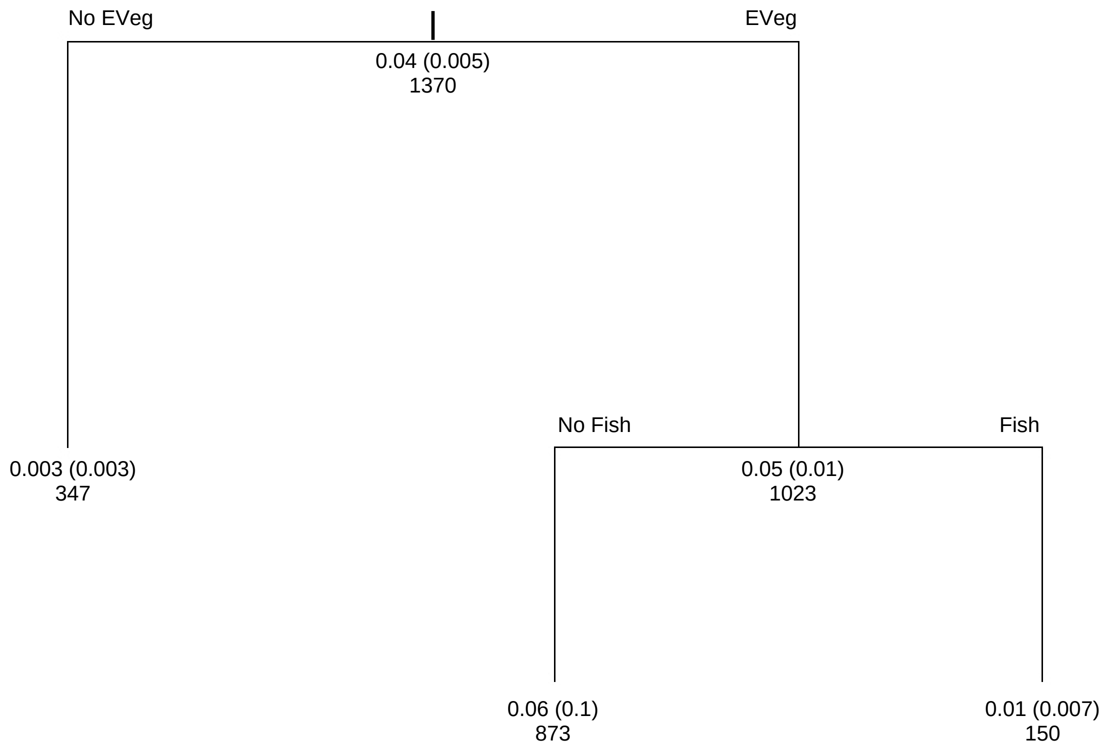
(f)


No Fish
Fish
0.08 (0.01)
1543
No EVeg
EVeg
0.09 (0.01)
1421
0.04 (0.02)
122
0.06 (0.02)
239
0.09 (0.01)
1182

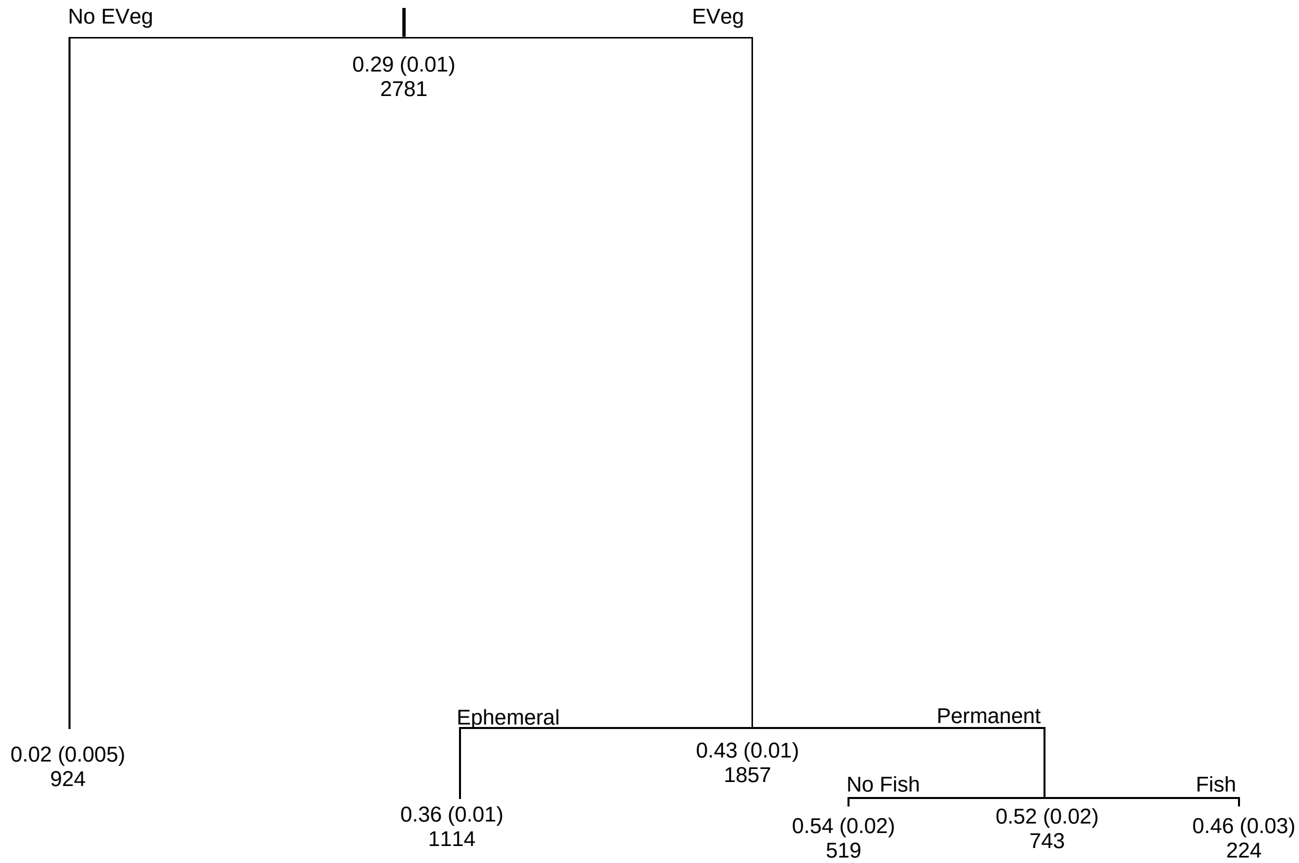
(g)


No EVeg
EVeg
0.26 (0.01)
2667
No Fish
Fish
0.03 (0.006)
729
Ephemeral
Permanent
0.35 (0.01)
1938
0.34 (0.01)
1122
0.36 (0.01)
1746
0.39 (0.02)
624
0.24 (0.03)
192

(h)

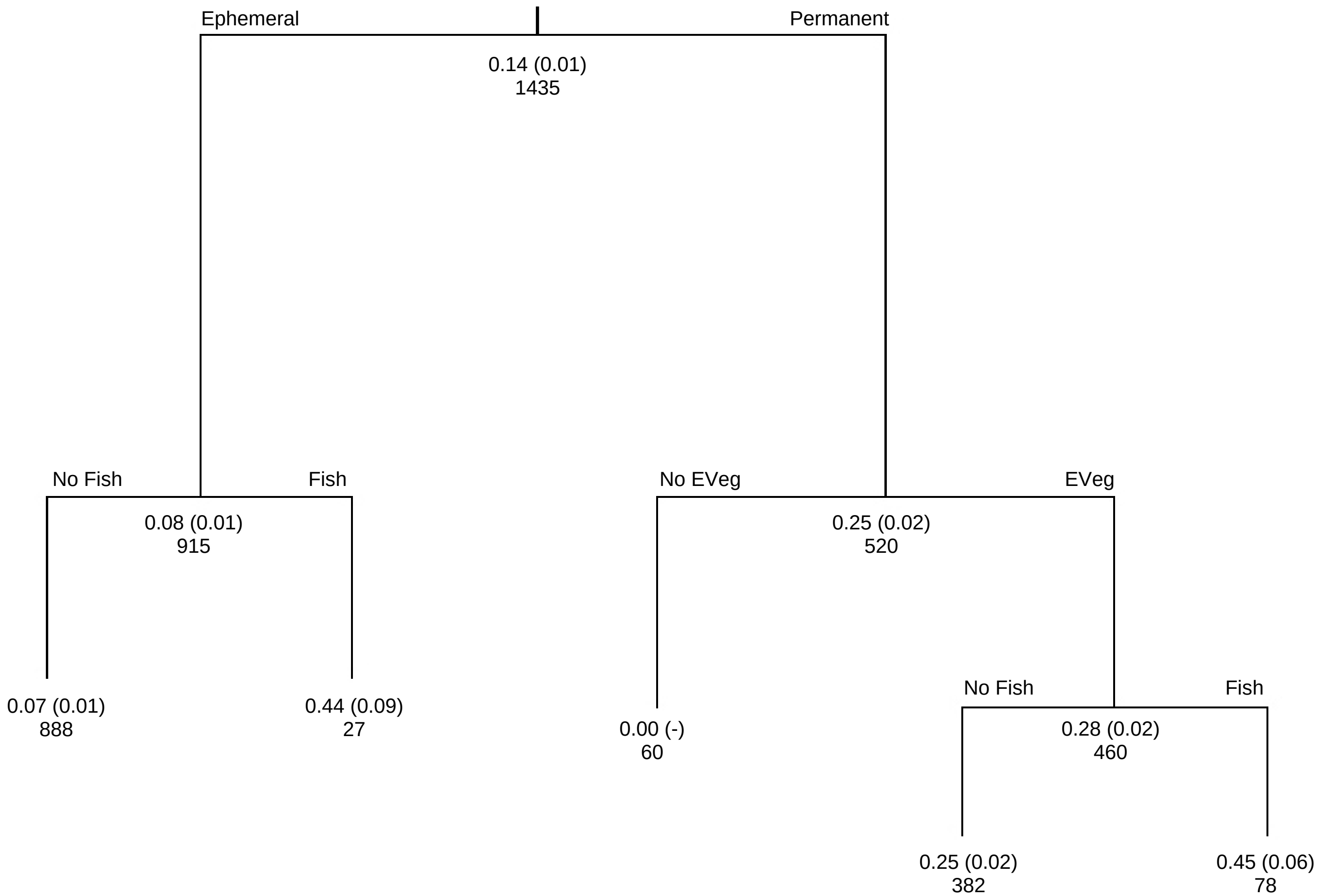

No EVeg
EVeg
0.04 (0.005)
1370
No Fish
Fish
0.003 (0.003)
347
0.05 (0.01)
1023
0.06 (0.1)
873
0.01 (0.007)
150

(i)

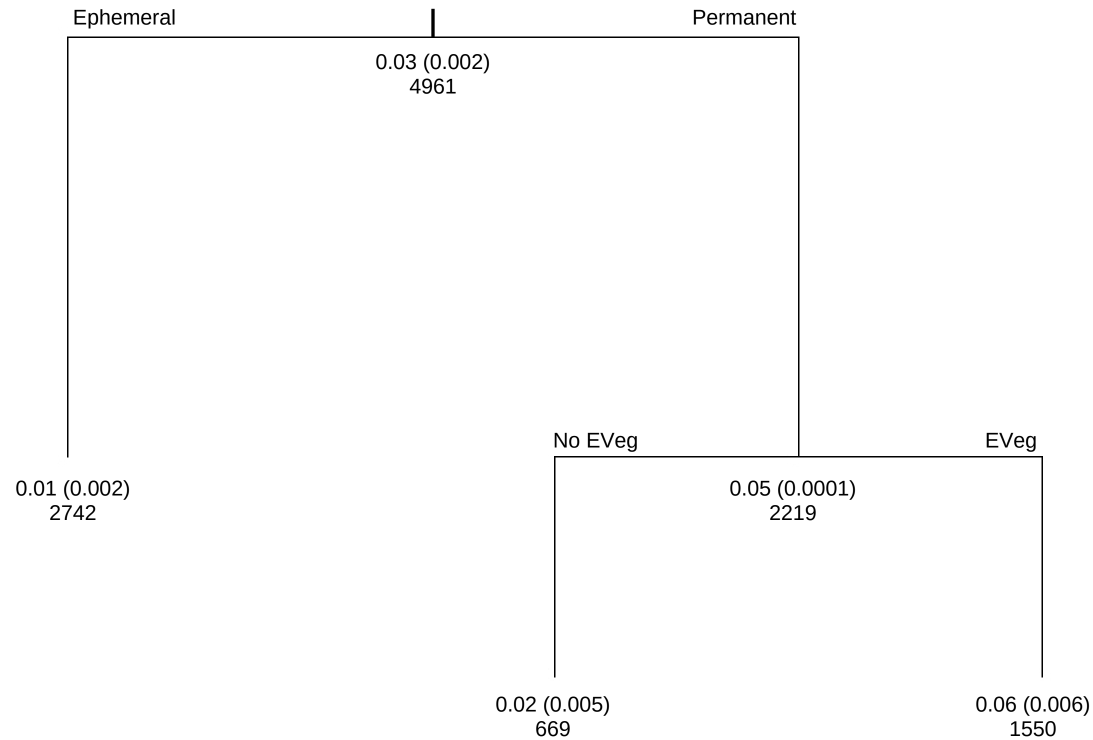


No EVeg
EVeg
0.29 (0.01)
2781
0.02 (0.005)
924
Ephemeral
Permanent
0.43 (0.01)
1857
0.36 (0.01)
1114
No Fish
Fish
0.54 (0.02)
519
0.52 (0.02)
743
0.46 (0.03)
224

(j)

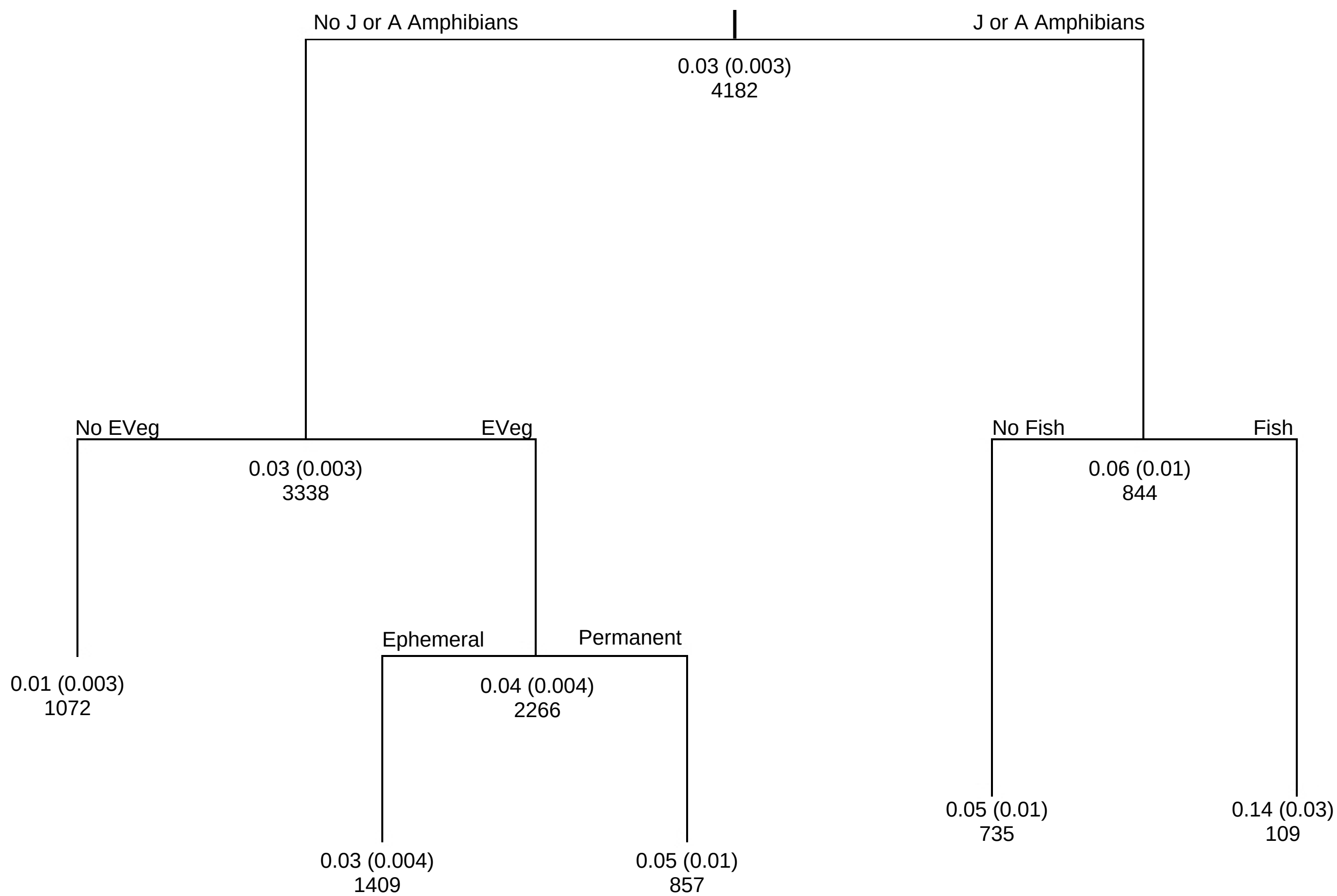

Ephemeral
Permanent
0.14 (0.01)
1435
No Fish
Fish
0.08 (0.01)
915
No EVeg
EVeg
0.25 (0.02)
520
0.07 (0.01)
888
0.44 (0.09)
27
0.00 (-)
60
No Fish
Fish
0.28 (0.02)
460
0.25 (0.02)
382
0.45 (0.06)
78

(k)



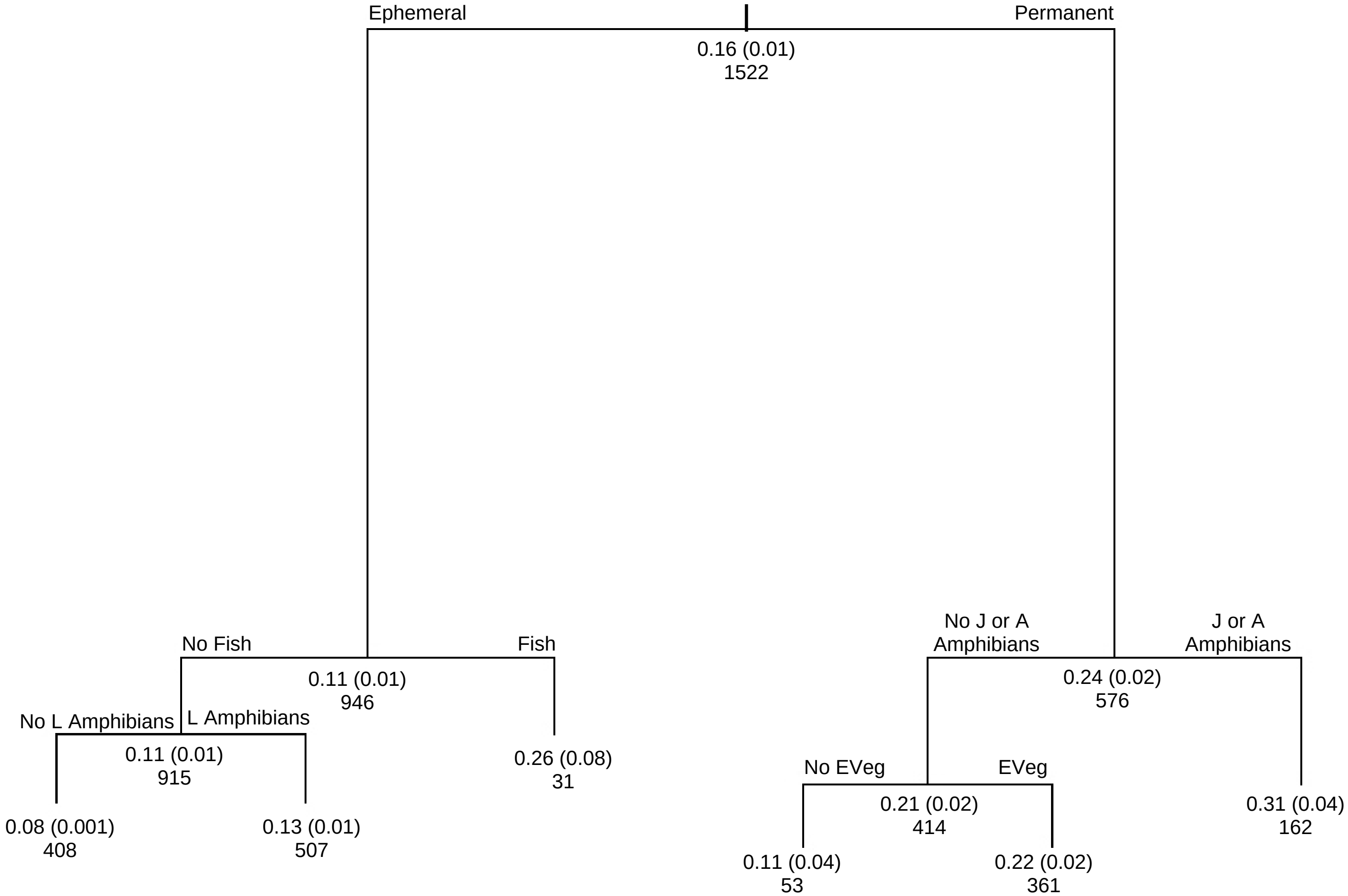
Ephemeral
Permanent
0.03 (0.002)
4961
No EVeg
EVeg
0.01 (0.002)
2742
0.05 (0.0001)
2219
0.02 (0.005)
669
0.06 (0.006)
1550

(l)



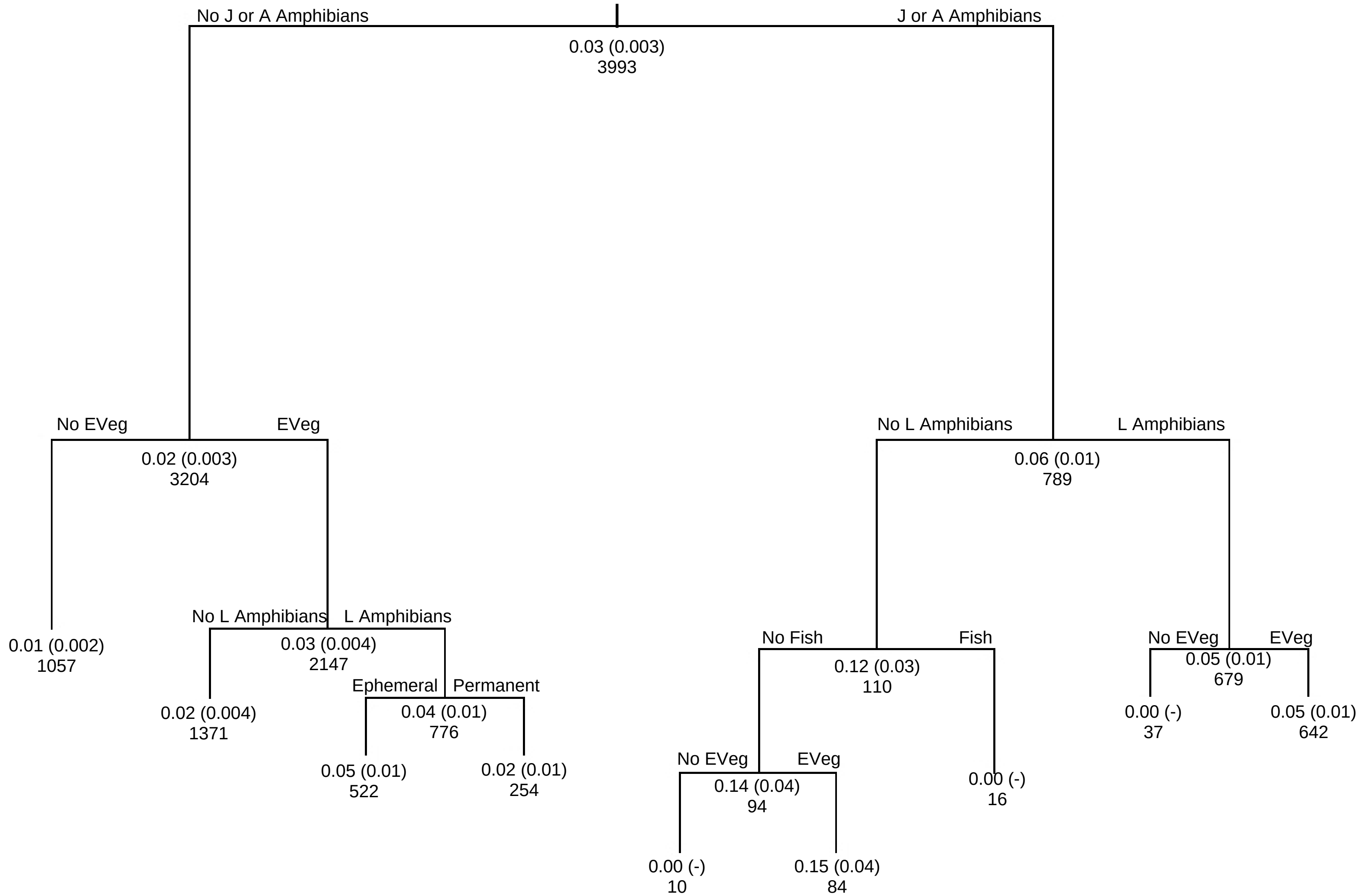
No J or A Amphibians
J or A Amphibians
0.03 (0.003)
4182
No EVeg
EVeg
0.03 (0.003)
3338
No Fish
Fish
0.06 (0.01)
844
Ephemeral
Permanent
0.04 (0.004)
2266
0.01 (0.003)
1072
0.03 (0.004)
1409
0.05 (0.01)
857
0.05 (0.01)
735
0.14 (0.03)
109

(m)



Ephemeral
Permanent
0.16 (0.01)
1522
No Fish
Fish
0.11 (0.01)
946
No L Amphibians
L Amphibians
0.11 (0.01)
915
0.08 (0.001)
408
0.13 (0.01)
507
0.26 (0.08)
31
No J or A Amphibians
J or A Amphibians
0.24 (0.02)
576
No EVeg
EVeg
0.21 (0.02)
414
0.11 (0.04)
53
0.22 (0.02)
361
0.31 (0.04)
162

(n)



No J or A Amphibians
J or A Amphibians
0.03 (0.003)
3993
No EVeg
EVeg
0.02 (0.003)
3204
No L Amphibians
L Amphibians
0.06 (0.01)
789
0.01 (0.002)
1057
No L Amphibians
L Amphibians
0.03 (0.004)
2147
0.02 (0.004)
1371
Ephemeral
Permanent
0.04 (0.01)
776
0.05 (0.01)
522
0.02 (0.01)
254
No Fish
Fish
0.12 (0.03)
110
No EVeg
EVeg
0.14 (0.04)
94
0.00 (-)
10
0.15 (0.04)
84
0.00 (-)
16
No EVeg
EVeg
0.05 (0.01)
679
0.00 (-)
37
0.05 (0.01)
642

CHAPTER 3

PREDICTED DISTRIBUTION AND HABITAT SUITABILITY MODELS FOR AMPHIBIANS AND REPTILES IN MONTANA

Abstract

Models predicting spatial distribution and habitat suitability are critical for natural resource managers who often need to make decisions that impact species for which there is limited information. I used presence-only data in conjunction with pseudo-absences in program Maxent to model distribution and habitat suitability for 31 species of amphibians and reptiles in Montana to inform management and conservation efforts. My primary goals were to: (1) identify variables that limit species' distributions; (2) identify areas in need of field surveys; (3) create lists of predicted species within administrative boundaries at the regional (>10,000 km^2), landscape (township or 100 km^2), and large local habitat patch (>16 Ha) scales; and (4) identify marginal, suitable, and optimal habitat classes for species at various spatial scales. Models identified scale dependent responses to environmental variables, opportunities to extend the known ranges of species, areas that support potentially isolated populations in need of conservation efforts, areas that are critical for maintaining landscape connectivity, areas that may provide the best habitat for reintroduction of species that have declined, and areas where exotic and nonindigenous species are most likely to become established. Continuous models performed well as evaluated by the area under the receiver operating characteristic plot (0.858-0.940). Binary outputs using a threshold that balances the predicted area against omission and commission error rates performed well as evaluated by absolute validation index (AVI) or percentage of presences above the threshold (0.89-0.98). When compared to predictions from the deductively based models produced by the Montana Gap Analysis Project, continuous Maxent models offered more realistic depictions of amphibian and reptile species distributions when survey data was available for a region and in most cases reduced predicted area while simultaneously increasing predictive accuracy. However, deductive models like those produced by GAP are still important for representing some species distributions in areas lacking survey effort.

Introduction

Increasing threats to biological diversity have stimulated efforts to model geographic distribution and habitat suitability for a variety of species in the last two decades. These efforts have been made possible by advances in computer processing speeds, availability and ease of use of geographic information systems (GIS), including regional environmental data, and software employing a variety of modeling algorithms (Turner et al. 2001, Scott et al. 2002). Most efforts use a variety of statistical and nonstatistical models to attempt to identify and represent Hutchinson's realized n-dimensional niche under the implicit assumption that environmental features in the locations where species are recorded represent suitable conditions in all environmental dimensions examined and that at least some environmental features are limiting to the species at locations where they are not detected (Andrewartha and Birch 1954, Hutchinson 1957, Heglund 2002). Thus, many standard approaches to modeling (e.g., general linear models, ordination, logistic regression) require presence and absence data for training and testing models. However, true absence data are very rarely available because even most formal surveys following a structured set of protocols fail to assess and correct for imperfect detectability (Kéry 2002, MacKenzie et al. 2002, MacKenzie et al. 2006). Ideally data used to construct predicted distribution and habitat suitability models would involve surveys of local habitat patches on multiple occasions to directly incorporate detectability into the model (MacKenzie 2006). Future surveys should strive to gather data in this manner, but in the mean time, methods that can make use of the large volumes of presence-only data available from past studies and aggregated in central data centers (e.g., zoological museums or Natural Heritage Programs) can be used to inform management decisions and identify and test hypotheses regarding species' distributions and habitat suitability at various spatial scales.

Montana's amphibian and reptile species (Table 3.1) are good candidates for a modeling effort because many of their distributions are poorly understood, many areas of the state lack occurrence records, and a high percentage are classified as Species of Concern (Maxell et al. 2003; Werner et al. 2004; MNHP and MFWP 2009). In Montana, the only models of predicted habitat available for amphibians and reptiles are deductively based models with simple rules for associations to mapped habitat created through expert opinion by the Montana Gap Analysis Project (GAP) (Scott et al. 1993, Hart et al. 1998, Redmond et al. 1998). These models are general and only appropriately applied at larger scales (e.g., > 100 km^2) so are best applied to regional level planning efforts. Thus, there is a need for inductive models based on presence-only data that are applicable at landscape (100 km^2) and larger habitat patch (>16 Ha) scales.

A number of recently-developed modeling approaches are capable of making use of presence-only data either by itself or in conjunction with background sampling of the environmental variables, also termed pseudo-absences. These include, but are not limited to, envelope models (e.g., BIOCLIM), ecological niche factor analysis (e.g., BIOMAPPER), rule sets derived with genetic algorithms (e.g., GARP), multivariate distances (e.g., DOMAIN and LIVES), and maximum entropy (e.g., Maxent) (see review by Elith et al. 2006). Many of these new methods are capable of fitting more complex models from smaller datasets, by using explicit mechanisms to prevent model complexity from increasing beyond what is supported by the data. The maximum entropy algorithm applied in the software package Maxent has recently been evaluated as equivalent or superior to other recently developed presence-only or presence/absence methods and superior to older presence/absence methods at predicting several hundred species distributions in many regions of the world (Phillips et al. 2004, 2006, Elith et al. 2006, Hernandez et al. 2008, Phillips and Dudik 2008). Maxent readily accepts ASCII grid layers exported from a GIS and presence records in comma-separated value format to produce predicted distribution grids easily imported into a GIS as well as a variety of outputs that allow for interpretation of the predictive performance of models and importance of individual environmental variables to explaining a species distribution (Phillips et al. 2004, Phillips and Dudik 2008).

I used the maximum entropy algorithm in Maxent version 3.2.19 (Phillips et al. 2004, 2006, Phillips and Dudik 2008) to create predicted distribution and relative habitat suitability models at the state-wide and range-wide scale for 14 amphibian and 17 reptile species known to occur in Montana (Table 3.1). My primary goals were to produce: (1) continuous state-wide maps depicting relative habitat suitability to identify variables that limit species' distributions and areas both within and outside of the known ranges of species that should be targeted for field survey; (2) conservative binary maps depicting suitable and non suitable habitat with low omission error rates that can be used to create checklists of predicted species within administrative boundaries at the regional (>10,000 km^2), landscape (township or 100 km^2), and large local habitat patch (Public Land Survey System (PLSS) section or smaller (e.g., >16 Ha)) scales; and (3) where possible, map outputs depicting unsuitable, marginal, suitable, and optimal habitat classes at various spatial scales to inform field surveys and management decisions.

Methods

Environmental Variables

Environmental input layers consisted of 10 continuous and 5 categorical variables I felt were biologically relevant to determining the distribution of species and for which I could obtain or develop state-wide 90 x 90 m grid cell coverages for Montana (Table 3.2). Metadata for each environmental layer is provided in Appendix C. Additional variables may be important in predicting distribution and habitat suitability (e.g., distribution of competitors or predators, Connell 1961), but I limited variables to state-wide input layers that could be easily obtained or developed for all species. Environmental input values are generally easily interpretable. However, the STATSGO soils layer (Table 3.2, Appendix C) consists of soil map units (≥ 625 ha) that are only individually interpretable as having similar general characteristics relative to surrounding areas. That is, due to the underlying complexities of the data associated with these map units, I was not able to make simple state-wide layers of soil characteristics (e.g., dominant soil type, depth, texture). Despite my inability to easily interpret responses to this layer, I chose to use it because: (1) many of Montana's amphibians and reptiles are either fossorial or depend on the water holding capacities of different soil types; (2) preliminary modeling indicated that inclusion of the layer resulted in better representations of species' distributions at the

PLSS section scale or smaller; and (3) I wanted to examine model output for patterns of the underlying soil characteristics species are associated with to inform development of state-wide soils layers for future modeling.

I created two sets of environmental layers to model each species' distribution and habitat suitability at two spatial extents: (1) statewide to identify variables that limit species' distributions and areas that should be targeted for field survey and (2) the known current range of each species to identify scale dependent factors limiting the species' range and examine and avoid potential biases in measures evaluating model performance (Lobo et al. 2008, Phillips and Dudik 2008).

Species Occurrences

I obtained 25,873 records housed in the Montana Natural Heritage Program's state-wide Point Observation Database for Montana's 14 amphibian and 17 reptile species (Table 3.3). These records came from a variety of sources, including voucher specimens in museums across the country, observations and voucher specimens from state-wide inventory and monitoring programs, observation and voucher specimen records found in the scientific literature, and records gathered by agency biologists and the general public (e.g., Chapter 2, Maxell et al. 2003). The vast majority of these records were gathered in the last 10 years, but some records date to the time of the Lewis and Clark expedition in 1805 (Maxell et al. 2003, Moulton 1983). I limited records used in training and testing the models to those within the known range of the species (Werner et al. 2004) and only used records with a spatial precision of less than 400 m to match the grain of records and environmental grid cells as closely as possible while not overly restricting the number of records available for modeling. I further limited the number of records by eliminating duplicate records for individual 90 x 90 m grid cells. These restrictions resulted in a 50 percent reduction in the total number of records available for training and testing models (Table 3.3). Because the number of records available for many species was limited, I used all occurrences irrespective of their date. Thus, there is some potential for introduction of noise from records gathered in time periods before those encompassed by the environmental data layers, some of which have undergone rapid change in recent decades (e.g., land cover types (Redmond et al. 1998)). However, I feel that this is a minor issue because there were relatively few older records (Maxell et al. 2003) and the majority were eliminated due to poor spatial precision.

Model Building

For species with >125 occurrence records, I selected a spatially balanced random subset of approximately 25% of the occurrences and set it aside to test model output (Table 3.3). For these species, the remaining 75% of the occurrences were used to train models. For species with <125 occurrences (Table 3.3), all occurrences were used to train models and model output was not evaluated with test data.

I used the maximum entropy algorithm employed in Maxent version 3.2.19 to create predicted distribution models (Phillips and Dudik 2008). Maxent first generates empirical distributions for all environmental variables over all pixels that are associated with species occurrences. These environmental "features" are then used to constrain the estimated distributions computed over the background pixels (pseudo-absences) of the environmental data layers so that estimated distributions match characteristics of the empirical distributions while otherwise maintaining the maximum entropy (most uniform distributions) possible (Jaynes 1957, Phillips et al. 2004, 2006, Phillips and Dudik 2008). To do this, Maxent makes use of the Gibbs distribution, which maximizes the product of the probabilities of the sample locations and takes the form:

$$P(x) = \exp(c1 * f1(x) + c2 * f2(x) + c3 * f3(x)\ldots) / Z$$

where c1, c2, c3, … are weighted constants, f1, f2, f3, … are the environmental features constrained to the characteristics of the empirical distributions, and Z is a scaling constant that ensures that P sums to 1 over all grid cells (Phillips et al. 2004, 2006). The manner in which the estimated distributions are constrained typically depends on the number of species occurrences available. Estimated environmental distributions can be constrained to match the empirical average (linear feature), average and variance (quadratic feature), covariance (product feature), proportional occurrence (threshold feature), or average below and constant above a certain point (hinge feature) (Phillips and Dudik 2008). I used default settings for features so that all were employed when there were at least 80 training samples, linear, quadratic and hinge features were employed when there were 15 to 79 samples, linear and quadratic features were used when there were 10 to 14 samples, and only linear features were used when there were fewer than 10 samples. Categorical environmental variables have a discrete feature for every possible value of the variable so that the estimated distributions have the same proportional representation of each categorical value. In practice Maxent avoids overfitting of models to the training data by "regularizing" or relaxing the features so that estimated distributions only have to be close to the empirical distributions rather than exactly equal to them. If there is evidence that a model is overfit, a regularization parameter can be adjusted in Maxent to delineate just how close a fit is needed between empirical features and estimated distributions. I used the default regularization value of 1 for all model runs.

Maxent is a machine learning based approach that begins with a uniform Gibbs distribution (the distribution with maximum entropy) and successively modifies each weighted constant on each iteration until either the change in the "gain" (the log of the number of grid cells minus the average of the negative log probabilities of the sample locations) falls below a set threshold or a set maximum number of iterations are performed. The gain value at the end of a model run indicates the likelihood of suitability of the presence samples relative to the likelihood for random background points (Maxent 2008). For example, a gain of 3 indicates the average likelihood of the presence samples is $e^3 = 20$ times higher than that of a random background point and the model will be relatively concentrated around the occurrence locations. The overall gain associated with individual environmental variables can be used as a measure of the importance of each variable. The Maxent algorithm is guaranteed to converge to values of c1, c2, c3, … that give the unique optimum estimated distribution P, and, therefore, the outputs are deterministic (Phillips et al. 2004, 2006, Phillips and Dudik 2008). Maxent can use every grid cell that has values for all the environmental variables to calculate the estimated distribution. However, because there are a large number of 90-meter grid cells for Montana (10,204 columns and 5,892 rows), and the modeling performance of Maxent does not improve significantly with very large numbers of pixels (Phillips and Dudik 2008), 0.1 percent of background pixels (equivalent to 60,000 pixels for state-wide models and proportionally smaller numbers for range-wide models) were used in estimated distributions to represent the variety of environmental conditions present in the data.

Maxent produces three grid cell based primary map outputs from the exponential function: (1) a raw probability value that is scale dependent and sums to 1 across all grid cells; (2) a scale independent cumulative output that ranges from 0 to 1, indicates the percentage of the distribution with raw values less than the cell being considered, and represents the omission rate at that threshold; and (3) a logistic output ranging from 0 to 1 for individual grid cells that results from placing the raw output values into an exponential function along with the maximum entropy probability function (Phillips and Dudik 2008). The logistic output has been shown to slightly out perform the raw and cumulative outputs and under some sampling effort restrictions the logistic output theoretically indicates the probability of the species presence in a grid cell (Phillips and Dudik 2008). However, because most presence-only data is gathered under a variety of levels of sampling effort, it is probably best to interpret this as relative suitability of habitat. I converted ASCII logistic grid outputs into raster grids in ArcMap 9.2 and symbolized outputs in 3 ways (Appendix D). First, I used a stretched color ramp of continuous state-wide output with warm colors (reds) indicating areas with more suitable habitat and cooler areas (blues) indicating areas with less suitable habitat. Second, for the continuous range-wide models I used a low cutoff threshold identified by Maxent as balancing training omissions against the fraction of predicted area and the cumulative threshold value to create a binary map of predicted suitable and unsuitable habitat (Maxent 2008). To this map I added training and test occurrences to visually examine model fit. Third, for species that had test data available (Table 3.3), I coded the continuous range-wide logistic output into habitat suitability classes representing not suitable, low suitability, moderate suitability, and optimal suitability. To create these habitat suitability classes I classified the logistic values into 10 equal-interval classes in

ArcMap 9.2 and plotted the proportion of the test observations in each class over the proportion of the total grid cells in each class. Changes in the slope of the resulting curve and comparison with a curve from a random model allowed me to identify threshold values defining these habitat suitability classes; the low binary cutoff threshold was again used to delineate the boundary between suitable and unsuitable habitat (Table 3.4, Figure 3.1, Hirzel et al. 2006).

Model Evaluation

I used receiver operating characteristic (ROC) curves to evaluate performance of both state-wide and range-wide continuous models. ROC curves are an extension of the confusion matrix used to evaluate binary models (e.g., Figure 3.2). They plot sensitivity (or the true positive rate) against 1 - specificity (the false positive rate) across all thresholds of a continuous model. Because they provide information across thresholds, they avoid the subjectivity of choosing a single threshold and the total area under the ROC curve (AUC) provides a single measure of the overall performance of the model, namely the probability that the model will correctly order any randomly chosen pair of positive and negative values at any given threshold (Hirzel et al 2006). AUC values vary from 0 to 1 with a random model performing at a value of 0.5. However, in instances such as this analysis where random background points are used as pseudo-absences, AUC should be interpreted as the probability that a randomly chosen presence site is ranked above a randomly chosen background site (Phillips and Dudik 2008). Part of my reason for creating state-wide and range-wide models was to examine potential biases in AUC resulting from models applied inappropriately at the extents of administrative or regional boundaries rather than the boundaries of species' known ranges. If the extent of the modeled area is significantly larger than the species' range, AUC would be expected to be biased high because many background points will fall outside the known range of the species (Fielding and Bell 1997, Chefaoui and Lobo 2008). Thus, I examined AUC values for training data and test data when available (Table 3.3) on both state-wide and range-wide continuous models.

I evaluated low binary cutoff thresholds of range-wide continuous models (Table 3.4) and Montana Gap Analysis Project (GAP) models (Hart et al. 1998) with test occurrences using the absolute validation index (AVI) or the proportion of presence evaluation points falling above the threshold or within the GAP predicted distribution (Hirzel et al. 2006). I also evaluated these binary outputs with the point biserial correlation calculated as Pearson's correlation coefficient (COR) and average deviance (Phillips and Dudik 2008). While AUC is rank based, COR measures how well calibrated the prediction values are to the translated predicted presence or absence (1 or 0, respectively). COR varies from -1 to 1 with 0 representing a random model. Like COR, deviance measures how well calibrated prediction values are, but in addition, it penalizes errors in scaling of prediction values. For positive occurrences deviance is calculated as minus 2 times the natural log of the associated logistic output value (Phillips and Dudik 2008). Deviance varies from 0, when presence samples are associated with a logistic value of 1, to around 13.8, when logistic values approach 0.001.

For lentic breeding amphibian and aquatic reptile species recently evaluated with a state-wide inventory of lentic water bodies (Chapter 2), I used the presence and non-detection data to evaluate range-wide low binary cutoff thresholds and GAP predicted distributions with commission error rates, omission error rates, map accuracy, and Kappa calculated from the classic 2 x 2 confusion matrix (Fielding and Bell 1997, Hirzel et al. 2006). I also used COR and deviance to evaluate these models with the presence and non-detection data. To do this I calculated COR as described above. I calculated deviance as described above for positive occurrences and for sites where species were not detected, I calculated it as minus 2 times 1 minus the natural log of the associated logistic output value (Phillips and Dudik 2008). Deviance varies from 0, when non-detection samples are associated with a logistic value 0 to around 13.8, when logistic values approach 0.999.

AUC, AVI, COR, deviance, Kappa, and other evaluation measures are not informative with regard to the spatial patterns of performance of models in terms of the degree of deviance of individual observations from their expected values of 1 in areas where they are predicted to be present and 0 where they are predicted to be absent. For example, one might expect deviance to be higher at the edge of a species' range where they are limited by one or more environment factors. Similarly, deviance is expected to be higher in areas predicted to be of lower suitability. To more thoroughly evaluate spatial patterns of predictions, I plotted all test occurrences and on top of range-wide binary and habitat suitability class maps with symbols sized relative to the magnitude of the deviance value (e.g., Figure 3.3) (Lobo et al. 2008, Phillips and Dudik 2008). I then examined patterns of deviance at the scale of the range-wide binary model produced by the low cutoff threshold to see if areas of high deviance were concentrated on the margins or particular portions of a species' range. These patterns could potentially indicate that the species is limited by different factors in different portions of its range to the extent that separate modeling efforts for these regions are warranted. I also examined habitat suitability class predictions with the map of test occurrence deviances to evaluate whether low, moderate, and optimal habitat suitability classes matched my field experience with individual species.

In addition to ROC plots, Maxent produces a number of other outputs that I used to evaluate models (Maxent 2008). First, the increase in regularized gain resulting from individual environmental variables is summarized as a percent gain or contribution to the model and is used to rank the relative importance of variables. I summarized these ranks for state-wide and range-wide continuous models to identify scale dependent responses (Table 3.5). Second, jackknife charts show changes in regularized training gain, test gain, and AUC when individual environmental variables are included in the model on their own and when individual variables were removed from the model with the full compliment of environmental variables (e.g., Figure 3.4). Third, response curves for individual environmental variables showing how the logistic prediction changes as each variable is varied across its range when only that variable is included and while all other environmental variables are held constant at their average sample value (e.g., Figure 3.5).

Results and Discussion

Detailed discussions of model output for individual species including maps of continuous state-wide logistic output, binary predicted suitable habitat within species' known ranges, and, where possible, predicted low, moderate, and optimal habitat suitability classes are in Appendix D.

Responses to Environmental Variables

The STATSGO soils and geology categorical variables were ranked as the environmental variables of greatest and second greatest average importance, respectively (Table 3.7). Despite my inability to easily interpret species responses to these variables because of their large number of nominal categories, their inclusion significantly improved the predictive capability of the models as measured by levels of training and test gain and AUC, jackknife plots of training and test gain and AUC, and overall fit of the models with my field experiences on the distribution of these species. I feel that their inclusion is appropriate because geology and soils are essential to determining local habitat complexity through rates of erosion and potential plant communities and habitat complexity is of central importance to determining species diversity through niche separation (MacArthur 1958).

After STATSGO soils and geology, environmental variables of progressively less importance to models on average were land cover, elevation, soil temperature and moisture regime, Euclidian distance from major streams, slope, relative effective annual precipitation, average maximum July temperature, average minimum January temperature, ruggedness, aspect, and the solar radiation indices (Table 3.5). Solar radiation indices often made models worse when they were included and do not seem worthy of inclusion in future modeling efforts in their present form. However, examination of other forms of solar indices is probably warranted for future modeling efforts given that the biology of these species is largely dependent on maintaining optimal body temperatures (Huey 1991). For other variables, there was considerable variation as to which mattered most for individual species. For example, although terrain ruggedness was of low importance on average, 3 of Montana's 4 lizard species seem to respond to some level of terrain ruggedness because it apparently does a good job of identifying the rock outcrop habitats the species are associated with. Not surprisingly, many of Montana's amphibian and reptile species showed strong responses to elevation, minimum January temperature, soil temperature and moisture regimes, and slope (Table 3.7, Appendix D). The 1992 National Land Cover Data layer was ranked third in overall average importance of the environmental variables and the majority of land cover types species were associated with made good biological sense. However, there were some sampling artifacts associated with this layer. For example, a number of species were associated with altered land cover types such as roads and residential areas where animals are more likely to be seen and reported. Similarly, some riparian species were associated with cropland cover types due to the large agricultural presence in riparian areas, especially along major rivers. Models did not seem to be negatively impacted by these sampling artifacts in terms of over predicting the presence of species on roads or residential areas. It should be noted that the interpretability of marginal responses to environmental variables (i.e., when all other variables were held at their average value in a model) was usually compromised when they had an importance rank below 5 or 6.

In most cases importance ranks for environmental variables for state-wide and range-wide models matched one another fairly closely (Table 3.7). However, a scale dependent response to environmental variables was evident for a number of species (Appendix D). Average minimum January temperature ranked as important in determining the distributions of *P. idahoensis* (> -11 °C), *P. regilla* (> -15 °C), *E. coerulea*, (> -9 to -12 °C), *E. skiltonianus* (> -15 to -16 °C), and *O. vernalis* (< -17 °C) in models at the state-wide extent, but was essentially of no importance in models limited to the extent of each species' range. Similarly, soil temperature and moisture regime (moist soils) ranked as important in determining the distributions of *P. idahoensis*, *P. regilla*, and *E. coerulea*, in state-wide models, but was not important in models limited to the extent of each species' range. For *C. serpentina* average maximum July temperature (>29 °C) was important in the state-wide model, but not the range-wide model. For *E. coerulea*, elevation was ranked as more important within the species' range (less than 1,200 m) than it was at the state-wide scale. Finally, for *T. radix* elevations below 1,000 m were important at the state-wide scale, but were much less important within the extent of the species' range.

<u>Model Evaluations with Presence-only Data</u>

AUC, the probability that a randomly chosen occurrence site will have a higher logistic score than a randomly chosen pseudo-absence background site, was lower for both training and test data in models run at the range-wide extent as compared to the state-wide extent (Table 3.6, state-wide data not shown). The AUC values in the state-wide models are inflated because background pseudo-absence points are randomly chosen across the state-wide extent while occurrences are limited to what is assumed to be suitable habitat within the extent of the species' range. This disconnect between the sampling extent for occurrences and pseudo-absences in the state-wide models is essentially a problem of how to define available habitat and biases the AUC for test occurrences high by a value that is an inverse function of the size of the range of the species; the smaller the species range, the greater the bias in test AUC (Figure 3.6) (Fielding and Bell 1997, Chefaoui and Lobo 2008, Lobo et al. 2008, Phillips and Dudik 2008). Thus, while running models at an extent that is greater than the known range of a species can serve valuable heuristic purposes (e.g., identifying possible extensions in the known range of a species), it is inappropriate to use the resulting AUC values in a comparative manner for models of the same species run at different extents (Lobo et al. 2008). I therefore only present evaluations of models run at species' range extents and, for similar reasons, I adjusted the number of background points for each species so that they always represented 0.1 percent of the total pixels within the species' range. This does not completely solve the issue of potential biases resulting from differences in sampling for occurrence data and background pseudo-absences. For example, consider a species that has a broad distribution with highly clustered occurrences within that distribution. A uniform random sampling of the species' range will still result in a disconnect between the sample occurrences and the background samples and AUC would be expected to be biased high as a result. On the other hand, AUC may be biased low in some situations because background pseudo-absences are expected to sometimes come from areas of suitable habitat not documented with an occurrence. When this is the case, it can be shown that the maximum achievable AUC value is 1 – *a* / 2 where *a* represents the fraction of pixels covered by the species' distribution (Phillips et al. 2006). To counter all these potential sources of bias when using background points as pseudo-absences, I would ideally have targeted sampling of background points so that they match the sampling distribution of the occurrence data (Lobo et al. 2008, Phillips and Dudik 2008). Until this can be easily implemented in predictive distribution modeling software, potential biases in AUC can be identified by visually examining the spatial distributions of occurrence data relative to modeling extents.

AVI, the percentage of occurrences predicted by the binary low cutoff threshold model, was high for all training (0.87 – 1.0) and test (0.89 – 0.98) occurrences and on average the AVI was 0.21 higher for the low cutoff threshold models than the AVI for GAP models (mean = 0.68, range 0.20 – 0.98) (Table 3.6). The only low cutoff threshold models that had a lower AVI than the GAP models were those for *A. tigrinum*, *S. bombifrons*, and *C. viridis* (0.05, 0.01, and 0.02 lower, respectively). Thus, the low cutoff threshold models essentially performed equal to, or far better than, the GAP models as measured by AVI. The best performing predictive models are limited to the smallest area of suitable habitat possible while simultaneously keeping omission rates (the inverse of AVI) for test occurrences to a minimum. In addition to outperforming the GAP models for AVI metrics, the low cutoff threshold models simultaneously reduced the overall area predicted for each species' distribution by an average of 47,157 km^2. The predicted areas are smaller than those predicted by GAP for 21 species (mean (SD) = 82,412 (50,653) km^2) and larger than those predicted by GAP for 9 species, all of which were modeled in GAP as on streams or within buffers around streams (mean (SD) = 39,020 (37,520) km^2) (Table 3.7, Hart et al. 1998).

The overall higher performance of the maximum entropy models for AVI is somewhat expected given the relative simplicity of the GAP models. GAP models are deductive or rule-based and generally involve a combination of turning on appropriate habitats, filtering by appropriate elevations, and buffering around critical habitat features such as streams (Hart et al. 1998). These deductive models are important because they encapsulate expert opinion in a straightforward and interpretable manner. However, they suffer from being overly simplistic because they did not make use of available occurrence data. For example, many of the GAP models with the lowest AVI values (0.2 – 0.3) were those for stream species such as *P. idahoensis* and *A. montanus* which only predicted streams within suitable habitat cover types or those for *A. macrodactylum*, *C. picta*, and *O. vernalis* which buffered streams by a certain distance (Table 3.6, Appendix D). Expert reviewers agreed that the rules for describing these species' distributions are essentially correct (Hart et al. 1998), but evaluation with occurrence data shows that they are too restrictive and that larger buffers would have been more appropriate to do a better job of encompassing occurrences under the same general approach. Essentially, this is what the low cutoff threshold models did in their simultaneous improvement of AVI values and increase in predicted areas for these 9 species (Tables 3.7 and 3.8, Appendix D).

Evaluations of low cutoff threshold models with point biserial correlations (COR) for presence-only test data show that significant ($P < 0.001$ for all species) positive correlations (range = 0.36 – 0.59) exist between higher predicted logistic values and the species presence (Table 3.6). In addition, deviance of test occurrences from their expected logistic value of 1 were relatively small (range 1.01 – 2.47) indicating that the models were well fit to the occurrence data on average. However, just because the models are well fit on average does not mean that they are well fit across the entire extent of the area modeled. Visual inspections of deviance values associated with each test occurrence revealed no patterns in deviance values at either the range-wide extent or local-landscape scales. Visual inspections also indicated that models usually did a good job of appropriately representing suitable habitat for species for which I have field experience. Furthermore, low, moderate, and optimal habitat suitability class thresholds were easily defined and appeared to be appropriate for all species at the scale of the species' range as well as the level of a PLSS section in most areas (Figure 3.1, Appendix D). However, models for *A. tigrinum*, *C. picta*, *C. bottae*, *C. constrictor*, *H. nasicus*, *L. triangulum*, *P. catenifer*, *T. radix*, and *C. viridis* appeared to under predict habitat in regions of eastern Montana were occurrence data was not available. This may indicate that the models for these species are overfit in these areas, but it is also possible that they are performing appropriately and those areas such as the drier region between the Yellowstone and Missouri Rivers truly do not have suitable habitat for some of these species. Until these areas dominated by private and tribal lands have received more survey effort, there are three approaches to dealing with potential overfitting of models in these regions. First, with the exception of *H. nasicus*, *L. triangulum*, and *T. radix*, all of these species have distributions that cross the Continental Divide. Modeling distributions and habitat suitability for these species with two separate models on either side of the Continental Divide may improve model performance in both regions of these species' known range. Second, the continuous model outputs could be used in conjunction with GAP or other deductive models to err on the side of caution for predictions of these species' distributions until survey data is available for these regions. Third, the regularization factor that Maxent uses to constrain model estimated distributions to the characteristics of environmental features could be increased to effectively increase the entropy and predicted area of the model.

Model Evaluations with Lentic Site Survey Data

Evaluation of low cutoff threshold, optimal cutoff threshold, and GAP models with presence and nondetection data from the state-wide lentic breeding amphibian and aquatic reptile survey (Chapter 2) showed patterns of low omission error rates, high commission error rates, low levels of overall map accuracy, and low Kappa values for both the low cutoff threshold and GAP models across all species (Table 3.8). High commission errors, low map accuracies, and low Kappa values for both the Maxent and GAP models is, in part, a result of the fact that none of the environmental data layers have a grain appropriate for use at the site level. Although a general area may be suitable habitat for a species, for many species, presence at a site is dependent on site-level variables such as emergent vegetation, water permanence, and presence of fish. Furthermore, even though these surveys followed a structured set of protocols, they were conducted with single site visits, which fail to assess and correct for imperfect detectability (Kéry 2002, MacKenzie et al. 2002, 2006) so species present at a site may have gone undetected. Ideally sampling for constructing predicted distribution and habitat suitability models would involve surveying local habitat patches on multiple occasions to directly incorporate detectability into the model (MacKenzie 2006). This is certainly the most powerful approach to training and testing models and future surveys should strive to gather data in this manner.

Using the optimal habitat suitability cutoff threshold still resulted in high commission error rates, slightly higher omission error rates than the low cutoff threshold, but much higher overall map accuracy and higher Kappa values (Table 3.8). This is a result of the large number of sites where species were not detected. Increasing the cutoff threshold only reduces the commission error rate by a small percentage, but this represents a large number of sites so that overall map accuracy is raised significantly. Because avoiding omission error rates is of much more practical importance than avoiding commission errors or raising map accuracy, the only real importance of this information is that the optimal habitat suitability class is associated with a higher probability of detection. Thus, potential impacts to areas identified as optimally suitable habitat should be of special significance to landscape and project level planning.

Evaluations of low cutoff threshold models with point biserial correlations (COR) of the presence nondetection data show that significant ($P < 0.001$ for all species) positive correlations (range = 0.39 – 0.74) exist between predicted logistic values and the species' presence or nondetection (Table 3.6). Point biserial correlation values were similar to those for the presence-only test data, but were generally higher, probably as a result of the large number of non-detections in habitat with low logistic values. Similarly, average deviances from the presence (expected logistic value = 1) nondetection (expected logistic value = 0) sites were lower (range 0.55 – 1.88) than for the presence-only test occurrences. Thus, overall, large differences in logistic values seem to correlate well with species occurrences and this seems to be fairly well calibrated on average indicating that the models were well fit to the presence nondetection data on average (Phillips and Dudik 2008).

Implications for Information Needs and Conservation

Despite the overall good performance of models, large regions in Montana lack occurrence data for amphibians and reptiles and this probably resulted in poor model fit in these regions. Targeting these areas with inventory efforts would be beneficial for understanding the status and distribution of species and would improve future models. These areas include the Blackfeet, Crow, Northern Cheyenne, and Fort Peck Indian Reservations and a large section of northeast Montana that is dominated by private land with tilled agriculture.

State-wide continuous models indicated a number of opportunities to extend or fill in large gaps in known ranges of most species modeled (Appendix D). In most cases, potential range extensions suggested by the state-wide continuous models are on the scale of 20-60 km. However, several potential range extensions suggested by the state-wide models are on the order of 200 km. In many cases, these potential extensions of known range seem likely to be documented with adequate survey effort because populations have been documented in adjacent areas and there are seemingly suitable habitat connections. In other cases, these regions are not currently occupied or were never colonized. One species for which this seems likely is *P. regilla* with predicted extensions of known distribution into the Seeley and Swan Valleys, portions of the lower North and South Forks of the Flathead River drainage, the lower portions of Rock Creek, portions of the lower Clark Fork River drainage between Missoula and Superior, and portions of the Garnet Range (Figures D-11a-c). These areas all seem to have reasonable habitat, but recent calling and lentic site surveys have failed to detect them (pers. obs.). Populations isolated on the margins of their known range in the southern portion of the Bitterroot Valley and along the lower Blackfoot and Clark Fork Rivers just east of Missoula (Maxell et al. 2003) clearly deserve special management consideration to ensure their viability. Similarly, the predicted distribution of *D. aterrimus* is much broader than that documented by recent electrofishing surveys along the Idaho border (Figures D-3a-b, unpublished data). Given their extremely limited distribution, special conservation measures are clearly warranted for documented populations of this species and within the area of their predicted distribution additional surveys should be undertaken if timber harvest, road construction, or other activities altering habitat cover types are under consideration.

One of the longer potential range extensions suggested by the state-wide models is the extension of *C. bottae* from the main Northern Rocky Mountains into island ranges including the Little Rocky, Big Snowy, Pryor, and Wolf Mountains. All of these are isolated by grassland steppe habitats, but may remain somewhat connected by a network of riparian areas, which the species utilizes (Stewart 1977, St. John 2002). Presence in all of these areas seems possible because the species is highly cryptic these areas have relatively little human activity. In June of 2008, an adult *C. bottae* was found and photographed under a rock on a forested slope on the east side of the Wolf Mountains near Kirby in southeastern Bighorn County (Lisa Wilson and David Stagliano, Montana Natural Heritage Program, pers. comm.). This represents a 185 km extension in the known range of the species in Montana (Maxell et al. 2003, Werner et al. 2004) and is a 56 km extension from the nearest reported locality in Wyoming (Baxter and Stone 1985). The observation was made in an area that was predicted as suitable habitat by the low cutoff threshold of both the state-wide and range-wide models despite the fact that the closest training point was 185 km away (Figure D-11a-b). This affirms that the environmental variables used in the Maxent algorithm have the ability to predict species' distributions quite distant from occurrences used for training, and strengthens the intriguing hypothesis that the range of *C. bottae* extends into each of the other areas mentioned above (Figure D-11a). Visual encounter surveys with the aid of artificial cover objects (Hoyer and Stewart 2000) seem likely to yield additional exciting observations that will further our understanding of the distribution, status, and habitat use of this species.

The state-wide continuous model predicts large extensions in the known range of *R. luteiventris* into the Big Snowy Mountains in central Montana and the Bighorn Mountains on the Crow Indian Reservation (Figure D-13a). It is possible that the species has gone undetected and unreported in both areas because the most suitable habitat in these regions is on private and tribal lands that have not been surveyed. Of the two areas, presence in the Bighorn Mountains may be more likely given that isolated populations most genetically similar to Northern Rocky Mountains populations have been documented in adjacent portions of this mountain range in Wyoming (Dunlap 1977, Funk et al. 2008). Isolated populations of *R. sylvatica* have also been documented in this region of Wyoming (Dunlap 1977, Baxter and Stone 1985).

A relatively continuous corridor of suitable habitat extending into the intermountain valleys along the upper Missouri River and its tributaries as far as the Beaverhead, Ruby, and Madison Valleys is supported by the state-wide models for several Montana Species of Concern that are primarily distributed on the plains of eastern Montana; *S. bombifrons*, *B. cognatus*, *P. hernandesi*, *S. graciosus*, *H. nasicus*, and *L. triangulum* (MNHP and MFWP 2009, Appendix D). Intermountain valleys on the upper Missouri River have not received much survey attention because they are dominated by private lands. However, occurrence records for some species back up the distributions predicted by the models and indicate a critical need for inventory and conservation efforts. *P. hernandesi* has been collected around the Three Forks area on three occasions between 1888 and 1953, but all of the collection localities have poor spatial precision (Maxell et al. 2003). The state-wide models constructed without these records predict suitable habitat in several areas in the vicinity of Three Forks as well as a few other areas in the upper Missouri drainage that can be targeted for survey (Figures D-19a-b). Similarly, *L. triangulum* has been reported as having been collected near Three Forks in the late 1940s (Nelson 1950) and a more recent observation was reported by personnel at Lewis and Clark Caverns State Park (Grant Hokit, Carroll College, pers. comm.). Both of these areas are predicted by the Maxent models and have rock outcrop habitats that are similar to areas in eastern Montana where the species has been detected (pers. obs.). Prior to 2003, *S. bombifrons* had only been reported at four locations upstream of the Gates of the Mountains (Maxell et al. 2003). However, calling surveys since 2003 have added 8 additional localities that indicate a more continuous distribution between Townsend and Dillon and a recent observation above Hebgen Lake (Clint Sestrich, Gallatin National Forest, pers. comm.) suggests that a continuous distribution up the Madison River Valley is possible (Figure

D-6b). While *B. cognatus, S. graciosus*, and *H. nasicus* have not been reported in the area, the combination of model predictions and their presence near the Gates of the Mountains (*B. cognatus* and *H. nasicus*) or near Yellowstone Park (*S. graciosus*) indicates that an upper Missouri River distribution is possible for them as well (Appendix D, Maxell et al. 2003, Rawson and Pils 2005). With the possible exception of *S. graciosus*, which has an overall more southwestern distribution, populations of most of these species inhabiting the intermountain valleys of the upper Missouri River watershed would likely have been somewhat isolated by the canyons near the Gates of the Mountains, which had limited riparian habitat (Moulton 1983). Beginning in the 1890s, a series of dams were constructed and eventually flooded approximately 110 km of riparian areas and surrounding cliff and rock outcrop habitats these species would have depended on to depths of up to 50 m (Wright 1958). Thus, any populations of these low elevation riparian and rock outcrop dependent species that remain in the upper Missouri River headwaters above the Gates of the Mountains today have likely been isolated from their main range on the Great Plains for over 100 years. Furthermore, habitats in this region are becoming more and more fragmented by agriculture and subdivisions. This highlights the urgency for surveys guided by model outputs and implementation of conservation measures where necessary.

State-wide models indicate critical habitat corridors for maintaining the connectivity of populations of *R. pipiens*, *C. picta*, *C. constrictor*, *P. catenifer*, and *C. viridis* which range across eastern Montana into the intermountain valleys of the upper Missouri River and the larger intermountain valleys west of the Continental Divide (Appendix D, Maxell et al. 2003). Models for all of these species indicate a clear separation in habitat between areas east and west of the Continental Divide, but stringers of habitat of varying quality extend up the Big Hole River to Divide Creek and over Deer Lodge Pass to Sand Creek, Silver Bow Creek, and the upper Clark Fork River. *C. constrictor*, *P. catenifer*, and *C. viridis* would clearly have no trouble traversing the sagebrush steppe and grassland habitats in this corridor and may only be limited by availability of hibernacula (e.g., Imler 1945, Hirth et al 1969). *R. pipiens* juveniles are known to disperse up to 8.0 km from their natal ponds to their adult seasonal territories (Dole 1971, Seburn et al. 1997) and *C.*

picta have been observed 2.8 km straight line distance from the nearest surface water in eastern Montana (pers. obs.). Thus, it seems well within the dispersal capabilities of these species to cross the approximately 800 m distance and 24 m of elevation change between Divide Creek and Sand Creek with suitable aquatic overwintering areas only separated by several km. As compared with all other passes on the Continental Divide in Montana, Deer Lodge Pass seems to be the most likely corridor to provide connectivity for species limited to intermountain valley sagebrush steppe and grassland habitats. In the case of *C. picta*, which has low genetic variation across its range, indicating a recent range expansion across the Great Plains sometime in the last 14,000 years (Starkey et al. 2003), it seems likely that this same route is the most probable path by which the species colonized the Pacific Northwest (see range overview in Stebbins 2003). Similarly, although other routes are possible, it seems most likely that *C. viridis* colonized western Montana from the Great Plains via this route sometime after the last Glacial Lake Missoula flood around 13,500 years ago (Alt and Hyndman 1995, Pook et al. 2000, Ashton and de Queiroz 2001). Populations of *C. viridis* west of the Continental Divide are likely becoming more and more isolated from eastern populations as a result of human encroachment and subsequent persecution and will probably require conservation measures in coming years. Habitat suitability models may be of some use in translocating snakes from areas of recent human development.

In a pattern converse to the intermountain valley dependent species, the predicted distribution models for *A. macrodactylum*, *B. boreas* and *R. luteiventris* indicate that grassland and shrub-steppe habitats in the intermountain valleys of southwestern Montana may act as barriers to these species (Figures D-7a and D-13a). These species are known to use flood plain habitats, but typically do not use arid grassland habitats (Werner et al. 2004). This pattern is especially evident east of the Continental Divide for *B. boreas* and *R. luteiventris*. In the case of these species, critical dispersal corridors include: (1) the area around the Gates of the Mountains which connects the main Northern Rocky Mountain chain to the Big Belt Mountains; (2) the area around Maudlow which connects the Bridger and Big Belt Mountains; (3) Homestake, Deerlodge, and Elk Park passes near Butte which connect the Boulder and Highland Mountains; (4) the area just west of Virginia City which connects the Greenhorn and Gravelly ranges to the Tobacco Root Mountains; (5) the area around the Smith River Canyon which connects the Big Belt Mountains to the Little Belt Mountains; (6) and the Lake Sutherland and Martinsdale Cutoff areas which connect the Little Belt Mountains to the Castle and Crazy Mountains, respectively.

Model outputs have important implications for the conservation and restoration of *R. pipiens* in western Montana where populations have nearly been extirpated (Maxell et al. 2003, Werner 2003). The state-wide continuous model supports an historic distribution in western Montana prior to declines that was restricted to major valley bottoms and interestingly, despite several records in the lower Flathead Valley that were used to train the model, predicts more suitable habitat extending from just north of Flathead Lake up to the Canadian border near Eureka where the last historic populations still exist (Figure D-14a). This region may be the best area to attempt to reintroduce populations. No test occurrences were available to examine patterns of deviance in western Montana and it is possible that deviance in western Montana was high as a result of the influence of the large number of eastern Montana occurrences used to train the state-wide model. It may therefore, be worth creating a model to guide future reintroduction efforts that is based solely on records from the intermountain valleys of western Montana. In the mean time, the existing state-wide and range-wide models identify several areas along the upper Missouri River above the Gates of the Mountains that should be targeted for surveys to potentially locate and protect remaining breeding populations (Figure D-14a-c).

There is a great need to identify areas of potential spread for *R. catesbeiana*, because this exotic species represents a major threat to native vertebrate and invertebrate populations (Bury and Whelan 1984, Maxell 2000, Maxell et al. 2003, Werner et al. 2004). Only a state-wide continuous model was run because they have been found in a number of new localities in recent years and the extent of their range is uncertain. The model (Figures D-12a-b) indicates that a number of additional areas are capable of supporting populations, including: (1) the Flathead Valley; (2) a number of major drainages in northwest Montana including the Bull, Fisher, and Thompson Rivers; (3) the Missouri

River around Helena; (4) the Paradise Valley; (5) areas south of Billings around Bighorn Lake; and (6) streams around Lewistown. Average minimum January temperatures greater than -12 °C, and ideally greater than -9 °C, were an important environmental variable in the model indicating that documented ongoing increases in winter temperatures from climate change (Mote 2003) are likely to expand the areas where this species is capable of surviving beyond those shown by this model. The growing area that seems likely to be available for colonization in the future and the large impacts this species is likely to have on native vertebrate and invertebrate populations highlights the importance of undertaking control measures on currently established populations and educating the public to reduce or eliminate future spread. Models that include predicted state-wide average minimum January temperatures, maximum July temperatures, and precipitation under a variety of likely future climate scenarios may be helpful for prioritizing control efforts.

Alternative Map Outputs

I want to briefly discuss two additional map outputs that can be used for regional and landscape-level planning. First, the continuous 90 x 90 m grid cell logistic output can be summarized with zonal statistics to create maps or lists of predicted species within various administrative boundaries. Appropriate thresholds for this conversion are dependent on species, administrative boundaries, and questions of interest. For example, at the level of a PLSS section, applying the same low cutoff threshold used for the binary map to the average grid cell value within each section seems to reasonably represent species' distributions when the species is not limited to small landscape features such as narrow riparian corridors (e.g., Figure 3.7). Second, in the same way that GAP was designed to examine gaps in protection of biodiversity (Scott et al. 1993), the continuous or binary low cutoff threshold maps can be easily combined to summarize overall habitat suitability for various groups of species to identify core habitats, critical corridors, and gaps in stewardship protection (e.g., Figure 3.8). The overall better performance and continuous logistic output of the inductive Maxent models seem to offer advantages over the more simplistic deductive GAP models for regional and landscape level planning; most notably they seem to be applicable at finer spatial scales.

Conclusions and Recommendations

Model Performance

Maxent is one of several newer modeling approaches that have the ability to use limited amounts of presence-only data to model the predicted distribution and habitat suitability of species (Phillips et al. 2004, 2006, Phillips and Dudik 2008). These newer approaches are preferable to traditional modeling approaches that may not be applicable without absence data, limit the ways in which variables can interact to forms that are probably unrealistic, and do not have the flexibility to easily scale the complexity of models according to sample size (O'Connor 2002; Elith et al. 2006). The maximum entropy algorithm performed well in predicting the distribution and habitat suitability of Montana's 31 amphibian and reptile species and this modeling effort identified several ways to improve future models.

While state-wide models resulted in AUC values that were likely biased high relative to range-wide models as a result of a disconnect between the sampling distributions for occurrences and randomly chosen background points, there is obvious value in developing models at both scales. This helps identify scale dependent variables that likely limit species' distributions. State-wide models clearly provide a useful heuristic tool to identify areas where native species' known ranges are most likely to be expanded and were exotic species are most likely to become established to better target survey and control efforts. Range-wide binary models based on a low cutoff threshold are appropriate for regional and local-landscape planning efforts and can be used to populate species lists associated with a variety of administrative boundaries. Delineation of several habitat suitability classes using the ratios of the percent of observations in a class to the percent of the overall map area represented by that logistic class appeared to be appropriate for all species it was applied to and is likely to provide useful insights to biologists and natural resource managers in planning and conservation efforts. Simple or weighted addition of logistic outputs for complexes of different species based on status, taxonomic group, foraging guild, or other criteria may also be useful for conservation planning efforts. When compared to predictions from the deductively based models produced by the Montana Gap Analysis Project, continuous Maxent models offered more realistic depictions of amphibian and reptile species distributions when survey data was available for a region and in most cases reduced predicted area while simultaneously increasing predictive accuracy. However, deductive models like those produced by GAP are still important for representing some species distributions in areas lacking survey effort where the Maxent models may have under predicted distributions.

Suggestions for Future Modeling Efforts

(1) Models benefited greatly from the presence of STATSGO soils and surficial geology layers even though these nominal map units are not readily interpretable. Detailed analysis of the soil characteristics associated with patterns of the soil map units occupied should be conducted to inform development of state-wide layers summarizing soil characteristics such as soil depth, soil texture, and percent sand, silt, or clay, which can all be included as more interpretable continuous variables in future models.

(2) Solar radiation indices were not of value in their present form. However, examination of other forms of solar indices is probably warranted for future modeling efforts given that the biology of these species is largely dependent on their ability to behaviorally maintain optimal body temperatures.

(3) A 30-meter resolution ecological systems map is currently being developed for Montana and should be used in future modeling efforts because it promises to more accurately represent many ecological communities across Montana.

(4) Predicted temperature and precipitation regimes under likely future climate scenarios should be used in future modeling efforts in parallel with models run with current moisture and temperature regimes to provide insights into how species are likely to be impacted solely from these changes. Identifying likely future shifts in vegetation communities is also clearly of importance and would be a valuable addition to predicted distribution models under climate change. However, examining only the response to likely temperature and precipitation changes, the variables likely to be best predicted (Mote 2003), is likely to be useful in the identification of species most

likely to be impacted if vegetation communities are not able to respond quickly enough to support shifts in species' ranges.

(5) A sensitivity analysis should be performed to assess the value of increasing the regularization factor that Maxent uses to constrain estimated distributions to the characteristics of environment variables associated with occurrences. This would assist with the potential problem of models that appeared to under predict the distributions of some species in areas where occurrence data is lacking.

(6) Ideally future sampling for occurrence data used to construct predicted distribution and habitat suitability models would involve surveying local habitat patches on multiple occasions as part of a state-wide sampling scheme to directly incorporate detectability into the model (MacKenzie et al. 2003, 2006, MacKenzie 2006). This is certainly the most powerful approach to training and testing models and future surveys should strive to gather data in this manner.

Inventory and Conservation Needs

(1) Large regions of Montana lack occurrence data for amphibians and reptiles and this probably resulted in poor model fit in these regions. Targeting these areas with inventory efforts would be beneficial for understanding the status and distribution of species and would improve future models. These areas include the Blackfeet, Crow, Northern Cheyenne, and Fort Peck Indian Reservations and a large section of northeast Montana that is dominated by private land with tilled agriculture.

(2) Stream species such as *A. montanus*, *C. serpentina*, and *A. spinifera* still have relatively little data despite the fact that fisheries biologists work in these habitats across the state on a regular basis. Documentation of these species in the course of other fisheries surveys would greatly benefit our understanding of their distribution and status.

(3) Numerous extensions to the known ranges of species are supported by the models. In particular, significant extensions seem likely for *E. skiltonianus, S. graciosus, C. bottae*, and *L. triangulum*. This highlights the need for systematic inventories for reptile species across Montana and the need to conduct focal surveys for a number of amphibian and reptile species using the model output to guide survey efforts.

(4) A number of species with ranges primarily on the plains of eastern Montana have populations in the intermountain valleys of the upper Missouri watershed that are likely isolated above large hydrologic developments. Models can guide inventory efforts for these populations to identify conservation measures that can be taken for isolated populations of *S. bombifrons*, *R. pipiens*, *P. hernandesi*, *S. graciosus*, *C. viridis*, and possibly *L. triangulum* and *H. nasicus*.

(5) Model output highlights potential corridors that may be critical for maintaining connectivity of intermountain valley species such as *C. picta*, *C. constrictor*, *P. catenifer*, and *C. viridis* across the Continental Divide. An inverse pattern of potential corridors between mountain ranges is highlighted in western Montana for *R. luteiventris* and *B. boreas* whose distributions are primarily limited to mountainous areas and major river valleys.

(6) Models highlight the need to conserve isolated populations of *D. atterimus*, *P. regilla* and *C. viridis* west of the Continental Divide where pressures from human activities are increasing rapidly.

(7) Reintroduction efforts in western Montana for *R. pipiens* might best be focused on the northern portions of the Flathead Valley, but models using data from only west of the Continental Divide should be constructed to see if this pattern holds without occurrences from eastern Montana. In the mean time, the existing state-wide and current range-wide models identify several areas along the upper Missouri River above the Gates of the Mountains that should be targeted for survey to locate and protect any remaining breeding populations in western Montana.

(8) Models indicate that *R. catesbeiana* populations are capable of becoming established in a much broader region than they are currently limited to. This highlights the need to undertake control efforts on existing populations, targeting those most likely to spread first.

Table 3.1. Environmental variables used in models. Metadata for each environmental layer is provided in Appendix C.

Variable	Type	Description
Aspect	categorical	Dominant degrees of aspect grouped into 9 categories including flat
Elevation	continuous	Elevation in meters from the National Elevation Dataset (NED)
Geology	categorical	931 categories of surficial geology
Land Cover	categorical	1992 National Land Cover Data (NLCD) – 21 classes
Max Temp	continuous	Estimated average maximum daily July temperature in degrees Fahrenheit for 1971-2000
Min Temp	continuous	Estimated average minimum daily January temperature in degrees Fahrenheit for 1971-2000
Precip	continuous	Relative Effective Annual Precipitation (REAP) in 1 cm intervals as an indicator of available soil moisture
Ruggedness	continuous	Vector ruggedness measure (VRM) of local terrain
Slope	continuous	Degrees of slope
Soils	categorical	694 soil mapping units from the state soil geographic data (STATSGO) on general soil associations developed by the National Cooperative Soil Survey
Soil TM	categorical	Soil temperature and moisture regime – 12 categories
Solar E	continuous	Solar radiation index (SRI) at each tenth degree of latitude at the equinox
Solar SS	continuous	Solar radiation index (SRI) at each tenth degree of latitude at the summer solstice
Solar WS	continuous	Solar radiation index (SRI) at each tenth degree of latitude at the winter solstice
Stream ED	continuous	Euclidian distance from major streams in 1 meter intervals

Table 3.2. Observation records in the Montana Natural Heritage Program's state-wide database, number with suitable spatial precision, and number of spatially unique records available for training and testing models.

Species	Total No. Records	No. Records <400 m uncertainty	No. Training Records	No. Test Records
A. macrodactylum	2174	1798	1122	387
A. tigrinum	1146	937	587	202
D. atterimus	55	52	45	-
P. idahoensis	159	142	65	-
A. montanus	972	398	214	73
S. bombifrons	455	369	263	88
B. boreas	1599	1336	789	288
B. cognatus	294	225	164	55
B. woodhousii	1017	840	599	205
P. maculata	3169	2864	2013	682
P. regilla	387	298	176	59
R. catesbeiana	61	45	37	-
R. luteiventris	5150	4140	2431	854
R. pipiens	1475	1192	788 / 766 [a]	270 / 265 [a]
C. serpentina	92	59	43	-
C. picta	1188	939	608	215
A. spinifera	176	147	85	-
E. coerulea	86	48	48	-
P. hernandesi	187	83	78	-
S. graciosus	197	120	102	-
E. skiltonianus	55	21	18	-
C. bottae	175	65	63	-
C. constrictor	571	367	260	88
H. nasicus	129	70	66	-
O. vernalis	40	32	22	-
L. triangulum	57	22	22	-
P. catenifer	744	513	371	126
T. elegans	1566	977	609	212
T. radix	721	589	424	145
T. sirtalis	950	635	377	132
C. viridis	826	474	338	115

[a] Number of records used for state-wide historic range model / number of records used for current range model.

Table 3.3. Habitat suitability class cutoffs resulting from plots of the ratio of the percent of observations to percent of pixels in each habitat suitability class. Species without cutoff values for moderate and optimal habitat suitability classes did not have test data available to define these classes.

Species	Low	Moderate	Optimal
A. macrodactylum	0.123	0.25	0.55
A. tigrinum	0.093	0.45	0.65
D. atterimus	0.174	-	-
P. idahoensis	0.008	-	-
A. montanus	0.107	0.35	0.75
S. bombifrons	0.044	0.15	0.65
B. boreas	0.073	0.35	0.65
B. cognatus	0.046	0.2	0.55
B. woodhousii	0.072	0.25	0.65
P. maculata	0.099	0.35	0.65
P. regilla	0.060	0.15	0.55
R. catesbeiana	0.006	-	-
R. luteiventris	0.094	0.35	0.55
R. pipiens	0.077	0.2	0.65
C. serpentina	0.031	-	-
C. picta	0.052	0.15	0.55
A. spinifera	0.008	-	-
E. coerulea	0.048	-	-
P. hernandesi	0.047	-	-
S. graciosus	0.024	-	-
E. skiltonianus	0.041	-	-
C. bottae	0.032	-	-
C. constrictor	0.058	0.15	0.55
H. nasicus	0.056	-	-
O. vernalis	0.065	-	-
L. triangulum	0.025	-	-
P. catenifer	0.052	0.15	0.45
T. elegans	0.075	0.25	0.65
T. radix	0.049	0.3	0.65
T. sirtalis	0.035	0.15	0.65
C. viridis	0.099	0.35	0.65

Table 3.4. Variable importance rankings (state-wide model / range-wide model) with overall averages ordered left to right.

Species	Soils	Geology	Land Cover	Elevation	Soil TM	Stream ED	Slope	Precip	Max Temp	Min Temp	Ruggedness	Aspect	Solar E	Solar WS	Solar SS
A. macrodactylum	5 / 3	4 / 5	6 / 6	8 / 7	2 / 4	12 / 11	3 / 1	1 / 2	10 / 10	7 / 8	9 / 9	11 / 12	14 / 15	15 / 14	13 / 13
A. tigrinum	1 / 1	2 / 2	4 / 3	9 / 6	8 / 5	14 / 12	3 / 4	6 / 7	10 / 14	5 / 8	7 / 11	15 / 10	11 / 9	12 / 13	13 / 15
D. atterimus	4 / 5	1 / 11	5 / 14	6 / 8	8 / 6	12 / 13	7 / 7	2 / 3	15 / 10	9 / 2	3 / 1	10 / 4	13 / 9	14 / 12	11 / 15
P. idahoensis	5 / 4	7 / 6	8 / 9	4 / 3	1 / 14	2 / 1	9 / 5	11 / 12	12 / 8	6 / 15	3 / 2	10 / 7	15 / 10	14 / 13	13 / 11
A. montanus	3 / 2	2 / 1	6 / 8	8 / 10	7 / 9	9 / 6	4 / 3	1 / 5	10 / 11	11 / 13	5 / 4	12 / 7	13 / 14	14 / 15	15 / 12
S. bombifrons	1 / 1	3 / 2	4 / 4	7 / 6	5 / 5	11 / 10	6 / 8	8 / 9	2 / 3	12 / 7	9 / 13	10 / 14	13 / 11	14 / 12	15 / 15
B. boreas	4 / 2	6 / 5	7 / 8	10 / 9	3 / 7	5 / 4	2 / 1	1 / 3	8 / 11	14 / 6	9 / 10	12 / 13	11 / 12	15 / 15	13 / 14
B. cognatus	1 / 1	2 / 2	7 / 7	4 / 4	5 / 6	9 / 9	6 / 5	15 / 13	3 / 3	13 / 11	8 / 15	10 / 8	14 / 10	12 / 12	11 / 14
B. woodhousii	2 / 1	3 / 2	9 / 8	5 / 5	14 / 14	4 / 4	8 / 9	10 / 11	1 / 3	6 / 6	7 / 7	11 / 10	13 / 12	12 / 13	15 / 15
P. maculata	1 / 1	3 / 2	5 / 4	8 / 11	4 / 5	9 / 6	2 / 3	7 / 7	15 / 15	6 / 9	11 / 14	14 / 8	12 / 12	10 / 13	13 / 10
P. regilla	5 / 2	3 / 4	8 / 7	4 / 1	2 / 9	11 / 14	7 / 3	6 / 8	9 / 6	1 / 11	10 / 5	15 / 13	12 / 12	14 / 10	13 / 15
R. catesbeiana	1 / -	4 / -	7 / -	10 / -	8 / -	3 / -	5 / -	14 / -	13 / -	2 / -	12 / -	9 / -	15 / -	11 / -	6 / -
R. luteiventris	5 / 3	6 / 4	7 / 5	8 / 8	3 / 6	10 / 10	2 / 1	1 / 2	4 / 7	11 / 12	9 / 9	12 / 11	13 / 14	14 / 15	15 / 13
R. pipiens	1 / 1	2 / 2	4 / 5	3 / 4	7 / 7	6 / 3	5 / 6	14 / 12	13 / 14	8 / 8	9 / 11	10 / 9	15 / 15	11 / 10	12 / 13
C. serpentina	1 / 3	3 / 1	6 / 6	5 / 4	13 / 9	2 / 2	10 / 11	14 / 13	4 / 15	9 / 12	7 / 14	8 / 5	11 / 8	15 / 7	12 / 10
C. picta	1 / 1	4 / 3	2 / 2	3 / 4	7 / 7	6 / 6	5 / 5	8 / 9	10 / 12	9 / 8	11 / 11	15 / 10	12 / 13	14 / 14	13 / 15
A. spinifera	3 / 4	2 / 2	4 / 5	6 / 3	9 / 6	1 / 1	12 / 12	5 / 9	7 / 13	14 / 14	8 / 7	11 / 10	13 / 8	10 / 11	15 / 15
E. coerulea	6 / 4	3 / 3	10 / 8	5 / 1	4 / 14	13 / 13	12 / 7	9 / 10	8 / 5	2 / 15	1 / 2	7 / 6	11 / 9	15 / 12	14 / 11
P. hernandesi	1 / 1	3 / 2	2 / 3	14 / 13	4 / 5	10 / 8	9 / 10	13 / 14	6 / 11	15 / 15	5 / 4	7 / 9	8 / 12	12 / 7	11 / 6
S. graciosus	1 / 3	7 / 6	6 / 7	4 / 1	2 / 4	10 / 8	15 / 11	3 / 2	9 / 13	11 / 10	5 / 5	8 / 9	13 / 12	14 / 15	12 / 14
E. skiltonianus	2 / 1	3 / 2	5 / 4	10 / 7	4 / 9	9 / 6	8 / 10	11 / 11	12 / 13	1 / 5	13 / 12	6 / 3	15 / 14	7 / 8	14 / 15
C. bottae	1 / 1	2 / 2	7 / 9	8 / 3	5 / 7	4 / 4	11 / 8	13 / 12	10 / 13	6 / 14	3 / 5	9 / 6	12 / 10	15 / 15	14 / 11
C. constrictor	1 / 1	2 / 2	7 / 6	4 / 4	15 / 15	6 / 5	11 / 12	5 / 7	3 / 3	13 / 14	9 / 11	12 / 10	8 / 8	10 / 9	14 / 13
H. nasicus	1 / 1	3 / 3	6 / 4	7 / 6	4 / 2	5 / 5	11 / 13	15 / 14	2 / 7	8 / 10	14 / 15	9 / 8	10 / 9	13 / 11	12 / 12
O. vernalis	4 / 3	3 / 1	5 / 5	1 / 2	6 / 4	7 / 6	12 / 10	14 / 14	13 / 13	2 / 8	8 / 12	9 / 7	10 / 11	15 / 15	11 / 9
L. triangulum	1 / 1	3 / 2	4 / 3	15 / 15	13 / 12	7 / 5	5 / 11	14 / 14	2 / 4	11 / 10	8 / 8	6 / 6	9 / 7	10 / 13	12 / 9
P. catenifer	1 / 1	3 / 3	6 / 6	5 / 5	8 / 8	4 / 4	11 / 9	7 / 7	2 / 2	14 / 14	15 / 15	10 / 11	9 / 10	12 / 12	13 / 13
T. elegans	1 / 1	4 / 3	9 / 7	11 / 10	6 / 5	3 / 2	5 / 6	2 / 4	7 / 9	8 / 8	10 / 11	12 / 12	13 / 13	14 / 14	15 / 15
T. radix	5 / 2	7 / 5	4 / 3	1 / 8	3 / 4	10 / 7	6 / 6	9 / 10	8 / 9	2 / 1	11 / 13	13 / 11	12 / 12	15 / 5	14 / 14
T. sirtalis	2 / 2	1 / 1	7 / 5	9 / 8	5 / 6	8 / 9	4 / 4	6 / 7	13 / 12	3 / 3	10 / 10	12 / 14	14 / 13	15 / 15	11 / 11
C. viridis	1 / 1	2 / 2	6 / 6	7 / 7	8 / 8	3 / 3	14 / 15	4 / 5	5 / 4	12 / 12	13 / 14	9 / 9	10 / 10	11 / 11	15 / 13
Average Rank	2.3 / 1.9	3.3 / 3	5.9 / 5.9	6.7 / 6.1	6.2 / 7.4	7.3 / 6.6	7.4 / 7.2	8 / 8.5	7.9 / 9.1	8.1 / 9.6	8.5 / 9.3	10.5 / 9.1	12.1 / 11.1	12.9 / 12	12.9 / 12.7

Table 3.5. Evaluations of range-wide continuous models with area under the receiver operating characteristic plot (AUC) and a low cutoff threshold of these models with absolute validation index (AVI), point biserial correlations (COR), and deviance. COR and deviance were also used to evaluate models for species with presence / non-detection (PND) data from a recent state-wide inventory of lentic water bodies. Other evaluations are for training and test presence only data as indicated; test data was not available for some species (Table 3.2). In addition, all training and test presence only data was used to evaluate Gap Analysis Project (GAP) models (Hart et al. 1998) with AVI. See text for descriptions of metrics. All COR values were statistically significant ($P < 0.0001$).

Species	AUC Training	AUC (SD) Test	AVI Training	AVI Test	GAP AVI	COR Test	COR PND	Deviance (SD) Test	Deviance (SD) PND
A. macrodactylum	0.932	0.909 (0.008)	0.98	0.96	0.20	0.47	0.15	1.14 (1.16)	1.88 (1.46)
A. tigrinum	0.918	0.878 (0.013)	0.98	0.93	0.98	0.50	0.31	1.53 (1.63)	1.39 (1.321)
D. atterimus	0.992	-	1.00	-	0.92	-	-	-	-
P. idahoensis	0.982	-	1.00	-	0.30	-	-	-	-
A. montanus	0.940	0.889 (0.024)	0.99	0.92	0.32	0.59	-	1.42 (2.01)	-
S. bombifrons	0.951	0.895 (0.015)	1.00	0.95	0.96	0.36	0.27	2.05 (1.87)	0.55 (0.83)
B. boreas	0.937	0.888 (0.011)	0.99	0.95	0.83	0.48	0.15	1.40 (1.62)	1.48 (1.24)
B. cognatus	0.966	0.877 (0.021)	0.98	0.89	0.77	0.47	0.18	2.47 (2.24)	0.58 (0.82)
B. woodhousii	0.954	0.940 (0.008)	0.98	0.98	0.92	0.38	0.54	1.32 (1.45)	0.56 (1.03)
P. maculata	0.886	0.884 (0.006)	0.98	0.97	0.93	0.39	0.45	1.29 (1.18)	1.14 (1.10)
P. regilla	0.956	0.933 (0.015)	0.99	0.97	0.84	0.49	0.38	1.15 (1.30)	0.69 (0.85)
R. catesbeiana	-	-	1.00	-	0.03	-	-	-	-
R. luteiventris	0.942	0.936 (0.004)	0.99	0.98	0.53	0.38	0.21	1.01 (1.07)	1.66 (1.38)
R. pipiens	0.945	0.923 (0.008)	0.99	0.96	0.53	0.43	0.28	1.47 (1.46)	1.44 (1.26)
C. serpentina	0.993	-	1.00	-	0.44	-	-	-	-
C. picta	0.959	0.929 (0.009)	0.99	0.97	0.22	0.38	0.23	1.50 (1.75)	0.88 (1.33)
A. spinifera	0.999	-	1.00	-	0.28	-	-	-	-
E. coerulea	0.982	-	0.98	-	0.67	-	-	-	-
P. hernandesi	0.972	-	1.00	-	0.87	-	-	-	-
S. graciosus	0.992	-	0.99	-	0.94	-	-	-	-
E. skiltonianus	0.998	-	1.00	-	0.79	-	-	-	-
C. bottae	0.994	-	1.00	-	0.92	-	-	-	-
C. constrictor	0.953	0.858 (0.020)	0.99	0.91	0.91	0.44	-	2.38 (2.35)	-
H. nasicus	0.973	-	1.00	-	0.76	-	-	-	-
O. vernalis	0.981	-	0.87	-	0.22	-	-	-	-
L. triangulum	0.994	-	1.00	-	0.68	-	-	-	-
P. catenifer	0.956	0.897 (0.013)	0.99	0.96	0.89	0.35	-	1.92 (2.00)	-
T. elegans	0.960	0.909 (0.011)	0.99	0.95	0.94	0.42	0.20	1.54 (1.74)	1.26 (1.33)
T. radix	0.928	0.908 (0.011)	0.99	0.98	0.85	0.35	0.18	1.56 (1.55)	1.65 (1.33)
T. sirtalis	0.981	0.935 (0.012)	1.00	0.91	0.80	0.58	0.28	1.77 (2.48)	0.83 (1.45)
C. viridis	0.948	0.875 (0.017)	0.99	0.90	0.92	0.54	-	1.84 (1.88)	-

Table 3.6. Area and percent of Montana with predicted suitable habitat for range-wide low cutoff threshold and Gap analysis models.

Species	Low Cutoff Area (km^2)	GAP Area (km^2)	Difference in Area Estimates (km^2) [a]	Percent change in estimated area [b]	Low Cutoff Percent of Montana	GAP Percent of Montana
A. macrodactylum	42455	10590	31,865	301	11.2	2.8
A. tigrinum	147503	286959	-139,456	-49	38.8	75.4
D. atterimus	364	3213	-2,850	-89	0.1	0.8
P. idahoensis	5891	876	5,015	572	1.6	0.2
A. montanus	33844	4918	28,927	588	8.9	1.3
S. bombifrons	136949	265254	-128,305	-48	36	69.7
B. boreas	85053	143761	-58,708	-41	22.4	37.8
B. cognatus	91507	175550	-84,043	-48	24.1	46.1
B. woodhousii	71792	167403	-95,611	-57	18.9	44
P. maculata	190628	282931	-92,303	-33	50.2	74.3
P. regilla	22360	27461	-5,101	-19	5.9	7.2
R. catesbeiana [c]	34734	61	34,673	56,841	9.1	0
R. luteiventris	82215	32183	50,032	155	21.6	8.5
R. pipiens [c]	119646	51884	67,762	131	31.5	13.6
C. serpentina	12686	12694	-8	0	3.3	3.3
C. picta	154087	32782	121,305	370	40.6	8.6
A. spinifera	8380	454	7,926	1,746	2.2	0.1
E. coerulea	11646	17025	-5,379	-32	3.1	4.5
P. hernandesi	79179	179586	-100,407	-56	20.8	47.2
S. graciosus	31386	124344	-92,958	-75	8.3	32.7
E. skiltonianus	8775	13724	-4,949	-36	2.3	3.6
C. bottae	39279	101288	-62,009	-61	10.3	26.6
C. constrictor	158427	259260	-100,832	-39	41.7	68.1
H. nasicus	73027	198272	-125,245	-63	19.2	52.1
O. vernalis	6052	2373	3,679	155	1.6	0.6
L. triangulum	36033	131857	-95,824	-73	9.5	34.6
P. catenifer	172658	283036	-110,378	-39	45.4	74.4
T. elegans	123959	302854	-178,895	-59	32.6	79.6
T. radix	116086	199169	-83,083	-42	30.6	52.3
T. sirtalis	83416	205875	-122,459	-59	22	54.1
C. viridis	144149	268409	-12,4260	-46	37.9	70.5

[a] Differences are low cutoff range model estimates minus GAP estimates.
[b] (Area difference / area estimated by GAP) x 100.
[c] State-wide model was used for *R. catesbeiana* because of potential to spread. Areas where *R. pipiens* has declined or been extirpated were excluded.

Table 3.7. Commission[a], omission[b], map accuracy[c], and Kappa Index[d] assessments of GAP analysis models and low and optimal habitat class binary cutoff thresholds for range-wide models using lentic site survey data from the Montana Amphibian Inventory Project.

Species	Low Cutoff Commission	Optimal Cutoff Commission	GAP Commission	Low Cutoff Omission	Optimal Cutoff Omission	GAP Omission	Low Cutoff Map Accuracy	Optimal Cutoff Map Accuracy	GAP Map Accuracy	Low Cutoff Kappa Index	Optimal Cutoff Kappa Index	GAP Kappa Index	N
A. macrodactylum	0.647	0.619	0.719	0.112	0.246	0.358	0.375	0.488	0.569	0.030	0.097	-0.061	2123
A. tigrinum	0.832	0.710	0.831	0.003	0.087	0.017	0.272	0.728	0.282	0.047	0.237	0.048	2556
S. bombifrons	0.953	0.807	0.963	0.000	0.024	0.000	0.264	0.922	0.072	0.022	0.216	0.003	1581
B. boreas	0.959	0.922	0.958	0.008	0.022	0.022	0.109	0.713	0.205	0.005	0.075	0.007	3418
B. cognatus	0.980	0.923	0.983	0.000	0.009	0.006	0.258	0.914	0.231	0.010	0.108	0.005	1552
B. woodhousii	0.794	0.513	0.866	0.003	0.043	0.012	0.668	0.910	0.457	0.229	0.471	0.096	1552
P. maculata	0.662	0.476	0.695	0.002	0.182	0.187	0.440	0.728	0.385	0.135	0.348	0.049	2764
P. regilla	0.936	0.745	0.904	0.002	0.014	0.015	0.382	0.898	0.681	0.043	0.333	0.102	1375
R. luteiventris	0.568	0.508	0.503	0.128	0.260	0.362	0.443	0.564	0.573	0.026	0.180	0.136	3422
R. pipiens	0.782	0.672	0.787	0.000	0.134	0.200	0.268	0.671	0.596	0.034	0.214	0.015	1546
C. picta	0.938	0.869	0.936	0.000	0.002	0.027	0.555	0.814	0.907	0.065	0.187	0.051	4961
T. elegans	0.960	0.892	0.966	0.004	0.015	0.033	0.211	0.802	0.115	0.013	0.135	0.000	4182
T. radix	0.828	0.776	0.840	0.009	0.119	0.157	0.232	0.627	0.212	0.028	0.119	0.001	1522
T. sirtalis	0.946	0.866	0.955	0.002	0.009	0.016	0.488	0.846	0.528	0.048	0.186	0.030	3993

[a] Number of lentic site surveys where species was predicted to be present, but was not detected.

[b] Number of lentic site surveys where species was predicted to be absent, but was detected.

[c] Sum of the number of lentic site surveys where species was predicted present and was detected and predicted absent and not detected divided by the total number of sites surveyed.

[d] ((Total number of sites surveyed * (sum of the number of lentic site surveys where species was predicted present and was detected plus predicted absent and not detected)) – ((total number of detections * total number of predicted presences) + (total number of non detections * total number of predicted absences))) / (total number of sites surveyed squared – ((total number of detections * total number of predicted presences) + (total number of non detections * total number of predicted absences))).

<u>Figure Legends</u>

Figure 3.1. Example ratio of percentage of observations to percent of pixels curve for determining habitat suitability classes for *B. boreas*. Low habitat suitability classes have a small proportion of the total number of occurrences relative to the pixel area represented by that logistic output class. Higher habitat suitability classes have larger proportions of the total number of occurrences relative to the pixel area represented by each logistic output class. The dashed line at a ratio of 1 represents a completely random model. Cutoff values assigned for this species were non-habitat <0.073, low habitat suitability = 0.073-0.35, moderate habitat suitability = 0.35-0.65, and optimal habitat suitability = > 0.65.

Figure 3.2. Example receiver operating characteristic (ROC) curve for the *B. boreas* range-wide continuous model. Training data is shown in red. Test data is shown in dark blue. Performance of a random model is shown in light blue. The curve plots sensitivity, also known as the true positive rate or 1 minus the omission rate, against 1 – specificity, also known as the false positive rate, across all thresholds of the continuous model. Because the ROC plot provides information across thresholds, it avoids the subjectivity of choosing a single threshold and the total area under the curve (AUC) provides a single measure of the overall performance of the model. For *B. boreas*, AUC = 0.888, indicating that for any given threshold the model will correctly evaluate a random selection from the occurrence data as having a higher score than a paired random selection from the background pseudo-absences 89 percent of the time. AUC values vary from 0 to 1 with a random model performing at a value of 0.5.

Figure 3.3. Example habitat suitability class evaluation at the landscape scale using deviances of test occurrences. Test occurrence deviances for *R. luteiventris* are mapped with black circles sized relative to the magnitude of deviance of the underlying predicted logistic value from that predicted by the occurrence (i.e., a logistic value of 1). The region shown is centered on Glacier National Park with areas classified as unsuitable habitat showing as an aerial photograph for perspective, and habitats classified as low,

moderate, and optimal suitability in yellow, orange, and red, respectively. Habitat suitability classifications perform well, with the magnitude of test occurrence deviances inversely correlated with predicted habitat suitability across the landscape. The two test occurrences with large deviances and white centers fell below the binary low cutoff threshold for suitable habitat (i.e. these represent omission errors in the low cutoff threshold binary model).

Figure 3.4. Example jackknife charts for *B. boreas* showing the importance of environmental variables as a function of the change in training gain (a), test gain (b), and AUC on test data (c) resulting from sole inclusion (dark blue bars) or sole exclusion (light blue bars) of the environmental variable in the model. The red bar indicates the maximum gain or test AUC achieved with inclusion of all variables. Geology, STATSGO soils, slope, and Euclidean distance from streams are the most important variables on their own and removal of slope from the model resulted in the greatest reduction in gain or AUC. The solar radiation indices (srie, sris, and sriw) are all unimportant on their own and the solar radiation index at the summer solstice (sris) actually resulted in a negative test gain indicating a model with this variable alone is worse than a null model.

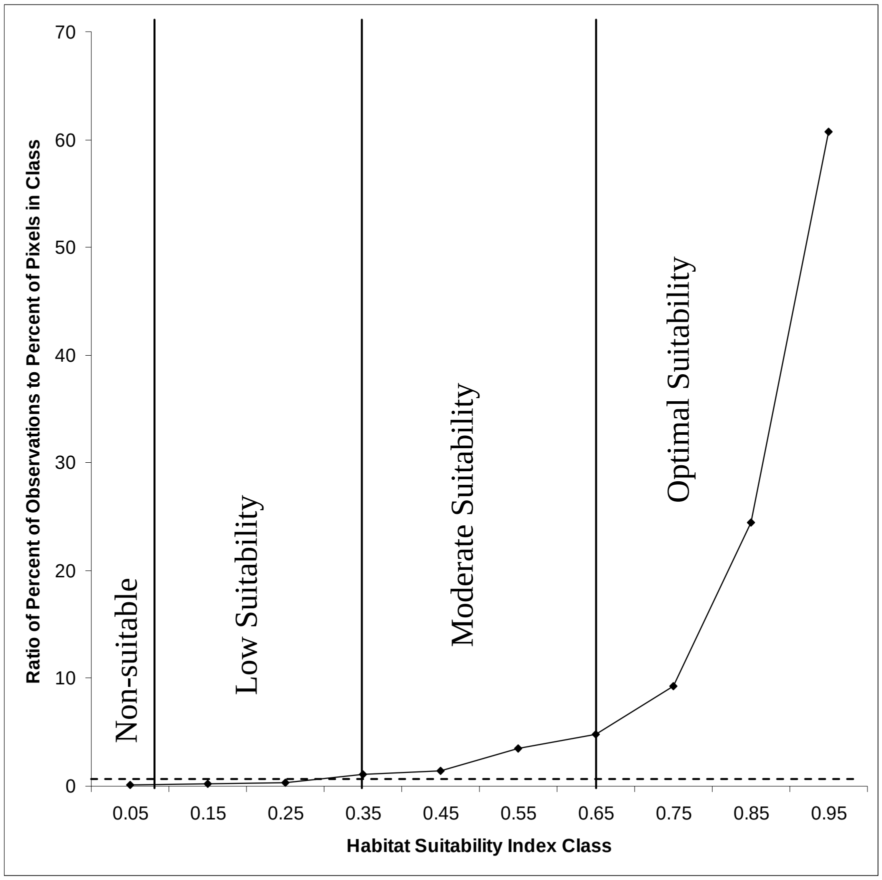
Figure 3.5. Example response curves for *B. boreas* to individual environmental variables showing how the logistic prediction changes as each environmental variable is varied while all other environmental variables are held constant at their average sample values. Maxent also provides response curves for the individual environmental variables on their own. Note that if any of the environmental variables are correlated, the marginal response curves can be misleading (e.g., two highly correlated variables with opposite response curves could counter act one another).

Figure 3.6. Reduction in AUC values as a function of range area for 30 amphibian and reptile species showing the importance of training and evaluating models within the range of a species rather than an arbitrary administrative boundary (e.g., a state). Difference in AUC between state-wide and range-wide models is inversely related to the size of the

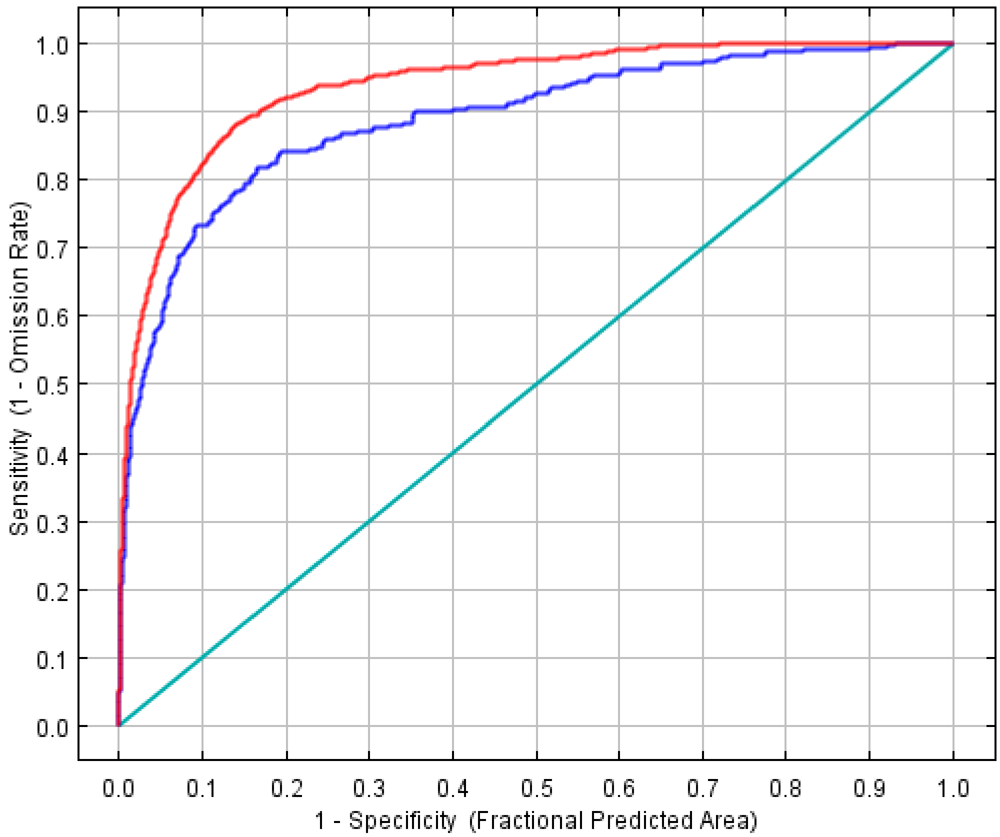
species' range relative to that administrative boundary. Training data (non significant slope, $p = 0.35$, $r^2 = 0.03$) is represented by diamonds and a dashed line. Test data (significant slope, $p = 0.0002$, $r^2 = 0.058$) is represented by triangles and a solid line.

Figure 3.7. Example of using continuous logistic output at the scale of 90 x 90 m grid cells to represent predicted occupancy at the scale of Public Land Survey System (PLSS) sections for *R. luteiventris*. The upper portion of the image is centered on the Tobacco Root Mountains in southwestern Montana with areas classified as unsuitable habitat showing as an aerial photograph to provide perspective, and habitats classified as low, moderate, and optimal suitability in yellow, orange, and red, respectively. PLSS sections are shown as predicted to contain suitable habitat when the average logistic value for the approximately 320 grid cells contained by each section is greater than the low binary cutoff threshold for the continuous model. While this appears to be appropriate for protecting the species core habitats, it clearly misses narrow riparian corridors in valley bottoms that are critical for maintaining connectivity between core areas. Thus, appropriate thresholds are dependent on the species, administrative boundaries, and questions of interest.

Figure 3.8. Example of overall predicted habitat suitability and species diversity for amphibians and reptiles in Glacier National Park. A National Agriculture Imagery Program (NAIP) color image (a) is included for reference to cumulative logistic output predicting areas of highest amphibian diversity (b), reptile diversity (c), and overall herpetofauna diversity (d). Warmer colors (reds) represent higher and cooler colors (blues) represent lower overall predicted habitat suitability.



70
60
50
40
30
20
10
0
Ratio of Percent of Observations to Percent of Pixels in Class
Non-suitable
Low Suitability
Moderate Suitability
Optimal Suitability
0.05
0.15
0.25
0.35
0.45
0.55
0.65
0.75
0.85
0.95
Habitat Suitability Index Class


Sensitivity (1 - Omission Rate)
1 - Specificity (Fractional Predicted Area)
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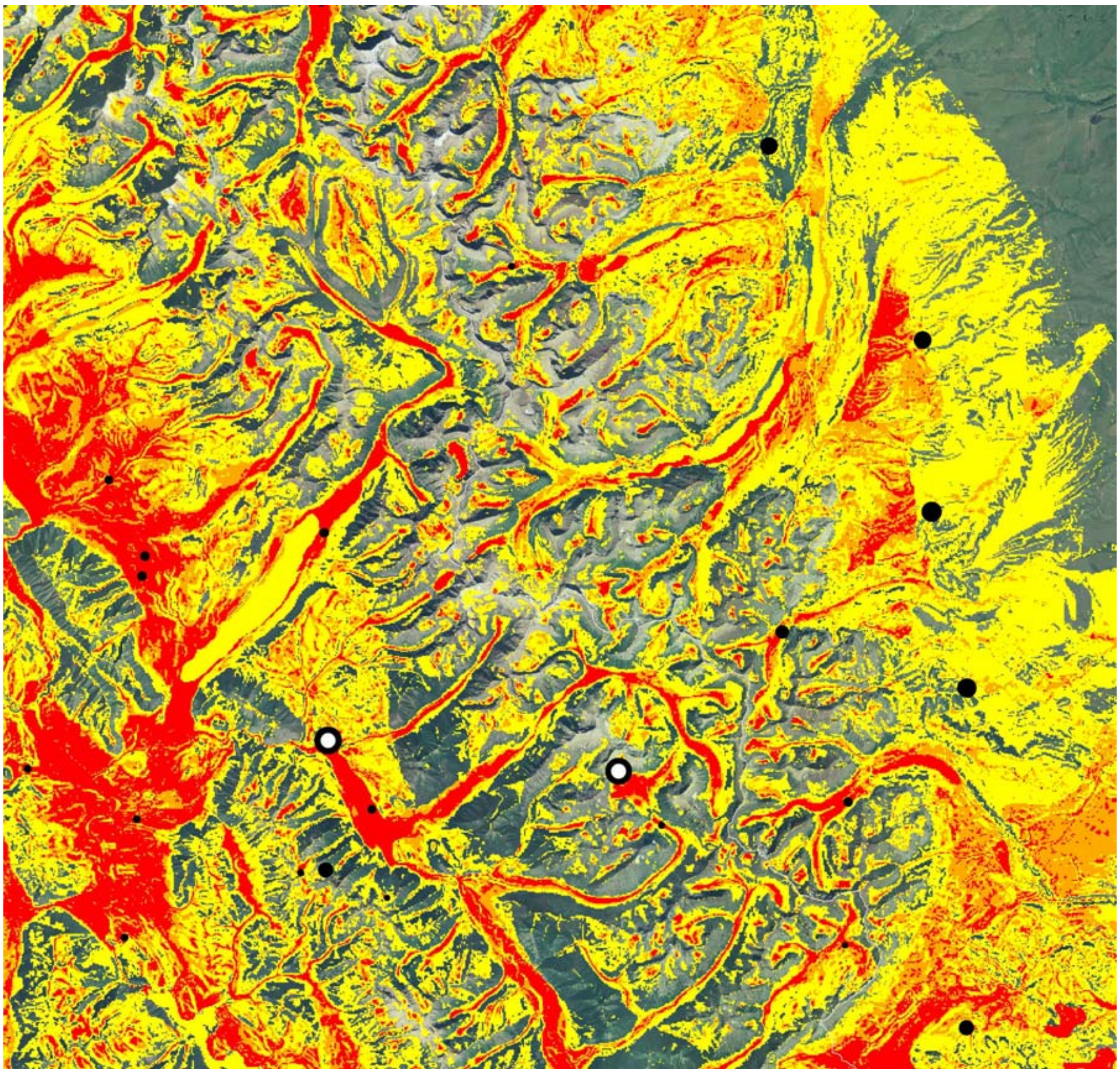
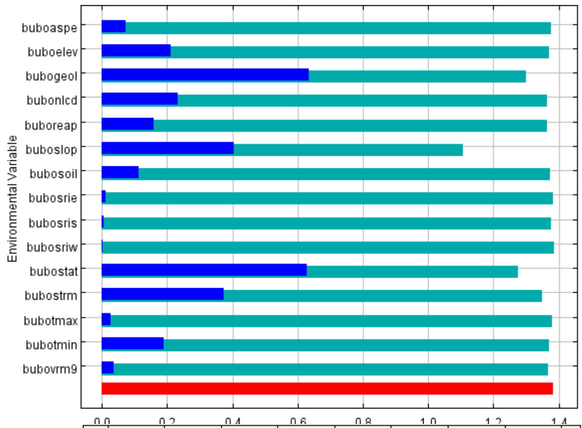
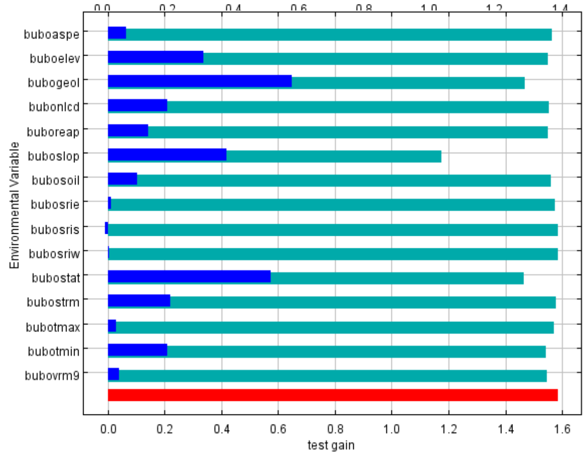
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0



(a)


Environmental Variable
buboaspe
buboelev
bubogeol
bubonlcd
buboreap
buboslop
bubosoil
bubosrie
bubosris
bubosriw
bubostat
bubostrm
bubotmax
bubotmin
bubovrm9

(b)


Environmental Variable
buboaspe
buboelev
bubogeol
bubonlcd
buboreap
buboslop
bubosoil
bubosrie
bubosris
bubosriw
bubostat
bubostrm
bubotmax
bubotmin
bubovrm9
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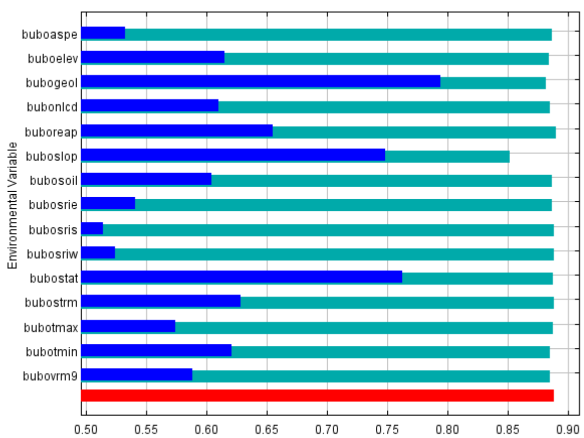
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
test gain

(c)


Environmental Variable
buboaspe
buboelev
bubogeol
bubonlcd
buboreap
buboslop
bubosoil
bubosrie
bubosris
bubosriw
bubostat
bubostrm
bubotmax
bubotmin
bubovrm9
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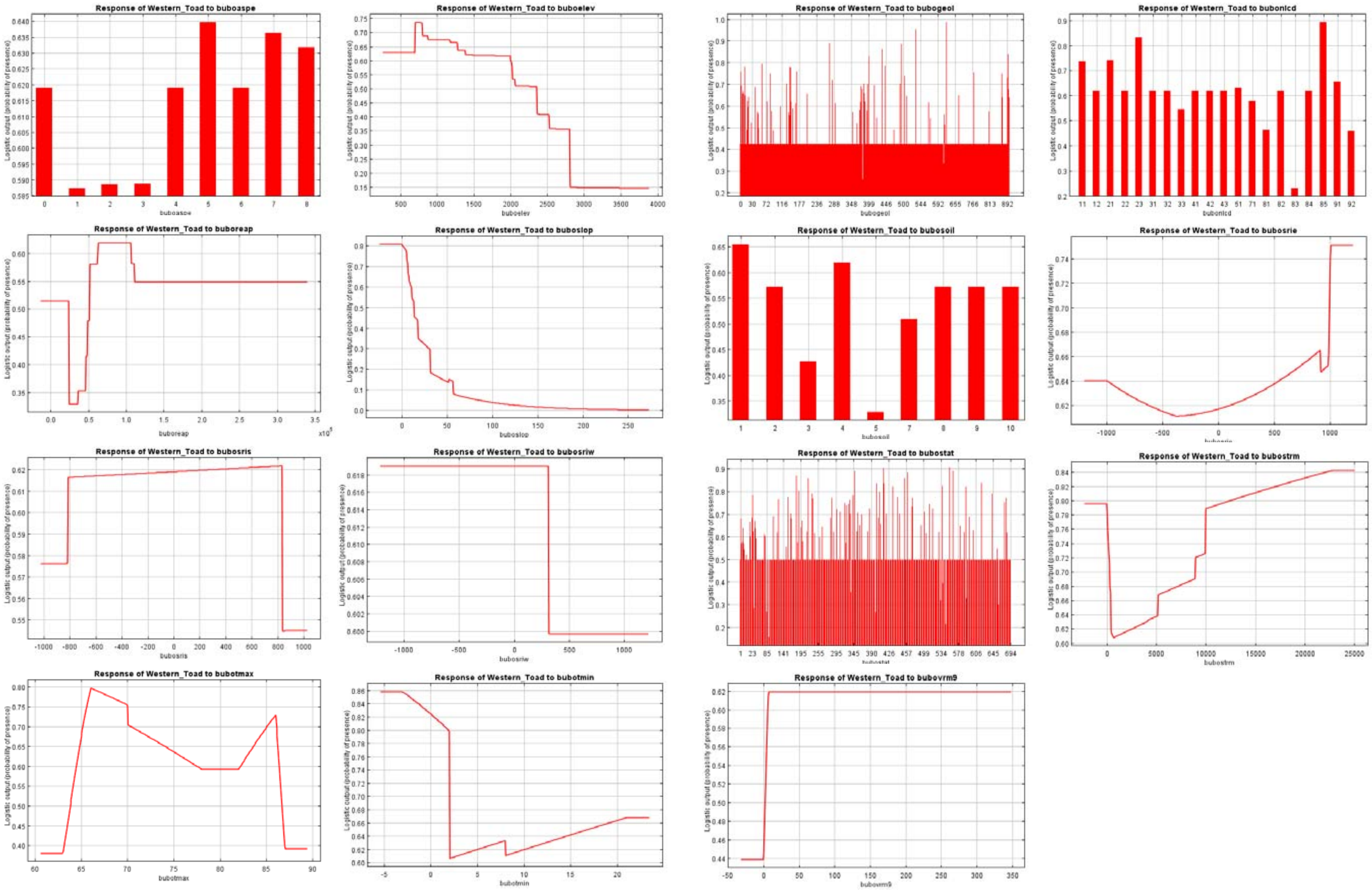
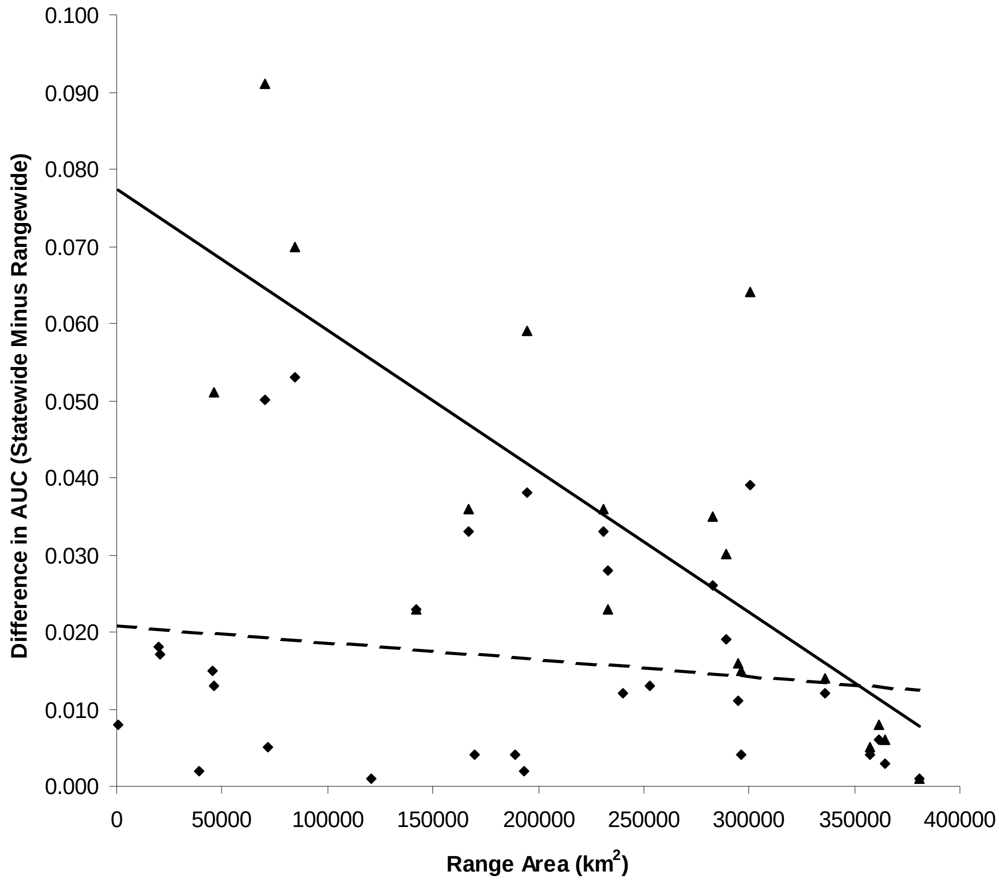
0.60
0.65
0.70
0.75
0.80
0.85
0.90
AUC


Response of Western_Toad to buboaspe
Response of Western_Toad to buboelev
Response of Western_Toad to bubogeol
Response of Western_Toad to buboniccd
Response of Western_Toad to buboreap
Response of Western_Toad to buboslop
Response of Western_Toad to bubosoil
Response of Western_Toad to bubosrie
Response of Western_Toad to bubosris
Response of Western_Toad to bubosriw
Response of Western_Toad to bubostat
Response of Western_Toad to bubostrm
Response of Western_Toad to bubotmax
Response of Western_Toad to bubotmin
Response of Western_Toad to bubovrm9
Logistic output (probability of presence)



0.100
0.090
0.080
0.070
0.060
0.050
0.040
0.030
0.020
0.010
0.000
Difference in AUC (Statewide Minus Rangewide)
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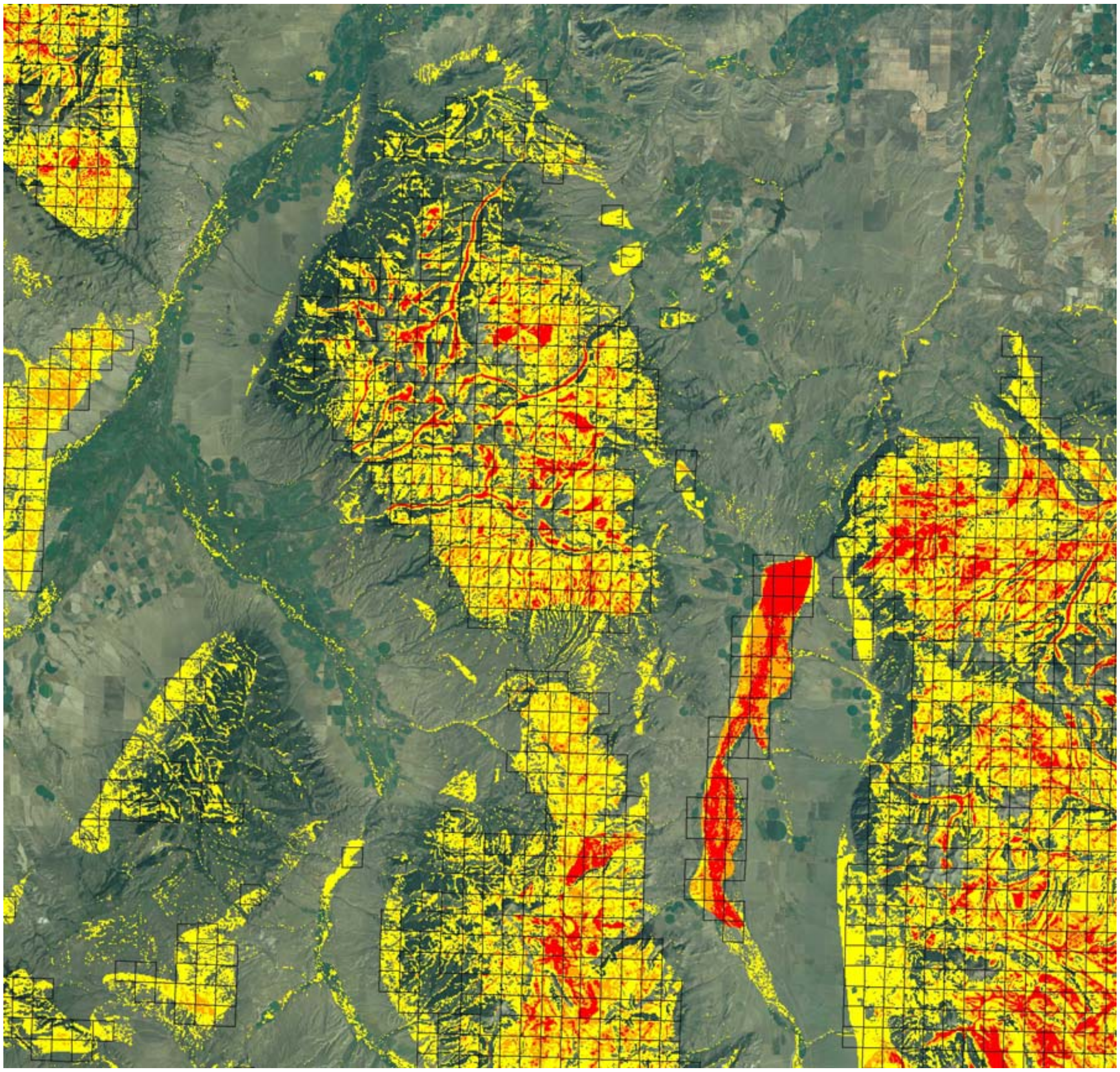
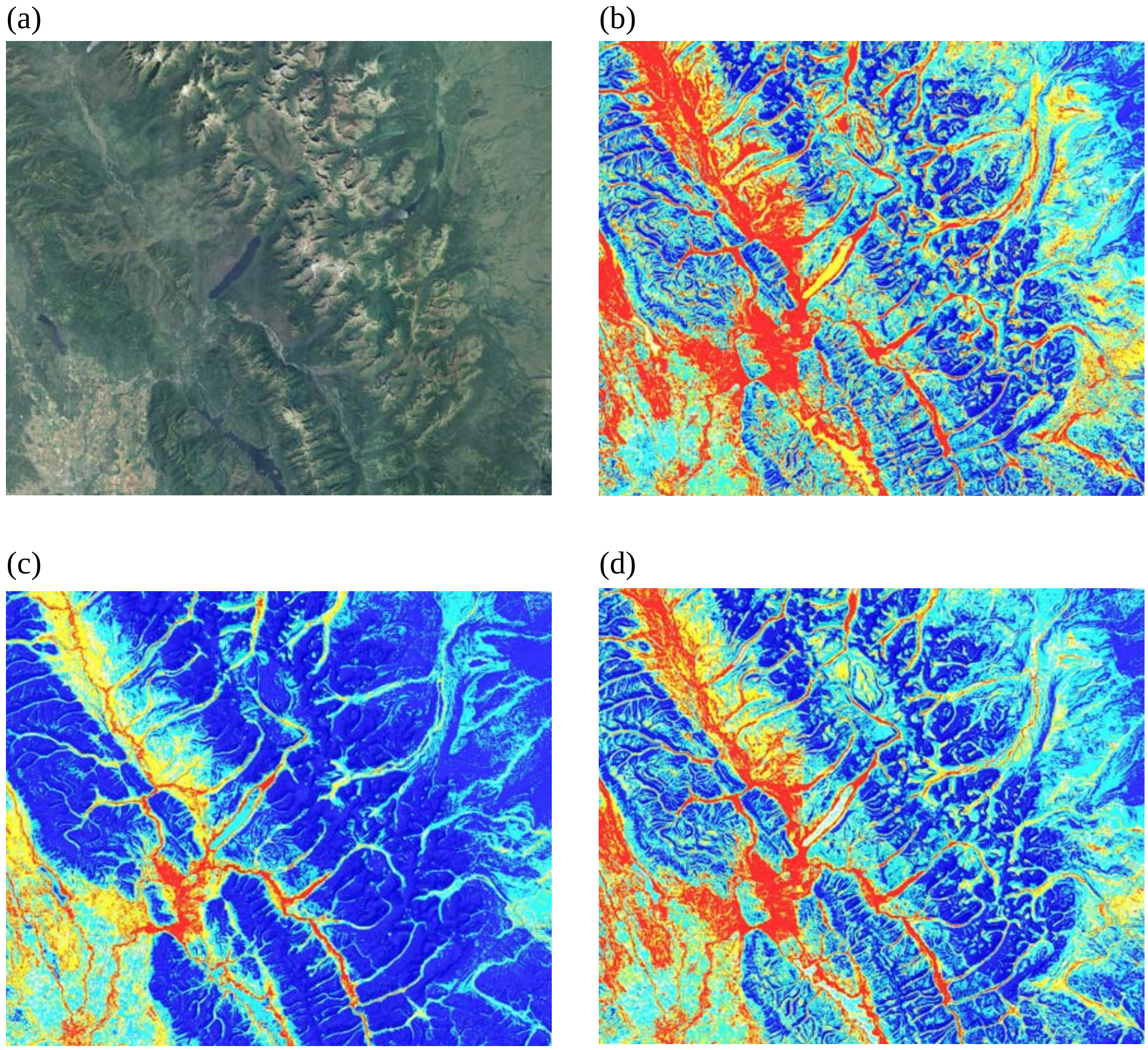
50000
100000
150000
200000
250000
300000
350000
400000
Range Area (km2)




(a)
(b)
(c)
(d)

CHAPTER 4

THE PATHOGENIC AMPHIBIAN CHYTRID FUNGUS, *BATRACHOCHYTRIUM DENDROBATIDIS,* IN MONTANA, USA

Introduction

Emerging infectious diseases are one of the greatest threats to global biodiversity and the health of human populations (Daszak et al. 2000, Jones et al. 2008). The chytrid fungus, *Batrachochytrium dendrobatidis* (hereafter Bd), has emerged as one of the greatest threats to amphibian species (Longcore et al. 1999). Bd has been associated with die-offs in Australia (Berger et al. 1998), Central America (Lips 1999, Lips et al. 2006), South America (Carnaval et al. 2006), Europe (Bosch et al. 2001, Garner et al. 2005), and western North America (Green and Kagarise Sherman 2001, Muths et al. 2003) and threatens hundreds of species with extinction (Stuart et al. 2004). However, Bd has also been reported as widespread in a number of regions not associated with amphibian declines such as Africa and the northeastern United States (Weldon et al. 2004, Longcore et al. 2007). In Montana, almost 60 percent of amphibians are state Species of Concern (MNHP and MFWP 2009) and evidence indicates that both the Western Toad (*Bufo boreas*) and Northern Leopard Frog (*Rana pipiens*) underwent declines in western Montana in the early to mid 1980s (Maxell et al. 2003, Werner 2003, Werner et al. 2004, Maxell et al. 2009).

Methods

In order to examine the potential timeline of arrival of Bd in Montana, we took tissue samples (toes from forelimbs or 5 x 5 mm ventral pelvic dermal patches) from 104 post-metamorphic museum voucher specimens at the Phil L. Wright Zoological Museum at the University of Montana and the zoology collection at Montana State University.

Voucher specimens were chosen in order to complement our recent sampling (Figure 4.1), encompass a broad date range (1899 to 1983), and included 25 *B. boreas*, 5 Woodhouse's Toad (*Bufo woodhousii*), 38 Columbia Spotted Frog (*Rana luteiventris*), and 36 *R. pipiens*.

In order to begin to understand the current distribution of Bd in Montana we collected swabs of ventral tissues and toes opportunistically in association with a variety of fieldwork assessing the distribution, status, and ecology of Montana's amphibians between 1998 and 2008 (e.g., Chapter 2, Werner 2003, Muths et al. 2008, Maxell et al. 2009). We collected swabs of ventral tissues from 465 live post-metamorphic individuals, clipped toes from 96 post metamorphic individuals found live, and clipped toes from 16 post metamorphic individuals found dead (only 4 of these were associated with mass mortality events). Samples were collected between 6 April and 23 September from 1998 to 2008 with 58 percent of samples collected in 2004, 17 percent in 2008, 11 percent in 2005 and 1 to 4 percent in other years. We targeted *B. boreas* and *R. luteiventris* most frequently for Bd sampling because of concerns over the status of *B. boreas* and the relatively high frequency with which post-metamorphic *R. luteiventris* were encountered (Table 4.1). In order to prevent the spread of Bd and other fungal and viral pathogens, field equipment and clothing that contacted water or mud was washed and then decontaminated with 10 percent bleach or Sparquat® between each watershed where samples were taken (Johnson et al. 2003, Johnson and Speare 2003, 2005). All tissues and swabs were stored at room temperature in microcentrifuge tubes filled with 95 percent ethanol. We typically worked and collected samples at lower elevations earlier in the year and progressively worked at higher elevations as the field season progressed, but we sampled the full range of elevations present in Montana in both June and July during the study. Swabs and toes from older museum specimens, live animals, and toes from recently collected voucher specimens were all collected using sterile procedures and were analyzed by Pisces Molecular (J. Wood, Boulder, Colorado, USA) for the presence of Bd using PCR (Annis et al. 2004). We also collected the following animals and shipped them to the National Wildlife Health Center for histopathology by DEG using the methods described in Green and Kagarise Sherman (2001): 5 healthy post-metamorphic

B. boreas on 5 July 2000 and 8 healthy post-metamorphic *R. luteiventris* on 2 July 2001 in Glacier National Park, 6 recently metamorphosed *B.* boreas from a mass mortality event at Schoolmarm Lake in southern Ravalli County on 2 August 2001 (pers. obs.), 7 *R. luteiventris* adults from a mass mortality event in the upper portions of the North Fork of Sweeney Creek in Ravalli County on 8 June 2001 (pers. obs.), and 2 larval *R. luteiventris* from a mass mortality event in Gallatin County near Yellowstone National Park on 11 July 2001 (Eric Atkinson, Marmot's Edge Conservation, pers. comm.).

To examine the potential correlation of human travel corridors used by humans or wildlife with the presence of Bd, we measured Euclidean distances from sample locations to nearest roads in ArcMap 9.2 using the 2000 census topologically integrated geographic encoding and referencing system (TIGER) roads data. We classified habitats and levels of human activity at sample locations based on field notes of the observer and high-resolution aerial photographs for 2005 from the National Agricultural Imagery Program (NAIP). We classified habitats into upland terrestrial sites, ephemeral lentic sites, permanent lentic sites with vegetation, permanent lentic sites without emergent vegetation, and lotic sites. We classified sites as high, moderate, or low human activity based on proximity to areas of human activity. For example, a site within a residential area or a commonly used county road or interstate highway was classified as high human activity. A site adjacent to a U.S. Forest Service road receiving regular vehicle traffic was classified as moderate human activity. A site 20 km from the nearest road within a large wilderness complex, but still associated with regular human activity because of its proximity to a commonly used pack trail was also classified as moderate human activity. However, a site 5 km from a road without any trail access was classified as low human activity. We recorded all elevations from digital 1:24,000-scale U.S. Geological Survey quadrangle maps.

We used one-tailed z-ratio tests to evaluate the statistical significance of differences between independent proportions of samples testing positive for Bd in our opportunistic samples. Pearson's r was used to estimate correlations of the proportion of samples testing positive for Bd between two species sampled at the same sites.

Results

Approximately 70 percent of the 104 museum voucher specimen tissues collected between 1899 and 1983 failed to fully digest and PCR failed to detect Bd in those that did digest; likely as a result of fixation in formalin at the time of preservation (Hyatt et al. 2007). Similarly, only 2 (10 percent) of the samples collected between 1998 and 2008 from voucher specimens that had been fixed or stored in formalin tested positive with PCR as compared to 36 percent of toe or ventral tissues and 38 percent of tissue swabs stored immediately in ethanol and tested with PCR. Histology detected Bd in 46 percent of the 13 live post-metamorphic animals from Glacier National Park. However, histology did not detect Bd in any of the animals collected at mass mortality events in Ravalli and Gallatin Counties. Both the mass mortality of *B. boreas* juveniles at Schoolmarm Lake in southern Ravalli County and the mass mortality of *R. luteiventris* in Gallatin County near Yellowstone National Park were due to ranavirus (David E. Greene, National Wildlife Health Center, pers. comm.). The die-off of *B. boreas* juveniles represents the first case of a die-off of a bufonid due to ranavirus to be diagnosed by the National Wildlife Health Center. No pathogens were detected in the tissues of the *R. luteiventris* adults found at the mass mortality event in the North Fork of Sweeney Creek in Ravalli County and we presume these mortalities are the result of winter kill.

Bd was detected in 6 of the 9 species for which samples were collected between 1998 and 2008 and analyzed with either PCR or histology (Table 4.1). Those species that did not have any samples test positive for Bd all had small sample sizes, making it likely that we failed to detect Bd in these species by chance alone. Samples testing positive for Bd were widespread across Montana and at elevations up to 2,524 m (Figures 4.1). There was broad overlap in the point estimates and 95% confidence intervals for both the proportions of samples and sites testing positive for Bd across all elevation classes sampled. The proportions of samples and sites that tested positive for Bd in April, May, and June, a wetter and cooler period of the active season (0.43 for samples and 0.56 for sites), were both significantly higher than they were in July, August, and September, a drier and warmer period of the active season (0.34 for samples and 0.32 for sites) ($z \geq$

1.99, $p \leq 0.023$). The proportion of samples and sites testing positive for Bd within 1 km of a road (0.40 for samples and 0.50 for sites), were both significantly higher than they were at distances greater than 1 km from roads (0.26 for samples and 0.27 for sites) ($z \geq 2.94$, $p \leq 0.002$). However, Bd was still broadly distributed in areas with lower road densities and one sample from Glacier National Park that was 17.5 km from the nearest road tested positive. There was no correlation between human activity, as classified from our field notes and NAIP imagery, and the proportion of sites testing positive for Bd (37 percent of high activity sites, 38 percent of moderate activity sites, and 31 percent of low activity sites) ($z \leq 0.211$, $p \geq 0.17$). Permanent lentic sites with emergent vegetation tested positive for Bd at a significantly higher rate (0.42) than ephemeral lentic sites (0.33) ($z = 1.65$, $p \leq 0.049$). Other major habitat types all had much lower proportions of sites test positive for Bd (permanent lentic sites without emergent vegetation = 0.14, lotic sites = 0.17, and terrestrial habitats = 0.14).

At 12 sites where we sampled 2 species, Bd detection rates between species were moderately positively correlated (Pearson's $r = 0.32$) for the 1 to 20 individuals that were sampled for each species (Table 4.2). Nine of these sites are permanent sites with emergent vegetation that may serve as aquatic overwintering sites for amphibians as well as a permanent reservoir for Bd. The mean proportion of samples testing positive at these 9 permanent sites was 0.58 as compared to 0.40 at the 3 ephemeral sites where multiple species were sampled. *B. boreas* was sampled for Bd at 5 permanent and 3 ephemeral lentic sites where other species where present (7 with *R. luteiventris* and 1 with Long-toed Salamander (*Ambystoma macrodactylum*)) (Table 4.2). At these sites, Bd detection rates in *B. boreas* were significantly higher (0.71) as compared to all other permanent and ephemeral lentic sites where *B. boreas* were sampled (0.38) ($z = 3.67$, $P = 0.0001$). In contrast, *R. luteiventris* was sampled for Bd at 6 permanent and 2 ephemeral lentic sites where other species were present (7 with *B. boreas* and 1 with *R. pipiens*) (Table 4.2). At these sites, Bd detection rates in *R. luteiventris* were significantly lower (0.20) as compared to all other permanent and ephemeral lentic sites where *R. luteiventris* were sampled (0.46) ($z = 3.45$, $P = 0.003$).

Discussion

The widespread occurrence of Bd in samples collected in 6 species across Montana in the past decade at a variety of elevations, habitat types, and at sites quite distant from human activity clearly indicates that Bd has been present in Montana for a significant period of time. However, our data do not provide any direct evidence for Bd as the cause of declines in *B. boreas* and *R. pipiens* populations in western Montana in the 1980s (Chapter 2, Maxell et al. 2003, Werner 2003). If Bd was the cause of declines in *B. boreas*, it is possible that the stability we have observed in breeding populations over the last 10-15 years (Maxell et al. 2009, BAM, BRH, JKW, PSC, unpublished data) is a result of selection for populations with greater resistance. Severe declines accompanying the outbreak of Bd have been followed by slow increases to pre-decline numbers in at least one species in the rainforests of eastern Australia (McDonald et al. 2005). In the case of *R. pipiens*, declines in populations in western Montana have resulted in the near extirpation of the species while populations in eastern Montana remain widespread and relatively common (Werner 2003, Chapter 2). Eight antimicrobial peptides have been isolated from the skin of *R. pipiens* that may confer resistance to Bd (Rollins-Smith et al. 2002), but populations in western Montana may have produced a different cocktail or volume of these peptides that afforded less resistance. It is also possible that unknown natural or anthropogenic stressors on populations in western Montana led to a weakened immune response to Bd (e.g., Maniero and Carey 1997, Simmaco 1997).

It is interesting that while we detected Bd in or on the skins of animals showing no signs of morbidity, we did not detect Bd in association with any of the mass mortality events where we were able to collect specimens. With the exception of Tiger Salamanders (*Ambystoma tigrinum*), which have had mortality events reported at 29 sites across the species' known range in Montana between 2002 and 2008 (Figure 4.2), mass mortality events have rarely been reported for Montana amphibians and populations of most species are believed to be stable (Chapter 2, Maxell et al. 2009). None of these mortality events was able to be screened for pathogens with histopathology or PCR, but they were associated with skin lesions similar in appearance to those associated with Bd infections in another ambystomatid (Brodman and Briggler 2008). *A. tigrinum* has shown resilience to Bd infection elsewhere (Davidson 2003, Berger et al. 2005) so we believe these mortality events are most likely due to *Ambystoma tigrinum* virus (ATV), which has caused mass mortalities in *A. tigrinum* populations in surrounding states and provinces (Bollinger et al. 1999, Jancovich et al. 2005, Schock et al. 2008).

Because we collected tissue samples opportunistically in association with a variety of fieldwork, patterns in our data should only be regarded as hypotheses to be tested with more rigorous sampling schemes. However, our findings are generally consistent with other literature on Bd in temperate North America (Longcore et al. 2007, Pearl et al. 2007, Muths et al. 2008). Habitats with longer hydroperiods and cooler portions of the active season that would better support the transmission and growth of Bd were associated with higher rates of detection. However, there was no trend evident in the proportion of sites or samples testing positive for Bd at various elevations that might be associated with temperature as reported by Muths et al. (2008) for the Rocky Mountains from Montana to Colorado. An association between Bd and human travel corridors has not been reported before to our knowledge, but there is evidence for this in our finding of significantly higher detection rates for Bd within 1 km of a road. Since our data do not support a direct connection between levels of human activity and Bd detection rates, this may indicate that human travel corridors facilitate non human transmission of Bd. A variety of wildlife make extensive use of human trails, closed roads, and minor roads and Bd would more easily survive between water crossings if travel costs and times were reduced for these species.

The patterns of higher Bd detection rates in *B. boreas* at sites where they co-occurred with other species relative to all other lentic habitats where they were sampled (Table 4.2) may indicate a pattern of cross infection between species that has important implications for conservation. *R. luteiventris* may be an important reservoir and vector for Bd in Montana because the species is highly aquatic and breeds in 65 percent of watersheds and 29 percent of lentic sites across western Montana (Chapter 2, Werner et al. 2004, Maxell et al. 2009). *R. luteiventris* has been found to produce two skin peptides ranatuerin-2La and esculentin-2L that have strong fungicidal impacts on Bd, potentially allowing them to carry the pathogen without succumbing to it (Rollins-Smith et al. 2002). *B. boreas* also produces skin peptides that have been shown to be highly potent at inhibiting Bd, but they produce very small quantities of these peptides and, as a result, their overall resistance to Bd is poor (Muths et al. 2003, Carey et al. 2006). Bufonids are experiencing the highest rate of disease related decline of any amphibian family (Stuart et al. 2004). Their weak defense against Bd is potentially a result of their thick keratin rich epidermal skin layer and the less complex cocktail of skin peptides the family produces (Woodhams et al. 2006).

The widespread nature of Bd in Montana amphibians indicates the need for studies of the population level demographic impacts of Bd as a function of intra and interspecific densities, phenology, thermoregulatory behaviors, environmental conditions, factors that provide differential resistance to pathogens such as the production of skin peptides, and a variety of natural and anthropogenic stressors (e.g., Woodhams et al. 2003, 2007a, 2007b). All of these topics have important implications for efforts to maintain existing populations and reintroduce species such as *R. pipiens* into regions where they have been extirpated. The known presence of Bd, the continued specter of other novel pathogens, the regular occurrence and unknown causes of mass mortality events in *A. tigrinum*, and the potential role of humans in transmission of Bd, all raise the importance of educating a variety of personnel on the importance of following washing and decontamination protocols (Johnson et al. 2003, Johnson and Speare 2003, 2005). Sale of live amphibians from the pet trade may represent a significant threat to native amphibian populations through the potential introduction of novel pathogens so we recommend establishment of procedures to ensure that animals in pet stores are pathogen free. Montana state law bans the unauthorized introduction of exotic species (Montana Code Annotated 87-5-705), but additional efforts should be made to educate the public about potential consequences. Finally, exotic American Bullfrogs (*Rana catesbeiana*) continue to be introduced and are expanding their range in Montana (Maxell et al. 2003, 2009). *R. catesbeiana* are a known reservoir and vector for the spread of Bd (Weldon et al. 2004, Garner et al. 2006) so we encourage control measures in areas with established populations.

Table 4.1. Proportion of sites and samples testing positive for Bd and mean positive rate across sites for 9 amphibian species at 164 locations across Montana between 1998 and 2008.

Species	N Sites/Samples	Proportion (95% CI) Sites Positive	Proportion (95% CI) Samples Positive	Mean (SE) Positive Rate Across Sites
Long-toed Salamander (*Ambystoma macrodactylum*)	2 / 7	0 (-)	0 (-)	0 (-)
Tiger Salamander (*Ambystoma tigrinum*)	3 / 7	0.33 (0.0 – 1.0)	0.14 (0.0 – 0.49)	0.33 (0.33)
Coeur d'Alene Salamander (*Plethodon idahoensis*)	3 / 5	0 (-)	0 (-)	0 (-)
Western Toad (*Bufo boreas*)	85 / 235	0.45 (0.35 – 0.56)	0.40 (0.34 – 0.46)	0.35 (0.05)
Great Plains Toad (*Bufo cognatus*)[1]	2 / 2	0 (-)	0 (-)	0 (-)
Woodhouse's Toad (*Bufo woodhousii*)	13 / 33	0.23 (0.03 – 0.51)	0.09 (0.01 – 0.20)	0.05 (0.03)
Boreal Chorus Frog (*Pseudacris maculata*)	8 / 8	0.38 (0.08 – 0.80)	0.38 (0.08 – 0.80)	0.38 (0.18)
Columbia Spotted Frog (*Rana luteiventris*)	45 / 259	0.49 (0.35 – 0.65)	0.41 (0.35 – 0.47)	0.34 (0.06)
Northern Leopard Frog (*Rana pipiens*)	15 / 34	0.47 (0.24 – 0.76)	0.29 (0.15 – 0.46)	0.34 (0.11)

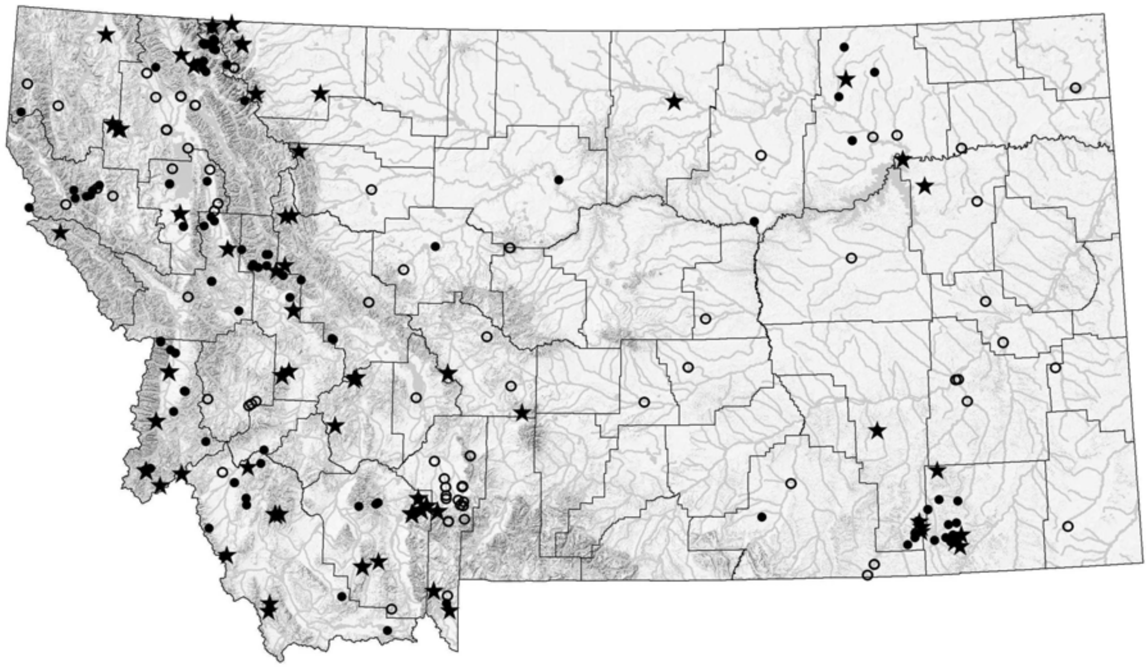
Table 4.2. Proportion of animals testing positive at sites with samples taken from multiple species during a single survey date. A maximum of two species was sampled per site. Overall Pearson's correlation r = 0.32 and covariance = 0.04 regardless of species. *R. luteiventris* positive test rates were lower than *B. boreas* positive test rates at all but one site where both were sampled, potentially indicating that *R. luteiventris* is better able to avoid or clear Bd infections.

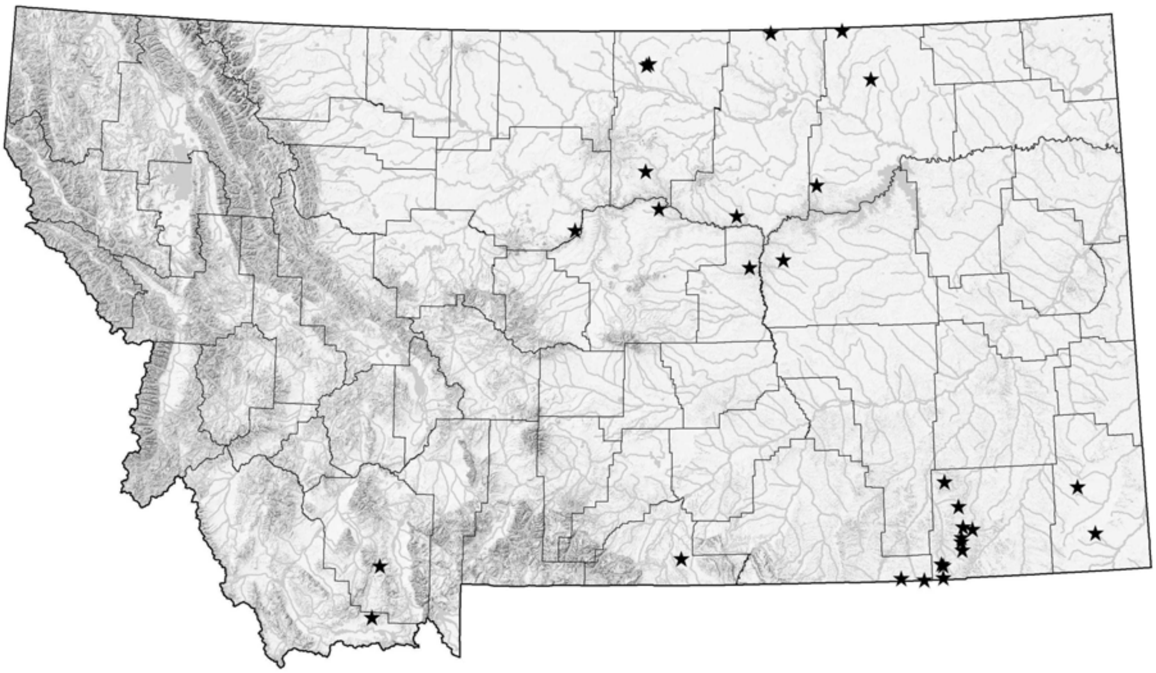
Site ID	Species 1	Proportion Positive (N)	Species 2	Proportion Positive (N)
38	*B. boreas*	0.4 (5)	*R. luteiventris*	0.4 (5)
41	*B. boreas*	0.86 (14)	*R. luteiventris*	0.0 (3)
55	*B. boreas*	0.2 (5)	*R. luteiventris*	0.0 (1)
76	*B. boreas*	1.0 (1)	*R. luteiventris*	0.5 (4)
77	*B. boreas*	1.0 (1)	*R. luteiventris*	0.22 (9)
82	*B. boreas*	1.0 (5)	*R. luteiventris*	0.4 (5)
86	*B. boreas*	1.0 (3)	*R. luteiventris*	0.5 (4)
102	*B. boreas*	0.0 (1)	*A. macrodactylum*	0.0 (6)
199	*B. woodhousii*	0.0 (1)	*A. tigrinum*	1.0 (1)
23	*P. maculata*	1.0 (1)	*R. pipiens*	1.0 (1)
201	*P. maculata*	0.0 (1)	*R. pipiens*	0.0 (2)
190	*R. luteiventris*	0.0 (20)	*R. pipiens*	0.0 (10)

Figure Legends

Figure 4.1. Spatial distribution of samples collected across Montana. Samples from museum specimens collected between 1899 and 1983 all tested negative for Bd via PCR (open circles). Samples collected between 1998 and 2008 that tested positive and negative for Bd are represented by black stars and solid black circles, respectively.

Figure 4.2. Lentic sites with mass mortalities of *A. tigrinum* (black stars) detected between 2002 and 2008.





APPENDIX A

Site Data Form for Lentic Breeding Amphibian and Aquatic Reptile Surveys

Locality Information

Date	Observer(s)	Owner	Site Detection: Aerial Photo Topo Map NWI Map Incidental	GPS EPE

Strata Number	HUC Number	Site Number	State	County	Map Name

Locality	T	R	S	Section Description

Map Elevation FT	UTM Zone:	UTM North	UTM East	Survey Type 0 1 2 3 4 5 6 7 8

Habitat Information

Begin Time	End Time	Total Person Minutes of Search	Camera and Photo Number(s)/Description(s)

Site Dry: Y N	Site Origin: Beaver Water Depressional Manmade Other______________	Support Reproduction? Y N	GIS Mapping 0 1 2 3 4 5 6 7

Habitat Type:	Lake/ Pond	Wetland/ Marsh	Bog/ Fen	Backwater/ Oxbow	Spring/ Seep	Active Beaver Pond	Inactive Beaver Pond	Site Multipooled	Ditch/ Puddle	Reservoir/ Stockpond	Well/ Tank

Weather: Clear Partly Cloudy Overcast Rain Snow	Wind: Calm Light Strong	Air Temp °C	Water Temp °C	Water pH

Color: Clear Stained	Turbidity: Clear Cloudy	Water Connectedness: Permanent Temporary Isolated	Water Permanence: Permanent Temporary	Max Depth: < 1 M 1-2 M >2 M	Percent of Site > 2 M 0 1-25 26-50 51-75 76-100

Site Length:	Site Width:	Percentage of Site Searched: 1-25 26-50 51-75 76-100	Percent of Site at ≤ 50 cm Depth: 0 1-25 26-50 51-75 76-100	~ Emergent Veg Area (M^2)

Percent of Site with Emergent Veg: 0 1-25 26-50 51-75 76-100	Percent of Site with Larval Activity: 0 1-25 26-50 51-75 76-100	Rank Emergent Vegetation Species in Order of Abundance: Sedges___Grasses__Cattails___Rushes___Water Lily___Shrubs___Other_____

Primary Substrate of Shallows: Silt/Mud Sand Gravel Cobble Boulder/Bedrock	North Shoreline Characteristics: Shallows Present: Y N Emergent Veg Present: Y N	Distance (M) to Forest Edge:

Grazing Impact None Light Heavy Structure Heavy Structure and Water Heavy Water	Water Dammed/Diverted Y N	Timber Harvest in Area Y N	Mining Activity Y N

Other Human Impacts Or Modifications:	Fish Detected? Y N	Time at First Detection:	Fish Species If Identified:

Fish Spawning Habitat Present? Y N U	Inlet Width:	Inlet Depth:	Inlet Substrate	Outlet Width	Outlet Depth	Outlet Substrate

Species Information

Amphibian Species		Time at first detection	E L M J A	No. Egg Masses		5-20mm larvae	≤10 ≤100 ≤1000 ≤10K >10K
20-50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	>50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	Number Juveniles		Number Adults	
Tissue Number		Voucher Number		Breeding with Fish?	Y N	If breeding with fish is cover present?	Y N
Amphibian Species		Time at first detection	E L M J A	No. Egg Masses		5-20mm larvae	≤10 ≤100 ≤1000 ≤10K >10K
20-50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	>50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	Number Juveniles		Number Adults	
Tissue Number		Voucher Number		Breeding with Fish?	Y N	If breeding with fish is cover present?	Y N
Amphibian Species		Time at first detection	E L M J A	No. Egg Masses		5-20mm larvae	≤10 ≤100 ≤1000 ≤10K >10K
20-50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	>50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	Number Juveniles		Number Adults	
Tissue Number		Voucher Number		Breeding with Fish?	Y N	If breeding with fish is cover present?	Y N
Amphibian Species		Time at first detection	E L M J A	No. Egg Masses		5-20mm larvae	≤10 ≤100 ≤1000 ≤10K >10K
20-50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	>50mm larvae	≤10 ≤100 ≤1000 ≤10K >10K	Number Juveniles		Number Adults	
Tissue Number		Voucher Number		Breeding with Fish?	Y N	If breeding with fish is cover present?	Y N

Reptile Species		Time at first detection	E J A	Number Individuals		SVL in CM		Tissue Number		Voucher Number	
Reptile Species		Time at first detection	E J A	Number Individuals		SVL in CM		Tissue Number		Voucher Number	
Reptile Species		Time at first detection	E J A	Number Individuals		SVL in CM		Tissue Number		Voucher Number	

Site Map For Lentic Breeding Amphibian and Aquatic Reptile Surveys

Grid Scale:

N↑

***** Indicate the following locations on the map: **T** = temperature, **G** = GPS reading, **C** = clinometer reading, and **P→** = photo locations and directions of photos. Indicate area with emergent vegetation with cross-hatching and indicate a 2-meter depth contour with a dashed line.

Other Notes:

Compass Bearing	70°	90°	110°	130°	150°	170°	190°	210°
Inclination (degrees)								

Definitions of Variables on Lentic Breeding Amphibian Survey Data Sheet

Locality Information

Date: Use MM-DD-YY format (e.g. 5/12/00 for May 12 of 2000).
Observers: List names or initials of individuals involved with survey of this site and circle the name of the recorder.
Owner: Use abbreviation of the government agency responsible for managing the land you surveyed. (e.g. USFS, BLM). If private land was surveyed list the owner's full name to indicate that you did not trespass.
Site Detection: Was site detected on aerial photo, topographic map, NWI map, or was it observed incidentally while in the field.
GPS EPE: The estimated positional error reported by the GPS receiver in meters.
Strata Number: The sample strata in which the 6^{th} level HUC watershed lies (one of nine defined in western Montana).
HUC Number: The sample number of the 6^{th} level HUC in one of the nine sample strata defined for western Montana.
Site Number: The number pre-assigned to the water body within each 6^{th} level HUC. If the water body was not pre-assigned a number because it was not on topographic maps or aerial photos then assign it a sequential number and draw it on the topo map.
State: Use the two-letter abbreviation.
County: Use the full county name.
Map Name: List the name of the USGS 7.5-minute (1:24,000 scale) topographic quadrangle map.
Locality: Describe the specific geographic location of the site so that the type of site is described and the straight-line air distance from one or more permanent features on a 7.5-minute (1:24,000 scale) topographic map records the position of the site (e.g., Beaver pond, 1.5 miles south of Elephant Peak and 1.3 miles east of Engle Peak).
T: Record the Township number and whether it is north or south.
R: Record the Range number and whether it is east or west.
S: Record the Section number.
Section Description: Describe the location of the site at the ¼ of ¼ section level (e.g., SENE indicates SE corner of NE corner).
Map Elevation: The elevation of the site as indicated by the topographic map in feet (avoid using elevations from a GPS)
UTM Zone: Universal Transverse Mercator zone recorded on the topographic map. Use NAD 27 as the map and GPS datum.
UTM North: Universal Transverse Mercator northing coordinate in meters as recorded on the topographic map or GPS receiver. Be sure to note any major differences between UTM coordinates on the map and those on the GPS receiver.
UTM East: Universal Transverse Mercator easting coordinate in meters as recorded on the topographic map or GPS receiver. Be sure to note any major differences between UTM coordinates on the map and those on the GPS receiver.
Survey Type: Circle the appropriate number defined as follows: 0 = private land so site was not surveyed; 1 = site not surveyed due to logistics; 2 = site is a lotic spring/seep not worth future survey; 3 = lentic site that is worth future survey; 4 = misidentified as a potential lentic site on the aerial photograph or on the topographic map (e.g., a shadow from a tree or a talus slope) and not worth future survey; 5 = inactive beaver dam that now only has lotic habitat and is not worth future survey; 6 = only lotic habitat is present and the site is not worth future survey, but it appears possible that the meadow was an historic beaver dam complex; 7 = a lentic site because it would hold water for at least a short time period during wetter conditions, but it is not worth future survey because it would never hold enough water long enough to support amphibian reproduction; 8 = site is not worth future survey for some reason other than those listed above.

Habitat Information

Begin Time: List the time the survey began in 24-hour format.
End Time: List the time the survey ended in 24-hour format.
Total Person Minutes of Search: Record the total person minutes the site was searched (e.g. if one person surveys for 15 minutes and another surveys for 30 minutes, but takes 5 minutes to measure a specimen the total person minutes is 40 minutes).
Camera and Photo Number(s) / Description (s): Identify the camera and the number of the photo as viewed on the camera's view screen and a description of the contents of the photograph (e.g., 13 = 1 x ASMO larvae and 14 = 1 x habitat). Take photos of all portions of the site and anything else that may be of interest (e.g., areas with fish versus areas with amphibians).
Site Dry: Circle whether the site was dry or not at the time of the survey.
Site Origin: Circle whether the site origin is glacial, beaver, water (i.e., flooding or spring), depressional, manmade, or describe other origin.
Support Reproduction: Is site capable of supporting reproduction so it is worth resurveying (e.g. in wetter years if now dry)?
GIS Mapping: Circle the appropriate number defined as follows: 0 = site not surveyed; 1 = a 4 in the survey type and site is not worth future survey; 2 = a 2, 5, 6, or 8 in survey type and site is not worth future survey; 3 = 7 in survey type and site is not worth future survey; 4 = a 3 in the survey type and site is dry, but is worth future survey; 5 = a 3 in the survey type and site has ephemeral water and is worth future survey (including high elevation sites that freeze solid); 6 = a 3 in the survey type, site is worth future survey, has emergent vegetation, and has permanent water that lasts all summer long and does not freeze solid in the winter so that it is likely to support aquatic overwintering; 7 = a 3 in the survey type, site is worth future survey, does not have functional amounts of emergent vegetation, and has permanent water that lasts all summer long and does not freeze solid in the winter so that it is likely to support aquatic overwintering.
Habitat Type: Circle the appropriate habitat type of the site being surveyed. If site is multi-pooled water information does not need to be gathered for every pool, but you may wish to record this information on the map. If breeding activity is limited to one pool at a multi-pooled site water information should be recorded for this pool and this should be noted in the comments.
Weather: Circle weather condition during survey.
Wind: Circle wind condition during survey (> 20 mph winds should be classified as strong).
Air Temp: Record air temperature at chest height in the shade. Record temperature in Celsius. °C = (°F – 32)/1.8
Water Temp: Record water temperature where larvae or egg masses are observed or at 2 cm depth 1 meter from the margin of the water body. Record temperature in Celsius. °C = (°F – 32)/1.8
Water pH: Record water pH at the same location water temperature was recorded.
Color: Circle whether the water is clear or stained a tea or rust color from organic acids.

Turbidity: Circle whether water is clear or cloudy.
Water Connectedness: Circle if water body has permanent connection to flowing water (Permanent), is connected to flowing water for a temporary period each year (Temporary), or is never connected to flowing waters or other water bodies (Isolated).
Water Permanence: Circle whether the site contains water throughout the entire year (Permanent), or contains water for only a portion of the year (Temporary).
Max Depth: Circle the category corresponding to the maximum depth of the water body.
Percent of Site > 2 M: Circle the percentage of the site with water depth greater than 2 meters deep.
Site Length: The length of the longest dimension of the standing water body.
Site Width: The width of the second longest dimension of the standing water body.
Percentage of Site Searched: Circle the percentage of the site surveyed.
Percentage of the Site at $\leq$ 50 cm Depth: Circle the appropriate percentage.
Approximate Area with Emergent Veg (M^2): The approximate area of the site that contains emergent vegetation.
Percentage of Site with Emergent Veg: Circle the percentage of the entire site with emergent vegetation.
Percentage of Site with Larval Activity: Circle the percentage of the site where amphibian larvae were observed.
Rank Emergent Veg Species in Order of Abundance: Record the rank order of abundance in front of the 3 most prevalent emergent vegetation species. If the vegetation present is “other” indicate what it is.
Primary Substrate: Circle the substrate that covers the majority of the bottom of the site.
North Shoreline Characteristics: Circle whether shallows and emergent vegetation are present or absent on the north shoreline.
Distance (M) to Forest Edge: Record the closest distance between the water’s edge and the forest margin in meters.
Grazing Impact: Circle the appropriate grazing category defined as follows: no grazing in vicinity of the site; grazing noted in the vicinity of the site, but no major impacts to wetland structure or water quality; heavy structural impacts to site (e.g.,vegetation destroyed creating bare ground, hummocks, pugging, or altered hydroregime); heavy structural impacts and water quality impacted due to animal waste; and water quality impacted due to animal waste.
Water Dammed/Diverted: Circle whether or not water has been dammed or diverted at the site (including blow outs or pits).
Timber Harvest: Circle whether or not timber has been harvested within 200 meters of the site.
Mining Activity: Circle whether or not there is evidence of mining activity within 200 meters of the site.
Other Human Impacts or Modifications: Briefly describe if, how, and when the site has been altered by human activities. If the site has not been altered record none for not altered. If multiple anthropogenic impacts exist document all of these using the back of the data sheet if necessary and qualify approximate timing of impact (e.g., recent versus historic).
Fish Detected?: Circle whether or not fish were detected.
Time at First Detection: If fish were detected, indicate the time in total person minutes of survey when they were first detected.
Fish Species if Identified: List the fish species identified.
Fish Spawning Habitat Present?: Are shallow waters with adequate gravels/cobbles present that would allow salmonid fishes to spawn? An active search for fry is also a good idea.
Inlet Width: What is the average width of the inlet stream in meters?
Inlet Depth: What is the average depth of the inlet stream in centimeters?
Inlet Substrate: What is the primary substrate at the inlet stream (Silt/Mud, Sand, Gravel, Cobble, or Boulder/Bedrock)?
Outlet Width: What is the average width of the outlet stream in meters?
Outlet Depth: What is the average depth of the outlet stream in centimeters?
Outlet Substrate: What is the primary substrate at the outlet stream (Silt/Mud, Sand, Gravel, Cobble, or Boulder/Bedrock)?

Species Information

For each species record the first two letters of the scientific genus and species names for all amphibian and reptile species found at the site (e.g., BUBO for *Bufo boreas*). Record the total number of person minutes of survey required before each life history stage of each species was encountered beside the E (egg), L (larvae), M (metamorph), J (juvenile), or A (adult). Record the number or category of number of each of the specified life history and/or size classes. For amphibians indicate whether they have bred in the same water body where fish are present, and if they have, indicate whether there is protective cover (e.g., extensive shallows with emergent vegetation, a log barrier, talus). Record the tissue number or range of tissue numbers for tissue samples collected (see tissue collection protocols). If the animal was swabbed in preparation for testing the animal for chytrid infection indicate the chytrid sample number in the Tissue Number field. Record the preliminary museum voucher specimen number for voucher specimens collected (see voucher specimen collection protocols).

Site Map for Lentic Breeding Amphibian and Aquatic Reptile Surveys

General: Include a rough sketch of the site including the shape of the site and the shape and spatial relations of surrounding biotic and abiotic features. Indicate the area covered with emergent vegetation with cross-hatching. Indicate a 2-meter depth contour for the water body with a dashed line. Indicate the location where the water temperature was taken, the location where the GPS position was taken, the location where clinometer readings for southern exposure were taken, and the location of any photographs with an arrow indicating the direction in which the photo(s) were taken. Make sure that the orientation of the sketch (i.e. the north arrow) corresponds to the orientation of the site.
Grid Scale: Indicate the approximate scale of the grid lines relative to the site sketched in meters.
Other Notes: Include any other notes of interest in this space. Examples: (1) areas of highest larval density; (2) thoughts on why a species may not have been detected at a site; (3) problems associated with the survey of the site (e.g., dangerous boggy conditions); (4) If a site was dry would it support reproduction during wetter years.
Southern Exposure: From a site on along the northern shoreline that would most likely to be used as an oviposition or larval rearing area (e.g., shallow waters with emergent vegetation in the NW corner of the water body) record the degree inclination from your position to the skyline (e.g., mountain or solid tree line) at each of the eight compass bearings listed. Note that the compass bearings are true north so you will need to adjust your compass according to the map being used to correct for the deviation from magnetic north (15 to 19.5 degrees in western Montana).

APPENDIX B

EVALUATION OF MONTANA'S LENTIC BREEDING AMPHIBIAN AND AQUATIC REPTILE SURVEY METHODOLOGY

In this appendix I: (1) evaluate levels of precision associated with the documentation of local, landscape, and species variables; (2) evaluate times to first detection of amphibian and gartersnake species; (3) make recommendations for collection and analysis of data in the future.

Levels of Precision Associated with Local, Landscape, and Species Variables

Between 1998 and 2004, 250 sites in western Montana were surveyed between 2 and 22 times each (Table B-1; Figure B-1B) using the standardized data form and definitions provided in Appendix A. Of the 250 sites with multiple surveys, 44 had multiple surveys conducted each year over 1 to 3 years (Table A1; Figure 1B). This history of multiple surveys allows the precision associated with documentation of habitat covariates to be evaluated using coefficients of agreement for categorical variables and coefficients of variation for continuous variables (Portney and Watkins 1993). Coefficients of agreement (CA) are calculated as the number of exact agreements in a categorical response divided by the total number of responses. Values for CA range from 1 to 0 indicating complete agreement or a complete lack of agreement amongst responders, respectively, and can be thought of as ranging from 0 to 100 percent agreement in response. Coefficients of variation (CV) are calculated as the standard deviation (SD) divided by the mean (X) of a continuous response variable. Thus, a CV = 1 indicates the standard deviation of the responses to a particular continuous variable was equivalent in magnitude to the mean value of the responses to that variable. Because both CAs and CVs are standardized by dividing by the mean, in the case of CV, or total number, in the case of CA, categorical or continuous variables can readily be compared to other categorical or continuous variables, respectively, and ranked as to the level of precision associated with the variable.

Tables B-2-4 summarize levels of precision associated with categorical and continuous variables defined on the standardized data form in Appendix A for sites highlighted in gray in Table B-1. Variables within these tables are sorted so that variables with the highest levels of precision are at the tops of the tables and variables with the lowest levels of precision are at the bottoms of the tables.

Table B-2 summarizes the degree of variation in responses of field personnel to habitat and species variables that should not vary between years. Because these variables are unlikely to vary between years all responses for these variables were pooled across all surveys and years. All of the categorical variables had high precision across responses with CA values ranging from 0.92 to 0.996, levels of precision that are unlikely to improve, but should be maintained. The level of precision associated with Distance to Forest was also fairly good (CV = 0.47) given that field estimates of distance tend to vary greatly between field crew members during the training period at the beginning of the field season. Regularly pacing out distances as a check on visual estimates throughout the summer seems to provide the best means of ensuring consistency of estimates of distance across observers.

Tables B-3 and B-4 summarize variation in responses to categorical and continuous variables that are likely to vary between years as a result of changes in weather, habitat, or species over time. Each of the variables summarized in these tables has three different measures associated with it depending on whether the measure of precision was calculated only from multiple surveys of the same site within a year (MSSWY), multiple surveys of the same site across years when multiple surveys were conducted each year (MSSAY), or from all surveys conducted at the site across all years (ASAY). These three levels of metrics were calculated in order to evaluate how precision of responses differed within a year versus between years. In general the MSSWY level metrics would be expected to have the highest level of precision since they were only calculated from surveys performed during the same year when habitats or species were most likely to be the same. For this reason, variables are sorted based on the precision of the MSSWY level of each variable in comparison to the MSSWY level of all other variables.

Most of the categorical variables in Table B-3a had high levels of precision with responses agreeing 74 to 99 percent of the time. Those variables that were associated with lower levels of precision were typically associated with estimates of percentages or estimates of distance. Most of the continuous variables in Table B-3b also had high levels of precision. It is not surprising that Fish Detection Time was variable because this would depend upon when each individual happened to first encounter fish. The fact that Area of Emergent vegetation had lower levels of precision is consistent with lower levels of precision being associated with variables involving estimates of distance. While several categorical variables associated with amphibian species in Table B-4a had high levels of precision (CA > 0.9), a few only agreed an average of 60-70 percent of the time. I would speculate that this is a result of variation in detection of different numbers of animals between observers at sites with large amounts of emergent vegetation. Because animals are often hidden from the direct view of field personnel at sites with large amounts of emergent vegetation, it is more likely that different number classes would be reported as a result of different levels of dipnetting effort. If true, this does raise the need to emphasize a consistent systematic approach toward dipnetting wetlands with large amounts of emergent vegetation. Most of the continuous variables in Table B-4b were fairly precise with SD less than the mean in virtually all cases. Variables with comparatively lower levels of precision were often associated with detection time which, for example, might depend on the direction a particular surveyor first approached the site. Other variables in Table B-4b that were associated with comparatively lower levels of precision were species numbers which, as stated earlier, may vary as a result of level of effort in areas where more active searching is necessary such as wetlands with dense emergent vegetation.

Time to First Detection for Amphibian and Gartersnake Species

Histograms of times at first detection for amphibians detected during field surveys (Figure B-1) show that larvae and juveniles or adults are detected within the first 10 minutes of survey approximately 80 percent of the time. This percentage and the general shape of the distribution of detection times remain remarkably consistent across most amphibian species for these life history stages. Larvae and juveniles or adults are almost always detected within 40 to 50 minutes of search time. Frequency distributions for times at first detection for eggs of these species are not as consistent, but eggs are detected within the first 10 minutes of survey 50 percent of the time for most species. Exceptions to this include egg strings of the western toad which may be cryptically wrapped around vegetation in larger wetlands and eggs of boreal chorus frogs which are very small and can be difficult to detect. Eggs are almost always detected within 60 minutes of search. Histograms of times at first detection for terrestrial and common gartersnakes are very similar with detection within the first 10 minutes approximately 55 percent of the time and with almost all detections occurring within 60 minutes of search.

The frequency distribution of total search times at sites where no species were detected (Figure B-2a) is similar in shape to distributions of times at first detection. This may indicate that sites where no species are detected are searched, on average, about the same amount of time as sites where species are detected. However, this does not mean that all sites are searched long enough to detect species present in complex habitats that may conceal them (e.g., Figure B-2b) and in no way ensures that species are detected if present.

Recommendations for Future

Most categorical and continuous variables that are currently being recorded as part of the Montana amphibian inventory program were associated with high levels of precision and do not appear to currently represent a threat to our ability to detect changes in these variables over time. In part this may be a result of the extensive review process all data is currently subjected to, with each site photo and map reviewed against the data for discrepancies as well as internal inconsistencies. However, there is always room for improvement. Variables with lower levels of precision were usually associated with estimates of distances, percentages, or areas. Regularly pacing out distances as a check on visual estimates throughout the summer seems to provide the best means of ensuring consistency of estimates of distance and area across observers. Increasing levels of precision on estimates of percentages might be achieved through office trainings on example site photos.

Recommendations for the amphibian inventory program include: (1) Regularly pacing out distances as a check on visual estimates; (2) hold two day spring training sessions using existing site photos and data sheets to expose field workers to a variety of issues in a common setting where everyone's questions can be addressed; (3) pair new hires with returning personnel; (4) rotate field crew partners on a regular basis throughout the summer in order to ensure that the entire crew retains a collective standard approach; (5) restandardize everyone in the middle of the field season by having them all survey a set of sites to determine detection probabilities and compare responses; (6) work more closely with agency biologists on ways they can use the data.

Table B-1. Summary of numbers of site surveys conducted each year between 1998 and 2004 for lentic sites surveyed more than once during this time period. Shaded Site IDs indicate sites with multiple surveys during at least one year which makes them suitable for assessment of variation in site evaluations and calculation of detection probabilities (Mean and SD = average and standard deviation of number of surveys for each site across years).

	Number of Surveys For Each Lentic Site By Year									
Site ID	1998	1999	2000	2001	2002	2003	2004	Total	Mean	SD
1013006				1			1	2	1.0	0.0
3003002				2	3	4		9	3.0	1.0
3008001			1	1	1			3	1.0	0.0
3008002			1	1	1			3	1.0	0.0
3008003			1	1				2	1.0	0.0
3008004				1	1			2	1.0	0.0
3008005				1	1			2	1.0	0.0
3008006				1	1			2	1.0	0.0
3008007				1	1			2	1.0	0.0
3008008				1	1			2	1.0	0.0
3008009				1	1			2	1.0	0.0
4001001			1			1		2	1.0	0.0
4001002				1		1	1	3	1.0	0.0
4027006			1	1	1	1	1	5	1.0	0.0
4027007			1		1	1	1	4	1.0	0.0
4027024				1		1	1	3	1.0	0.0
4027025				1	1	1	1	4	1.0	0.0
4034001				1	1		1	3	1.0	0.0
4038001			1	1				2	1.0	0.0
4044001			1	1	1	6	9	18	3.6	3.7
4044002			1	1	4	4	8	18	3.6	2.9
4044003			1	1	6	3	9	20	4.0	3.5
4044004			1	1	4	6	8	20	4.0	3.1
4044099			1	1	6	6	8	22	4.4	3.2
4044100				2	6	6	8	22	5.5	2.5
4044101						4	8	12	6.0	2.8
4044102						4	8	12	6.0	2.8
4049023					1		1	2	1.0	0.0
4056001			1		1			2	1.0	0.0
4056002			1		1			2	1.0	0.0
4056003			1		1			2	1.0	0.0
4056004			1		1			2	1.0	0.0
4056005			1		1			2	1.0	0.0
4056006			1		1			2	1.0	0.0
4056007			1		1			2	1.0	0.0
4056008			1		1			2	1.0	0.0
4056009			1		1			2	1.0	0.0
4056010			1		1			2	1.0	0.0
4056011			1		1			2	1.0	0.0
4056012			1		1			2	1.0	0.0
4056013			1		1			2	1.0	0.0
4056014			1		1			2	1.0	0.0
4056015			1		1			2	1.0	0.0
4056016			1		1			2	1.0	0.0
4056017			1		1			2	1.0	0.0
4056018			1		1			2	1.0	0.0

Table B-1 Continued

SITE_ID	1998	1999	2000	2001	2002	2003	2004	Total	Mean	SD
4056019			1		1			2	1.0	0.0
4056020			1		1			2	1.0	0.0
4056021			1		1			2	1.0	0.0
4056022			1		1			2	1.0	0.0
4056023			1		1			2	1.0	0.0
4056024			1		1			2	1.0	0.0
4056026			1		1			2	1.0	0.0
4057011						1	1	2	1.0	0.0
4057020						1	1	2	1.0	0.0
4058001			1	1	6	1		9	2.3	2.5
4058002			1	1	6	1		9	2.3	2.5
4058003			1	1	6	1		9	2.3	2.5
4058004			1	1	6	1		9	2.3	2.5
4058005			1	1	6	1		9	2.3	2.5
4058006			1	1	6	1		9	2.3	2.5
4058007			1	1	6	1		9	2.3	2.5
4058008			1	1	6	1		9	2.3	2.5
4058009			1	1	6	1		9	2.3	2.5
4058010			1	1	6	1		9	2.3	2.5
4058011			1	1	6	1		9	2.3	2.5
4058012			1	1	6	1		9	2.3	2.5
4058013			1	1	6	1		9	2.3	2.5
4058014			1	1	6	1		9	2.3	2.5
4058015			1	1	6	1		9	2.3	2.5
4058066			1	1	1	1		4	1.0	0.0
4058067			1	1	1	1		4	1.0	0.0
4058068			1	1	1	1		4	1.0	0.0
4058069			1	1	1	1		4	1.0	0.0
4058070			1	1	1	1		4	1.0	0.0
4058071			1	1	1	1		4	1.0	0.0
4058072			1	1	1	1		4	1.0	0.0
4058073				1	1	1		3	1.0	0.0
4058074				1	1	1		3	1.0	0.0
4058075				1	1	1		3	1.0	0.0
4058076				1	1	1		3	1.0	0.0
4058077			1	1	1	1		4	1.0	0.0
4058078			1	1	1	1		4	1.0	0.0
4058079			1	1	1	1		4	1.0	0.0
4058080				1	1	1		3	1.0	0.0
4058081			1	1	1	1		4	1.0	0.0
4058082			1	1	1	1		4	1.0	0.0
4058083			1	1	1	1		4	1.0	0.0
4058084			1	1	1	1		4	1.0	0.0
4060006						1	1	2	1.0	0.0
4060009						1	1	2	1.0	0.0
4063001			1	2	2			5	1.7	0.6
4064090				1	1		1	3	1.0	0.0
4072006				1	1			2	1.0	0.0
4078001						1	1	2	1.0	0.0
4993001			1	1	1		7	10	2.5	3.0
4995001			1	1	1	1		4	1.0	0.0
4995002			1	1	1	1		4	1.0	0.0

Table B-1 Continued

SITE_ID	1998	1999	2000	2001	2002	2003	2004	Total	Mean	SD
4995003			1	1	1	1		4	1.0	0.0
4995004			1	1	1	1		4	1.0	0.0
4995005			1	1	1	1		4	1.0	0.0
4995006			1	1	1	1		4	1.0	0.0
4995007			1	1	1	1		4	1.0	0.0
4995008			1	1	1	1		4	1.0	0.0
4995009			1	1	1	1		4	1.0	0.0
4995010			1	1	1	1		4	1.0	0.0
4995011			1	1	1	1		4	1.0	0.0
4995012				1	1	1		3	1.0	0.0
4995013			1	1	1	1		4	1.0	0.0
4995014			1	1	1	1		4	1.0	0.0
4995015			1	1	1	1		4	1.0	0.0
4995016			1	1	1	1		4	1.0	0.0
4995017			1	1	1	1		4	1.0	0.0
4995018			1	1	1	1		4	1.0	0.0
4995019			1	1	1	1		4	1.0	0.0
4995020			1	1	1	1		4	1.0	0.0
4995021			1	1	1	1		4	1.0	0.0
5006001				1		1	1	3	1.0	0.0
5012002						1	1	2	1.0	0.0
5014001			1	1	1	1	1	5	1.0	0.0
5014002				1	1	1		3	1.0	0.0
5014003				1	1	1		3	1.0	0.0
5014004				1	1			2	1.0	0.0
5014005				1	1			2	1.0	0.0
5014006				1	1			2	1.0	0.0
5014010				1	1			2	1.0	0.0
5014011				1	1			2	1.0	0.0
5014012				1	1			2	1.0	0.0
5014013				1	1			2	1.0	0.0
5014014				1	1			2	1.0	0.0
5014015				1	1			2	1.0	0.0
5014016				1	1			2	1.0	0.0
5014017				1	1			2	1.0	0.0
5014018				1	1			2	1.0	0.0
5014019				1	1			2	1.0	0.0
5014020				1	1			2	1.0	0.0
5014021				1	1			2	1.0	0.0
5014022				1	1	1		3	1.0	0.0
5014023				1	1	1		3	1.0	0.0
5014024				1	1	1		3	1.0	0.0
5014025				1	1	1		3	1.0	0.0
5014026				1	1	1		3	1.0	0.0
5014027				1	1	1		3	1.0	0.0
5014028				1	1	1		3	1.0	0.0
5014029				1	1	1		3	1.0	0.0
5014030				1	1			2	1.0	0.0
5017001					1	1		2	1.0	0.0

Table B-1 Continued

SITE_ID	1998	1999	2000	2001	2002	2003	2004	Total	Mean	SD
5017002					1	1		2	1.0	0.0
5026001				1	1			2	1.0	0.0
5026002				1	1			2	1.0	0.0
5026003				1	1			2	1.0	0.0
5026004				1	1			2	1.0	0.0
5026005				1	1			2	1.0	0.0
5999001	1				1	1		3	1.0	0.0
5999008	1						1	2	1.0	0.0
5999010	1				1	1	1	4	1.0	0.0
5999011	1					1		2	1.0	0.0
5999013		1			1	1		3	1.0	0.0
6002008						1	1	2	1.0	0.0
6015009						1	1	2	1.0	0.0
6015010						1	1	2	1.0	0.0
6021011					2			2	2.0	-
6024010						1	1	2	1.0	0.0
6025001					6	6	8	20	6.7	1.2
6025002				1	1	6	8	16	4.0	3.6
6025003				1	5	6	9	21	5.3	3.3
6025004				1	6	6	8	21	5.3	3.0
6025005				1	5	6	8	20	5.0	2.9
6025006				1	1	6	8	16	4.0	3.6
6025007				1	6	6	8	21	5.3	3.0
6025008				1	1	6	8	16	4.0	3.6
6025009				1	1		8	10	3.3	4.0
6025010				1	1			2	1.0	0.0
6025011				1	6	6	8	21	5.3	3.0
6025096							4	4	4.0	-
6025097							4	4	4.0	-
6025098							7	7	7.0	-
6025099					5	6	8	19	6.3	1.5
6025100				1	1	6	8	16	4.0	3.6
6025108						6	8	14	7.0	1.4
6025109					6	6	8	20	6.7	1.2
6025110					6	6	8	20	6.7	1.2
6028073						1	1	2	1.0	0.0
6043001						1	1	2	1.0	0.0
6043002						1	1	2	1.0	0.0
6046007				1			1	2	1.0	0.0
6046016				1			1	2	1.0	0.0
6046017				1			1	2	1.0	0.0
6046020				1			1	2	1.0	0.0
6046021				1			1	2	1.0	0.0
6046099						1	1	2	1.0	0.0
6047001						1	1	2	1.0	0.0
6047005						1	1	2	1.0	0.0
6049001					2			2	2.0	-
6049021					3			3	3.0	-
6049022					2			2	2.0	-
6049023					2			2	2.0	-
6052002						1	1	2	1.0	0.0
6057007						1	1	2	1.0	0.0

Table B-1 Continued

SITE_ID	1998	1999	2000	2001	2002	2003	2004	Total	Mean	SD
8001001			1		1			2	1.0	0.0
8001002			1		1			2	1.0	0.0
8001003			1		1			2	1.0	0.0
9004001			1		1			2	1.0	0.0
12014015					1		1	2	1.0	0.0
15301001				1	1			2	1.0	0.0
15301002				1	1	2	1	5	1.3	0.5
15303001				1	2	2		5	1.7	0.6
15304001				2	1	2		5	1.7	0.6
15305004					1	1		2	1.0	0.0
15400002				2	1	1	1	5	1.3	0.5
15400006						1	1	2	1.0	0.0
15407001		1		1	1	1	1	5	1.0	0.0
15407002		1		1	1	1	1	5	1.0	0.0
15407003		1				1	1	3	1.0	0.0
15407004				1		1	1	3	1.0	0.0
15408003				1	1			2	1.0	0.0
15410001				1	1	1	1	4	1.0	0.0
15413001				1	1	1	1	4	1.0	0.0
15414001					1		1	2	1.0	0.0
15414002					1		1	2	1.0	0.0
15418001				1			1	2	1.0	0.0
15419001				1	1	1	1	4	1.0	0.0
15420001				1	1	1	1	4	1.0	0.0
15424001					1	1		2	1.0	0.0
15428001						1	1	2	1.0	0.0
15428002						1	1	2	1.0	0.0
15428003						1	1	2	1.0	0.0
15504001				1		1		2	1.0	0.0
15505001				1		1		2	1.0	0.0
15506001				1	1	1	1	4	1.0	0.0
15509001					1	1	1	3	1.0	0.0
15510001					1	1	1	3	1.0	0.0
15510002					1	2		3	1.5	0.7
15510003					1	1		2	1.0	0.0
15510004					1	1		2	1.0	0.0
15510005					1	1		2	1.0	0.0
15607002					1	1	1	3	1.0	0.0
15609001					1		1	2	1.0	0.0
15609005					1		1	2	1.0	0.0
15611001				1	1	1		3	1.0	0.0
15612001						1	1	2	1.0	0.0
15612002					1	1	1	3	1.0	0.0
15612004					1	1	1	3	1.0	0.0
15612005					1	1		2	1.0	0.0
15613001					1	1	1	3	1.0	0.0
15613002					1	1	1	3	1.0	0.0
15613003						1	1	2	1.0	0.0
15613004					1	1	1	3	1.0	0.0
15613005					1	1		2	1.0	0.0
15621001				1		1		2	1.0	0.0
Totals	4	4	93	155	348	267	277	1148	164	137.6

Table B-2 Levels of precision associated with documentation of habitat and species variables that should not vary between years using coefficients of agreement (CA) and coefficients of variation (CV) to assess variation in responses to categorical and continuous variables, respectively.

Habitat Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
Mining Activity	CA	233	0.996	0.04	0.5	1
Water Dammed	CA	234	0.99	0.05	0.5	1
Support Reproduction	CA	249	0.99	0.06	0.5	1
Shallows Present on N	CA	229	0.98	0.07	0.5	1
Site Origin	CA	249	0.97	0.1	0.36	1
Timber Harvest	CA	245	0.97	0.1	0.5	1
Primary Substrate	CA	237	0.97	0.11	0.33	1
Fish Detected	CA	232	0.97	0.12	0.5	1
Water Permanence	CA	247	0.96	0.11	0.5	1
Habitat Type	CA	249	0.96	0.12	0.5	1
Fish Spawning Habitat	CA	191	0.96	0.12	0.5	1
Water Connectedness	CA	244	0.95	0.13	0.43	1
Emergent Veg Present N	CA	228	0.95	0.13	0.5	1
Inlet Substrate	CA	56	0.93	0.15	0.5	1
Outlet Substrate	CA	52	0.92	0.18	0.4	1
Fish Species	CA	Too many unidentified trout for evaluation				
Distance to Forest	CV	238	0.47	0.53	0	2.3

[1] Variables are sorted first by method of evaluation and then in descending order from those with higher levels of precision to those with lower levels of precision.

[2] CA values of 1 and 0 indicate complete agreement and a complete lack of agreement, respectively, of values recorded for the variable across all surveys. CV values simply represent the standard deviation divided by the mean. Thus, a CV = 1 indicates the standard deviation of the responses was equivalent in magnitude to the mean value of the responses.

[3] N indicates numbers of sites for which CA or CV could be calculated because of multiple surveys evaluating the variable.

[4] X, SD, Min, and Max are the overall mean, standard deviation, minimum, and maximum values for CA and CV values calculated for sites with multiple surveys where the variable was documented.

Table B-3a Levels of precision associated with documentation of habitat variables that are likely to vary between years using coefficients of agreement (CA) to assess variation in responses to categorical variables.

Habitat Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
Site Dry - MSSWY	CA	80	0.99	0.05	0.67	1
Site Dry - MSSAY	CA	44	0.94	0.12	0.5	1
Site Dry - ASAY	CA	249	0.98	0.09	0.5	1
Grazing Impact – MSSWY	CA	80	0.94	0.15	0.38	1
Grazing Impact – MSSAY	CA	44	0.88	0.18	0.42	1
Grazing Impact - ASAY	CA	233	0.94	0.15	0.42	1
Water Turbidity - MSSWY	CA	76	0.93	0.12	0.63	1
Water Turbidity - MSSAY	CA	43	0.9	0.15	0.5	1
Water Turbidity - ASAY	CA	228	0.94	0.15	0.5	1
Maximum Depth - MSSWY	CA	75	0.87	0.16	0.33	1
Maximum Depth - MSSAY	CA	42	0.86	0.18	0.5	1
Maximum Depth - ASAY	CA	231	0.91	0.17	0.33	1
Water Color - MSSWY	CA	75	0.87	0.15	0.5	1
Water Color - MSSAY	CA	43	0.87	0.16	0.5	1
Water Color - ASAY	CA	228	0.93	0.15	0.5	1
Dominant Emergent Veg - MSSWY	CA	74	0.81	0.2	0.33	1
Dominant Emergent Veg - MSSAY	CA	42	0.83	0.18	0.38	1
Dominant Emergent Veg - ASAY	CA	204	0.91	0.17	0.25	1
Percent Larval Activity - MSSWY	CA	75	0.78	0.21	0.25	1
Percent Larval Activity - MSSAY	CA	42	0.73	0.21	0.25	1
Percent Larval Activity - ASAY	CA	202	0.79	0.23	0.25	1
Percent Emergent Veg - MSSWY	CA	73	0.75	0.2	0.33	1
Percent Emergent Veg – MSSAY	CA	42	0.73	0.21	0.35	1
Percent Emergent Veg – ASAY	CA	230	0.85	0.22	0.33	1
Percent < 50 cm - MSSWY	CA	75	0.74	0.23	0.33	1
Percent < 50 cm - MSSAY	CA	43	0.74	0.24	0.3	1
Percent < 50 cm - ASAY	CA	226	0.86	0.21	0.3	1

[1] Variables are sorted in descending order from those with higher of levels of precision to those with lower levels of precision on the MSSWY method of calculation. MSSWY indicates values were calculated only from multiple surveys of a site conducted within a single year. MSSAY indicates values were calculated from all surveys at sites with multiple surveys conducted within at least one of the years of sampling (corresponds to shaded Site IDs in Table 1). ASAY indicates values were calculated from all surveys conducted across all years (i.e. sites were surveyed multiple times either within years, between years, or both corresponding to all Site IDs listed in Table 1).

[2] CA values of 1 and 0 indicate complete agreement and a complete lack of agreement, respectively, of values recorded for the variable across all surveys.

[3] N indicates numbers of sites for which CA could be calculated because of multiple surveys evaluating the variable.

[4] X, SD, Min, and Max are the overall mean, standard deviation, minimum, and maximum values for CA values calculated for sites with multiple surveys where the variable was documented.

Table B-3b Levels of precision associated with documentation of habitat variables that are likely to vary between years using coefficients of variation (CV) to assess variation in responses to continuous variables.

Habitat Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
Water pH – MSSWY	CV	106	0.05	0.03	0	0.13
Water pH – MSSAY	CV	26	0.06	0.04	0	0.16
Water pH – ASAY	CV	136	0.07	0.08	0	0.4
Inlet Width - MSSWY	CV	25	0.37	0.23	0	0.87
Inlet Width - MSSAY	CV	15	0.38	0.25	0	0.82
Inlet Width - ASAY	CV	56	0.29	0.28	0	1.05
Outlet Width - MSSWY	CV	25	0.4	0.28	0	1.11
Outlet Width - MSSAY	CV	15	0.44	0.21	0.13	0.81
Outlet Width - ASAY	CV	53	0.27	0.3	0	1.18
Inlet Depth - MSSWY	CV	26	0.54	0.36	0.12	1.86
Inlet Depth - MSSAY	CV	14	0.51	0.21	0.12	0.84
Inlet Depth - ASAY	CV	56	0.4	0.39	0	1.4
Outlet Depth – MSSWY	CV	36	0.57	0.33	0	1.86
Outlet Depth – MSSAY	CV	15	0.56	0.22	0.24	1.02
Outlet Depth – ASAY	CV	52	0.41	0.35	0	1.16
Emergent Vegetation Area - MSSWY	CV	73	0.84	0.44	0.13	2.15
Emergent Vegetation Area - MSSAY	CV	42	0.94	0.6	0.13	2.89
Emergent Vegetation Area - ASAY	CV	213	0.45	0.54	0	2.89
Fish Detection Time - MSSWY	CV	4	0.89	0.45	0.3	1.38
Fish Detection Time - MSSAY	CV	3	1.09	0.28	0.89	1.41
Fish Detection Time - ASAY	CV	24	0.73	0.43	0	1.41

[1] Variables are sorted in descending order from those with higher of levels of precision to those with lower levels of precision on the MSSWY method of calculation. MSSWY indicates values were calculated only from multiple surveys of a site conducted within a single year. MSSAY indicates values were calculated from all surveys at sites with multiple surveys conducted within at least one of the years of sampling (corresponds to shaded Site IDs in Table 1). ASAY indicates values were calculated from all surveys conducted across all years (i.e. sites were surveyed multiple times either within years, between years, or both corresponding to all Site IDs listed in Table 1).

[2] CV values simply represent the standard deviation divided by the mean. Thus, a CV = 1 indicates the standard deviation of the responses was equivalent in magnitude to the mean value of the responses.

[3] N indicates numbers of sites for which CV could be calculated because of multiple surveys evaluating the variable.

[4] X, SD, Min, and Max are the overall mean, standard deviation, minimum, and maximum values for CV values calculated for sites with multiple surveys where the variable was documented.

Table B-4a Levels of precision associated with documentation of species variables that are likely to vary between years using coefficients of agreement (CA) to assess variation in responses to categorical variables.

Species Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
RALU Cover from Fish - MSSWY	CA	2	1	-	1	1
RALU Cover from Fish - MSSAY	CA	2	1	-	1	1
RALU Cover from Fish - ASAY	CA	13	0.96	0.14	0.5	1
PSMA Breeding with Fish - MSSWY	CA	22	1	0	1	1
PSMA Breeding with Fish - MSSAY	CA	12	1	0	1	1
PSMA Breeding with Fish - ASAY	CA	16	0.97	0.13	0.5	1
RALU Breeding with Fish - MSSWY	CA	63	0.99	0.05	0.63	1
RALU Breeding with Fish - MSSAY	CA	36	0.99	0.04	0.75	1
RALU Breeding with Fish - ASAY	CA	133	0.95	0.13	0.5	1
AMTI Breeding with Fish - MSSWY	CA	19	0.98	0.08	0.67	1
AMTI Breeding with Fish - MSSAY	CA	11	0.98	0.08	0.75	1
AMTI Breeding with Fish - ASAY	CA	12	0.94	0.16	0.5	1
BUBO Breeding with Fish - MSSWY	CA	8	0.94	0.18	0.5	1
BUBO Breeding with Fish - MSSAY	CA	4	0.88	0.25	0.5	1
BUBO Breeding with Fish - ASAY	CA	48	0.94	0.15	0.5	1
AMMA Breeding with Fish - MSSWY	CA	8	0.94	0.18	0.5	1
AMMA Breeding with Fish - MSSAY	CA	10	0.97	0.11	0.67	1
AMMA Breeding with Fish - ASAY	CA	41	0.98	0.09	0.5	1
RALU Larvae Number Class - MSSWY	CA	38	0.71	0.18	0.33	1
RALU Larvae Number Class - MSSAY	CA	22	0.64	0.2	0.33	1
RALU Larvae Number Class - ASAY	CA	51	0.55	0.18	0.25	1
AMMA Larvae Number Class - MSSWY	CA	8	0.7	0.16	0.5	1
AMMA Larvae Number Class - MSSAY	CA	10	0.69	0.28	0.33	1
AMMA Larvae Number Class - ASAY	CA	35	0.78	0.24	0.33	1
BUBO Larvae Number Class - MSSWY	CA	5	0.69	0.21	0.5	1
BUBO Larvae Number Class - MSSAY	CA	3	0.53	0.13	0.41	0.67
BUBO Larvae Number Class - ASAY	CA	25	0.65	0.22	0.33	1
AMTI Larvae Number Class - MSSWY	CA	11	0.69	0.2	0.33	1
AMTI Larvae Number Class - MSSAY	CA	6	0.56	0.13	0.38	0.71
AMTI Larvae Number Class - ASAY	CA	7	0.55	0.12	0.38	0.71
PSMA Larvae Number Class - MSSWY	CA	19	0.59	0.21	0.25	1
PSMA Larvae Number Class - MSSAY	CA	12	0.52	0.12	0.33	0.67
PSMA Larvae Number Class - ASAY	CA	15	0.55	0.17	0.33	1

Table B-4a Continued

Species Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
AMMA Cover from Fish - MSSWY	CA	0	-	-	-	-
AMMA Cover from Fish - MSSAY	CA	2	1	0	1	1
AMMA Cover from Fish - ASAY	CA	5	1	0	1	1
AMTI Cover from Fish - MSSWY	CA	0	-	-	-	-
AMTI Cover from Fish - MSSAY	CA	0	-	-	-	-
AMTI Cover from Fish - ASAY	CA	0	-	-	-	-
BUBO Cover from Fish - MSSWY	CA	0	-	-	-	-
BUBO Cover from Fish - MSSAY	CA	1	1	-	1	1
BUBO Cover from Fish - ASAY	CA	11	0.95	0.1	0.75	1
PSMA Cover from Fish - MSSWY	CA	0	-	-	-	-
PSMA Cover from Fish - MSSAY	CA	0	-	-	-	-
PSMA Cover from Fish - ASAY	CA	0	-	-	-	-

[1] Variables are sorted in descending order from those with higher of levels of precision to those with lower levels of precision on the MSSWY method of calculation. MSSWY indicates values were calculated only from multiple surveys of a site conducted within a single year. MSSAY indicates values were calculated from all surveys at sites with multiple surveys conducted within at least one of the years of sampling (corresponds to shaded Site IDs in Table 1). ASAY indicates values were calculated from all surveys conducted across all years (i.e. sites were surveyed multiple times either within years, between years, or both corresponding to all Site IDs listed in Table 1). AMMA = Long-toed Salamander (*Ambystoma macrodactylum*), AMTI = Tiger Salamander (*Ambystoma tigrinum),* BUBO = Western Toad (*Bufo boreas*), PSMA = Boreal Chorus Frog (*Pseudacris maculata*), RALU = Columbia Spotted Frog (*Rana luteiventris*), Terrestrial Gartersnake (*Thamnophis elegans*), THSI = Common Gartersnake (*Thamnophis sirtalis*).

[2] CA values of 1 and 0 indicate complete agreement and a complete lack of agreement, respectively, of values recorded for the variable across all surveys.

[3] N indicates numbers of sites for which CA could be calculated because of multiple surveys evaluating the variable.

[4] X, SD, Min, and Max are the overall mean, standard deviation, minimum, and maximum values for CA values calculated for sites with multiple surveys where the variable was documented.

Table B-4b Levels of precision associated with documentation of species variables that are likely to vary between years using coefficients of variation (CV) to assess variation in responses to continuous variables.

Species Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
AMTI J & A Detection Time - MSSWY	CV	1	0.18	-	0.18	0.18
AMTI J & A Detection Time - MSSAY	CV	2	0.09	0.13	0	0.18
AMTI J & A Detection Time - ASAY	CV	2	0.09	0.13	0	0.18
PSMA J & A Detection Time - MSSWY	CV	2	0.34	0.23	0.18	0.5
PSMA J & A Detection Time - MSSAY	CV	2	0.34	0.23	0.18	0.5
PSMA J & A Detection Time - ASAY	CV	2	0.34	0.23	0.18	0.5
THEL Juv & Adult Numbers - MSSWY	CV	24	0.34	0.25	0	0.75
THEL Juv & Adult Numbers - MSSAY	CV	23	0.3	0.25	0	0.74
THEL Juv & Adult Numbers - ASAY	CV	32	0.26	0.25	0	0.74
THSI Juv & Adult Numbers - MSSWY	CV	11	0.4	0.36	0	1.01
THSI Juv & Adult Numbers - MSSAY	CV	19	0.22	0.25	0	0.84
THSI Juv & Adult Numbers - ASAY	CV	27	0.23	0.31	0	1.1
THSI J&A Detection Time - MSSWY	CV	1	0.47	-	0.47	0.47
THSI J&A Detection Time - MSSAY	CV	1	0.47	-	0.47	0.47
THSI J&A Detection Time - ASAY	CV	1	0.47	-	0.47	0.47
THEL J&A Detection Time - MSSWY	CV	6	0.53	0.3	0.18	1.1
THEL J&A Detection Time - MSSAY	CV	5	0.62	0.16	0.47	0.81
THEL J&A Detection Time - ASAY	CV	5	0.62	0.16	0.47	0.81
AMTI Juv & Adult Numbers - MSSWY	CV	16	0.55	0.3	0	1.16
AMTI Juv & Adult Numbers - MSSAY	CV	11	0.65	0.32	0	1.16
AMTI Juv & Adult Numbers - ASAY	CV	11	0.65	0.32	0	1.16
BUBO Egg Numbers - MSSWY	CV	1	0.56	-	0.56	0.56
BUBO Egg Numbers - MSSAY	CV	1	0.56	-	0.56	0.56
BUBO Egg Numbers - ASAY	CV	4	0.45	0.24	0.16	0.71
RALU J & A Detection Time - MSSWY	CV	35	0.62	0.44	0	1.69
RALU J & A Detection Time - MSSAY	CV	26	0.72	0.43	0	1.71
RALU J & A Detection Time - ASAY	CV	43	0.66	0.47	0	1.71
PSMA Larvae Detection Time - MSSWY	CV	19	0.66	0.39	0	1.36
PSMA Larvae Detection Time - MSSAY	CV	12	0.71	0.33	0	1.1
PSMA Larvae Detection Time - ASAY	CV	15	0.72	0.4	0	1.38
RALU Larvae Detection Time - MSSWY	CV	38	0.67	0.43	0	1.65
RALU Larvae Detection Time - MSSAY	CV	22	0.8	0.5	0	1.72
RALU Larvae Detection Time - ASAY	CV	41	0.73	0.46	0	1.72

Table B-4b Continued

Species Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
BUBO Larvae Detection Time - MSSWY	CV	5	0.74	0.43	0	1.04
BUBO Larvae Detection Time - MSSAY	CV	3	1.23	0.21	1.04	1.46
BUBO Larvae Detection Time - ASAY	CV	19	0.77	0.44	0	1.46
AMMA Larvae Detection Time - MSSWY	CV	8	0.74	0.43	0	1.38
AMMA Larvae Detection Time - MSSAY	CV	6	0.8	0.53	0	1.52
AMMA Larvae Detection Time - ASAY	CV	15	0.5	0.52	0	1.52
BUBO Egg Detection Time - MSSWY	CV	1	0.76	-	0.76	0.76
BUBO Egg Detection Time - MSSAY	CV	1	0.76	-	0.76	0.76
BUBO Egg Detection Time - ASAY	CV	3	0.54	0.47	0	0.85
RALU Juv & Adult Numbers - MSSWY	CV	61	0.77	0.42	0	1.71
RALU Juv & Adult Numbers - MSSAY	CV	41	0.8	0.55	0	2.88
RALU Juv & Adult Numbers - ASAY	CV	129	0.68	0.48	0	2.88
AMTI Larvae Detection Time - MSSWY	CV	11	0.78	0.29	0.38	1.33
AMTI Larvae Detection Time - MSSAY	CV	6	0.78	0.32	0.38	1.18
AMTI Larvae Detection Time - ASAY	CV	7	0.86	0.36	0.38	1.35
PSMA Juv & Adult Numbers - MSSWY	CV	21	0.83	0.55	0	2.37
PSMA Juv & Adult Numbers - MSSAY	CV	12	0.99	0.74	0	2.56
PSMA Juv & Adult Numbers - ASAY	CV	13	1.03	0.71	0	2.56
AMMA Egg Detection Time - MSSWY	CV	1	0.94	-	0.94	0.94
AMMA Egg Detection Time - MSSAY	CV	1	0.94	-	0.94	0.94
AMMA Egg Detection Time - ASAY	CV	0	-	-	-	-
BUBO Juvs & Adults - MSSWY	CV	5	0.96	0.58	0	1.56
BUBO Juvs & Adults - MSSAY	CV	3	0.94	0.9	0	1.8
BUBO Juvs & Adults - ASAY	CV	35	0.87	0.62	0	1.95
RALU Egg Numbers - MSSWY	CV	0	-	-	-	-
RALU Egg Numbers - MSSAY	CV	0	-	-	-	-
RALU Egg Numbers - ASAY	CV	24	0.31	0.28	0	1.1
AMMA Egg Numbers - MSSWY	CV	0	-	-	-	-
AMMA Egg Numbers - MSSAY	CV	0	-	-	-	-
AMMA Egg Numbers - ASAY	CV	6	0.66	0.47	0.16	1.2
BUBO J & A Detection Time - MSSWY	CV	0	-	-	-	-
BUBO J & A Detection Time - MSSAY	CV	0	-	-	-	-
BUBO J & A Detection Time - ASAY	CV	1	1.25	-	1.25	1.25
AMMA Juv & Adult Numbers - MSSWY	CV	0	-	-	-	-
AMMA Juv & Adult Numbers - MSSAY	CV	0	-	-	-	-
AMMA Juv & Adult Numbers - ASAY	CV	0	-	-	-	-

Table B-4b Continued

Species Variable[1]	Method of Evaluation[2]	N[3]	X[4]	SD[4]	Min[4]	Max[4]
AMMA J&A Detection Time - MSSWY	CV	0	-	-	-	-
AMMA J&A Detection Time - MSSAY	CV	0	-	-	-	-
AMMA J&A Detection Time - ASAY	CV	0	-	-	-	-
AMTI Egg Numbers - MSSWY	CV	0	-	-	-	-
AMTI Egg Numbers - MSSAY	CV	0	-	-	-	-
AMTI Egg Numbers - ASAY	CV	0	-	-	-	-
AMTI Egg Detection Time - MSSWY	CV	0	-	-	-	-
AMTI Egg Detection Time - MSSAY	CV	0	-	-	-	-
AMTI Egg Detection Time - ASAY	CV	0	-	-	-	-
PSMA Egg Numbers - MSSWY	CV	0	-	-	-	-
PSMA Egg Numbers - MSSAY	CV	0	-	-	-	-
PSMA Egg Numbers - ASAY	CV	0	-	-	-	-
PSMA Egg Detection Time - MSSWY	CV	0	-	-	-	-
PSMA Egg Detection Time - MSSAY	CV	0	-	-	-	-
PSMA Egg Detection Time - ASAY	CV	0	-	-	-	-
RALU Egg Detection Time - MSSWY	CV	0	-	-	-	-
RALU Egg Detection Time - MSSAY	CV	0	-	-	-	-
RALU Egg Detection Time - ASAY	CV	0	-	-	-	-

[1] Variables are sorted in descending order from those with higher of levels of precision to those with lower levels of precision on the MSSWY method of calculation. MSSWY indicates values were calculated only from multiple surveys of a site conducted within a single year. MSSAY indicates values were calculated from all surveys at sites with multiple surveys conducted within at least one of the years of sampling (corresponds to shaded Site IDs in Table 1). ASAY indicates values were calculated from all surveys conducted across all years (i.e. sites were surveyed multiple times either within years, between years, or both corresponding to all Site IDs listed in Table 1). AMMA = Long-toed Salamander (*Ambystoma macrodactylum*), AMTI = Tiger Salamander (*Ambystoma tigrinum)*, BUBO = Western Toad (*Bufo boreas*), PSMA = Boreal Chorus Frog (*Pseudacris maculata*), RALU = Columbia Spotted Frog (*Rana luteiventris*), Terrestrial Gartersnake (*Thamnophis elegans*), THSI = Common Gartersnake (*Thamnophis sirtalis*).

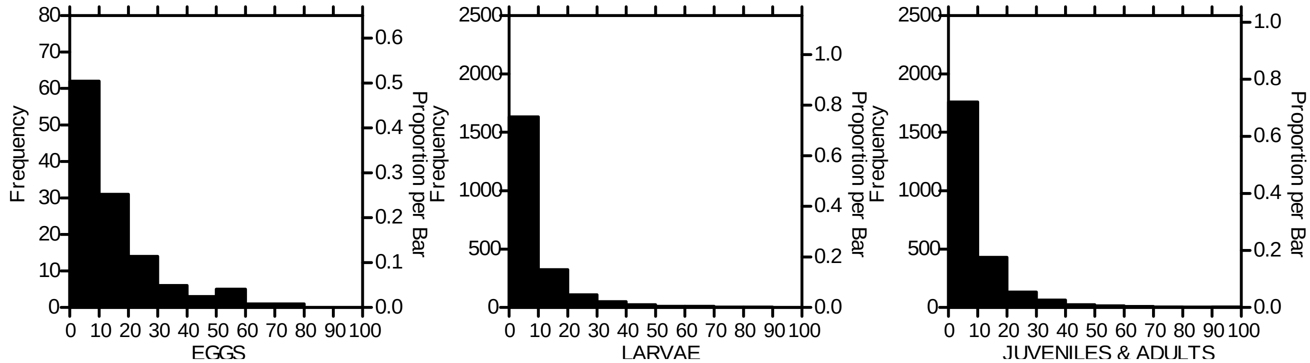
[2] CV values simply represent the standard deviation divided by the mean. Thus, a CV = 1 indicates the standard deviation of the responses was equivalent in magnitude to the mean value of the responses.

[3] N indicates numbers of sites for which CV could be calculated because of multiple surveys evaluating the variable.

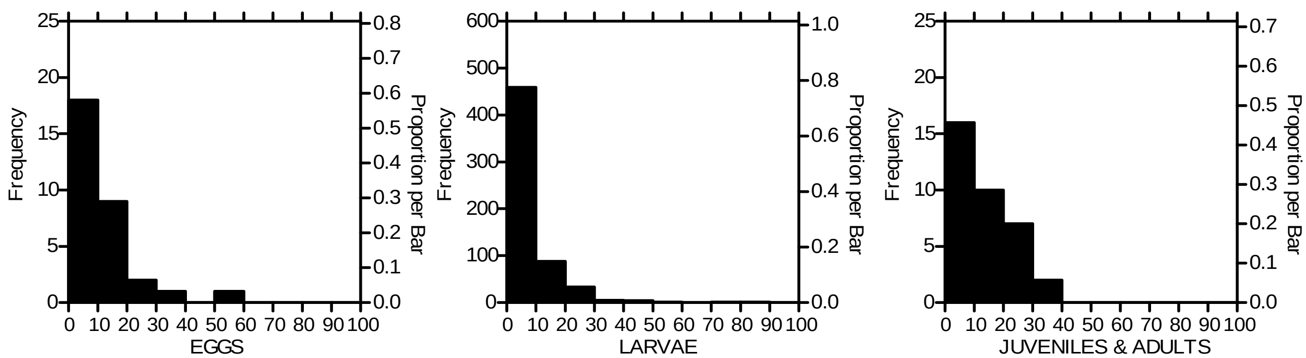
[4] X, SD, Min, and Max are the overall mean, standard deviation, minimum, and maximum values for CV values calculated for sites with multiple surveys where the variable was documented.

Figure B-1. Histograms of times to first detection for eggs, larvae, and juveniles or adults of all amphibian and reptile species together and for individual species.

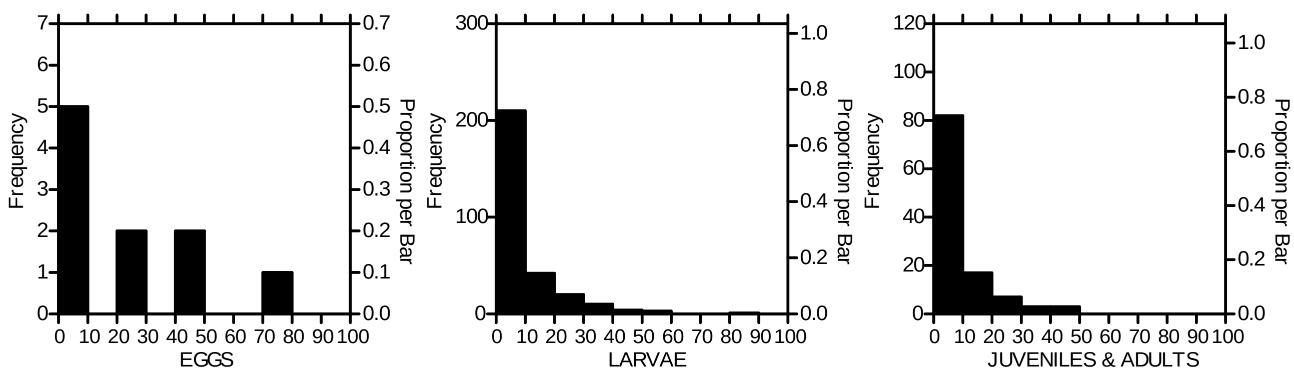
a – All Species



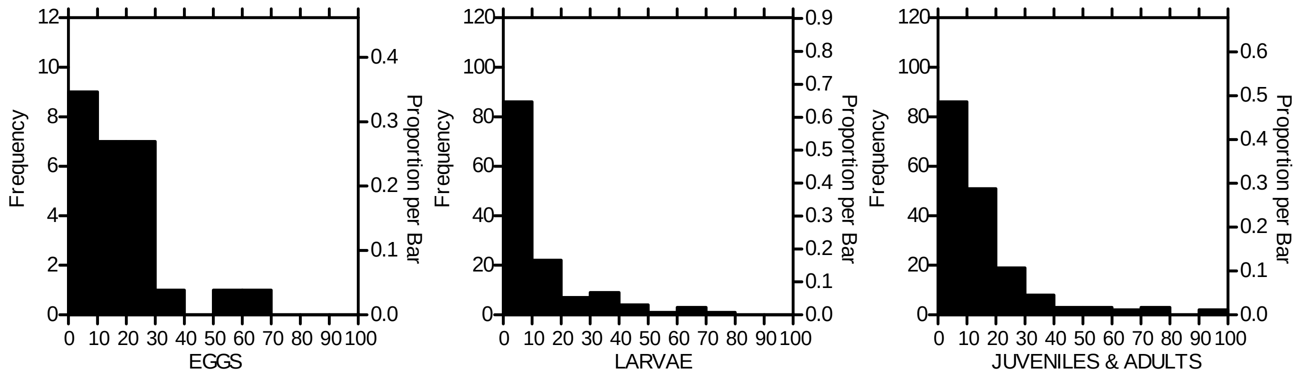
b – Long-toed Salamander (*Ambystoma macrodactylum*)



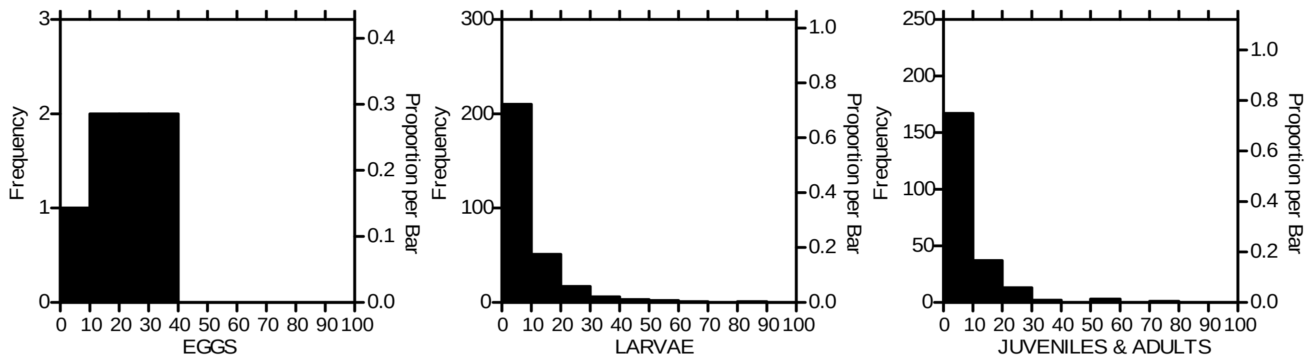
c – Tiger Salamander (*Ambystoma tigrinum*)



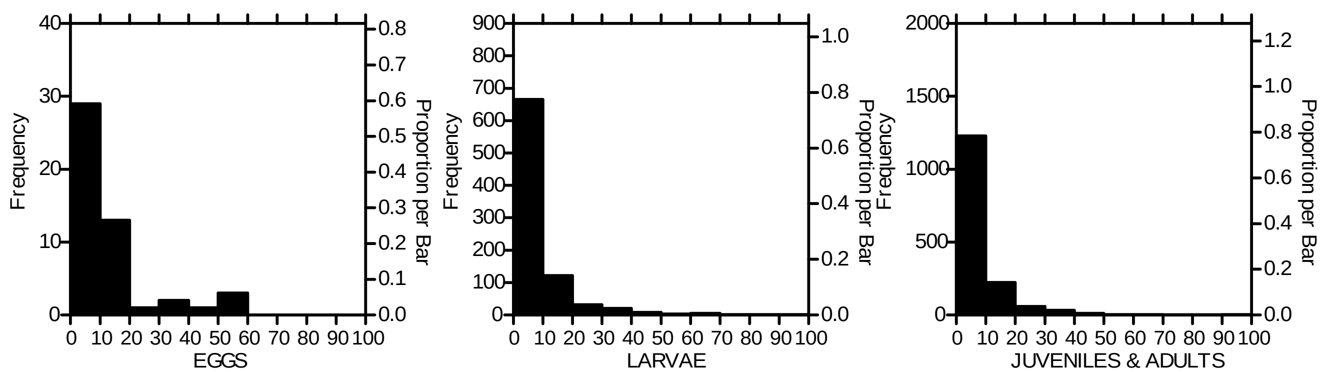
d – Western Toad (*Bufo boreas*)



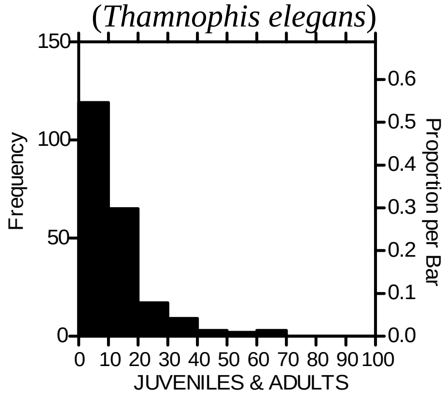
e – Boreal Chorus Frog (*Pseudacris maculata*)



f – Columbia Spotted Frog (*Rana luteiventris*)



g – Terrestrial Gartersnake (*Thamnophis elegans*)



h – Common Gartersnake (*Thamnophis sirtalis*)

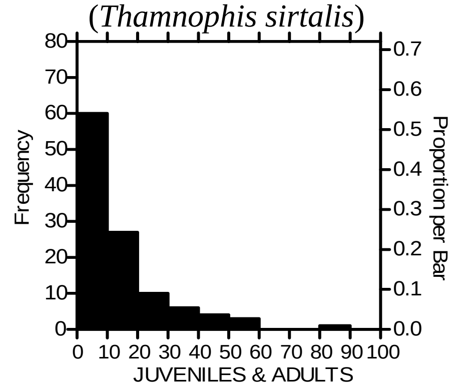
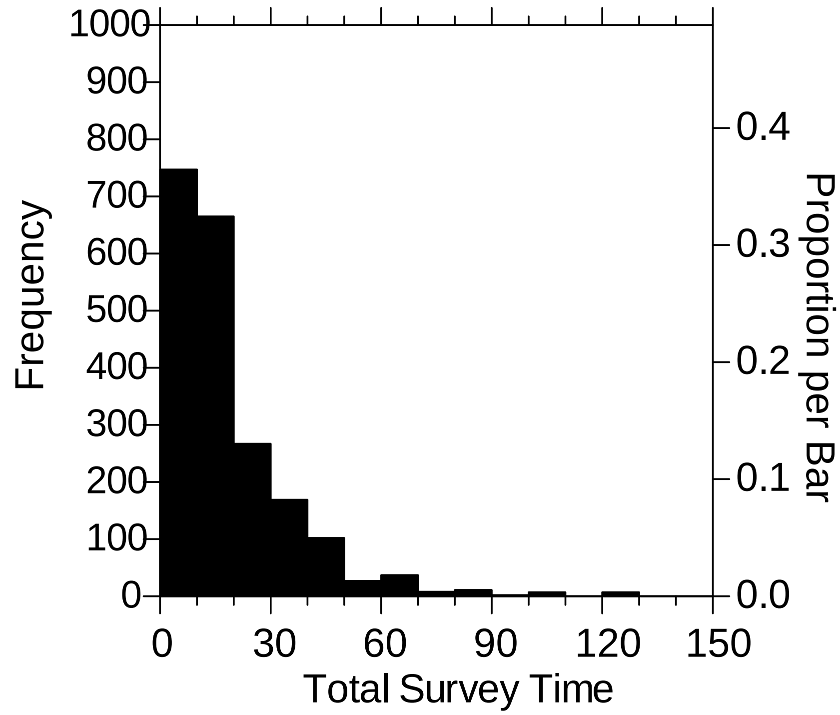
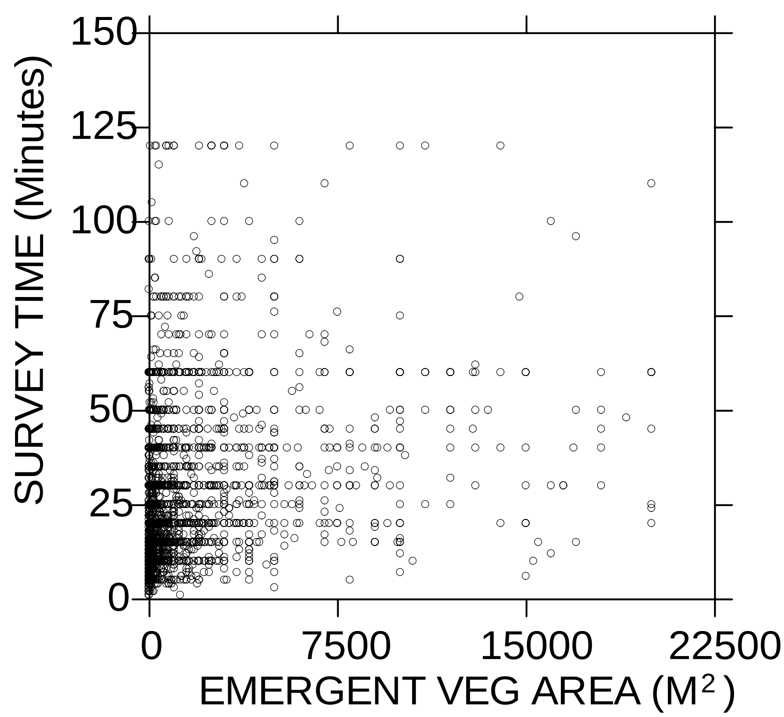


Figure B-2. Survey times at (a) sites without species detected and at (b) all sites surveyed which had at least 1 square meter and less than 25,000 square meters of emergent vegetation and which were surveyed for less than 2 hours.

a



b



APPENDIX C

Descriptions of Environmental Input Layers Used in Maximum Entropy Models

Input layers to models consist of 10 continuous (elevation, max temp, min temp, precip, ruggedness, slope, solar E, solar SS, solar WS, stream ED) and 5 categorical (aspect, geology, land cover, soil TM, soils) variables. These layers were dissolved or tiled together, converted to raster floating-point format, and resampled with a bilinear algorithm to a state-wide 90-meter grid cell coverage. Each source environmental layer is described below and, where appropriate, links to metadata are provided. Internet links were all accessed on 18 November 2008. The resulting state-wide grids all had 10,204 columns and 892 rows projected to North American Datum 1983 in Montana State Plane with an extent of top = 548006.61515, left = 111866.109204, right = 1030226.1092, bottom = 17726.6151503. In addition to a set of state-wide grids used to model all species at a state-wide scale, I created a set of environmental grids at the extent of each species' known range. All state-wide and range-wide raster grids were converted to ASCII grid format as required by Maxent.

Aspect (categorical)

Calculated using the Aspect tool in ArcMap 9.2 Spatial Analyst from the 10-meter National Elevation Dataset (NED) and resampled to 90-meter continuous floating-point grid cells using a bilinear algorithm. Cell values are compass directions of the maximum downslope rate of change in elevation. Continuous values were reclassified into 8 categorical values: 0 = Flat; 1 = North (337.5-22.5), 2 = Northeast (22.5-67.5), 3 = East (67.5-112.5), 4 = Southeast (112.5-157.5), 5 = South (157.5-202.5), 6 = Southwest (202.5-247.5), 7 = West (247.5-292.5), 8 = Northwest (292.5-337.5). For more information see the description of the elevation layer and the associated NED metadata link below.

Elevation (continuous)

The National Elevation Dataset (NED) is a 1/3 arc-second (10-meter) raster grid of decimal meter values assembled by the U.S. Geological Survey. The source layer was resampled to 90-meter continuous floating-point grid cells using a bilinear algorithm. Metadata on the 1/3 arc-second (10-meter) NED can be found at http://seamless.usgs.gov/products/3arc.php

Geology (categorical)

A polygonal coverage of 931 categories of surficial geology available in a mixture of 1:100,000 and 1:250,000 scales from the Montana State Geologic Mapping Program at the Montana Bureau of Mines and Geology (http://www.mbmg.mtech.edu/gmr/gmr-statemap.asp) and the Idaho Department of Water Resources (http://www.idwr.idaho.gov/gisdata/new%20data%20download/geology.htm). Individual source layers were appended and dissolved together using the Append and Dissolve tools in the ArcMap 9.2 Data Management Tools. The polygonal coverage was converted to raster and resampled to 90-meter continuous floating-point grid cells using a bilinear algorithm. A few grid cells in western Montana lacked values in areas were adjacent geology source data was not directly adjacent, but this did not present a problem for generating or interpreting predicted distribution models for species because these areas were small and only a handful of species observations overlapped these areas.

National Landcover Data (categorical)

The 1992 National Land Cover Data (NLCD) is based on 30-meter Landsat Thematic Mapper imagery. This was resampled to a 90-meter continuous floating-point grid cell coverage using a bilinear algorithm. A brief summary of raster cell value descriptions follows below. Data and metadata can be found at: http://nris.mt.gov/nsdi/nris/nlcdgrid.html

11	Open Water
12	Perennial Ice/Snow
21	Low Intensity Residential
22	High Intensity Residential

23	Commercial/Industrial/Transportation
31	Bare Rock/Sand/Clay
32	Quarries/Strip Mines/Gravel Pits
33	Transitional
41	Deciduous Forest
42	Evergreen Forest
43	Mixed Forest
51	Shrubland
61	Orchards/Vineyards/Other
71	Grasslands/Herbaceous
81	Pasture/Hay
82	Row Crops
83	Small Grains
84	Fallow
85	Urban/Recreational Grasses
91	Woody Wetlands
92	Emergent Herbaceous Wetlands

Max Temp (continuous)

A polygonal coverage of estimated average maximum daily temperatures for July in degrees Fahrenheit, for the climatological period 1971-2000. Estimates are based on Parameter-elevation Regressions on Independent Slopes Model (PRISM) derived raster data which uses known point temperature data and a digital elevation model (DEM) to generate gridded estimates of annual, monthly and event-based climatic parameters. General information on the underlying PRISM data and the source data itself can be downloaded from the Oregon Climate Service website at: http://www.ocs.orst.edu/prism/. The Montana data reprojected to Montana State Plane and resampled to a resolution of 600 meters representing 33 temperature ranges in degrees Fahrenheit is available at: http://nris.mt.gov/nsdi/nris/tmax71_00.html. Source polygons were converted to raster and resampled to a 90-meter continuous floating-point grid cell coverage using a bilinear algorithm.

Min Temp (continuous)

A polygonal coverage of estimated average minimum daily temperatures for January in degrees Fahrenheit, for the climatological period 1971-2000. Estimates are based on Parameter-elevation Regressions on Independent Slopes Model (PRISM) derived raster data which uses known point temperature data and a digital elevation model (DEM) to generate gridded estimates of annual, monthly and event-based climatic parameters. General information on the underlying PRISM data and the source data itself can be downloaded from the Oregon Climate Service website at: http://www.ocs.orst.edu/prism/. The Montana data reprojected to Montana State Plane and resampled to a resolution of 600 meters representing 29 temperature ranges in degrees Fahrenheit is available at: http://nris.mt.gov/nsdi/nris/tmin71_00.html. Source polygons were converted to raster and resampled to a 90-meter continuous floating-point grid cell coverage using a bilinear algorithm.

Precip (continuous)

Relative Effective Annual Precipitation (REAP) data indicates the average estimated amount of soil moisture available in 1 cm intervals for the period 1971-2000. Estimates are based on 1 km DAYMET http://www.daymet.org/ precipitation estimates for 1980-1997 from the University of Montana's Numerical Terradynamics Simulation Group (NTSG) adjusted to the period 1971-2000 using Temperature and Precipitation Summary Tables (TAPS) developed by the National Water and Climate Center and then sensitized to the landscape according to slope, aspect, and soil properties. REAP estimates were developed for each Montana county and edge-matched into a state-wide 30-meter raster coverage by the Montana State Office of the Natural Resources Conservation Service. The 30-meter source grid was resampled to a 90-meter continuous floating-point grid using a bilinear algorithm. Data and metadata are available at: http://nris.mt.gov/gis/gisdatalib/gisDataList.aspx?datagroup=statewide-regional&searchTerms=REAP.

Ruggedness (continuous)

Calculated from the 30-meter National Elevation Dataset (NED) using the vector ruggedness measure of Sappington et al. (2005). This is a 3-dimensional measure of the dispersion of vectors within a moving analysis window where vectors represent a trigonometric combination of slope and aspect in individual grid cells. The greater the dispersion of the resulting vectors, the more rugged the landscape. The 30-meter NED is a 1 arc-second raster grid of decimal meter values assembled by the U.S. Geological Survey. Metadata on the 30-meter NED can be found at: http://seamless.usgs.gov/products/1arc.php. The 30-meter coverage of vector ruggedness values was resampled to a 90-meter continuous floating-point grid cell coverage using a bilinear algorithm.

Slope (continuous)

The inclination of the slope in degrees calculated using the Slope function in ArcMap 9.2 Spatial Analyst from the 10-meter National Elevation Dataset (NED) and resampled to a 90-meter continuous floating-point grid cell coverage using a bilinear algorithm. See description of the elevation layer and the associated NED metadata link above.

Soils (categorical)

State Soil Geographic data (STATSGO) is a polygonal coverage of general soil associations developed by the National Cooperative Soil Survey. The soil maps for STATSGO are compiled by generalizing more detailed soil survey maps. Map unit composition for a STATSGO map is determined by transecting or sampling areas and expanding the data statistically to characterize the whole map unit. Therefore, soil map units depict the dominant soils making up the landscape and often contain dissimilar soil types. The approximate minimum area delineated is 625 hectares (1,544 acres). The polygonal coverage was converted to raster and resampled to 90-meter continuous floating-point grid cells using a bilinear algorithm. Background information and metadata is available at: http://dbwww.essc.psu.edu/doc/statsgo/statsgo_info.html and http://nris.mt.gov/nsdi/statsgo.pdf. Definitions for the 694 map units used in the input can be downloaded as .dbf files along with a STATSGO shapefile for Montana at:

http://nris.mt.gov/gis/gisdatalib/gisDataList.aspx?datagroup=statewide-regional&searchTerms=statsgo

Soil TM (categorical)

This is a generalized polygonal coverage of soil temperature and moisture regimes provided by the Natural Resources Conservation Service (USDA 1994). The polygonal coverage was converted to raster and resampled to 90-meter continuous floating-point grid cells using a bilinear algorithm. A brief summary of raster cell value descriptions for temperature / moisture follows below. A glossary of relevant soil terminology can be found at: https://www.soils.org/sssagloss/. Metadata on this layer can be found at: http://soils.usda.gov/use/worldsoils/mapindex/str.html

1	Cryic/Udic
2	Frigid/Udic
3	Frigid/Typic Ustic
4	Cryic/Typic Ustic
5	Frigid/Aridic Ustic
6	Frigid Aquic
7	Frigid/Typic Xeric
8	Water
9	Cryic/Typic Xeric
10	Cryic/Aridic Ustic
11	Mesic/Ustic Aridic
12	Cryic/Udic Ustic

Solar E, Solar SS, Solar WS (all continuous)

This is an index proportional to the amount of extraterrestrial solar radiation striking an arbitrarily oriented surface during solar noon at the equinox (Solar E), summer solstice (Solar SS) and winter solstice (Solar WS). The index was calculated at each tenth degree of latitude from the 30-meter National Elevation Dataset (NED) with ArcMap 9.2 Spatial Analyst using the formula presented in Keating et al. (2006). Variables in this formula include solar hour angle (held constant at solar noon), distance of the earth from the sun, declination of the earth, latitude, slope, and aspect. The 30-meter NED is a 1 arc-second raster grid of decimal meter values assembled by the U.S. Geological Survey. Metadata on the 30-meter NED can be found at: http://seamless.usgs.gov/products/1arc.php. Each 30-meter coverage of solar radiation index values was resampled to a 90-meter continuous floating-point grid cell coverage using a bilinear algorithm.

Stream ED (continuous)

Euclidian distances from major streams in meters were calculated using the Euclidean Distance tool in ArcMap 9.2 Spatial Analyst for 90-meter grid cells from the 1:100,000 scale U.S. Census Bureau TIGER files. Only streams that approximately matched the data shown in USGS 1:2,000,000 scale digital line graphics files were included. Data and metadata for the major stream source layer can be found at: http://nris.mt.gov/nsdi/nris/HD43.html.

APPENDIX D

Maximum Entropy Model Output for Montana's Herpetofauna

For each of Montana's 14 amphibian and 17 reptile species (Table 3.1) we: (1) present state-wide continuous predicted habitat suitability maps showing areas of higher predicted suitability in warmer colors (reds) and lower predicted suitability in cooler colors (blues); (2) present binary maps of predicted suitable and unsuitable habitat using the low threshold cutoff on continuous models within the species' range (Table 3.4) along with observations used to train (red squares) and, where possible, test (red stars) the models; (3) present maps of predicted low (yellow), moderate (orange) and high (red) habitat suitability class thresholds for species with test data available to make these determinations (Tables 3.3-3.4, Figure 3.1); (4) discuss the implications of state-wide continuous models for potential extensions to known range and areas within the range that currently lack records; (5) summarize characteristics of response curves for environmental variables ranked as important in predicting the species' distribution (Table 3.5); (6) summarize visual evaluations of the spatial arrangement and magnitude of test occurrence deviances in the context of how well we feel the low cutoff threshold represents overall distribution and habitat suitability classes represent our knowledge of marginal, moderate, and optimal habitat suitability; (7) compare output from Maxent models to output from the Montana Gap Analysis Project (Hart et al. 1998) in terms of portrayal of distribution and differences in the area of predicted suitable habitat; (8) discuss evaluations of continuous model output within each species' range using the area under the curve (AUC) of receiver operating characteristic (ROC) plots (Table 3.6); (9) discuss evaluations of binary predictions of suitable versus unsuitable habitat for GAP using all data and low cutoff range models of species for which test data was available using the absolute validation index (AVI) (Tables 3.6-3.7); (10) discuss evaluations of the low cutoff range models and GAP models using lentic site survey data from the Montana Amphibian Inventory and Monitoring Project (Chapter 2) for commission error rates, omission error rates, overall map accuracy, and the Kappa Index; and (11) provide suggestions on limitations and appropriate uses of the models.

Long-toed Salamander (*Amybstoma macrodactylum*)

The state-wide continuous habitat suitability model (Figure D-1a) indicates that the species' known range may be extended on: (1) state and Helena National Forest lands near Rogers Pass and Flesher Pass; (2) the eastern edge of the Elkhorn Mountains; (3) the Beaverhead-Deerlodge National Forest in the area around Whitetail Reservoir and on Bull Mountain between the Boulder River and Whitehall Creek in Jefferson County; and (4) BLM and Beaverhead-Deerlodge National Forest lands south of Butte in the vicinity of the Humbug Spires. Characteristics of environmental variables of importance (Table 3.5) include annual precipitation greater than 50 cm, cool and moist soil types, and slopes of less than 25 degrees. Solar radiation indices were the only variables that did not help improve models and they actually made them worse in the jackknife plots of test gain. Continuous range model AUC (0.909) and low cutoff threshold AVI (0.96) values for test data indicated high accuracy and AVI values were drastically better than GAP (0.20) (Table 3.6). The low cutoff threshold of the continuous range model resulted in a 301 percent (31,865 km^2) increase in area of predicted habitat relative to GAP models because GAP had modeled them simply as a 500-m buffer around streams (Table 3.7, Figure D-1b, Hart et al. 1998). Evaluation with the lentic site survey data yielded lower omission errors for the low cutoff threshold than GAP, but higher commission errors and lower overall map accuracy (Table 3.8). Kappa was low for GAP and the low cutoff threshold. High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and, in general, the magnitude of deviances is inversely correlated with predicted habitat suitability, indicating that the habitat suitability classifications performed well at the landscape scale. However, habitat in some valley bottoms may be under predicted as a result of a lack of surveys on private lands (Figure D-1c). As a result of the probable under predictions on private lands in areas with few records it may be best to use the continuous output in tandem with a deductive model when considering this species in management plans for these areas.

Figure D-1a. Continuous state-wide habitat suitability index for *A. macrodactylum*.

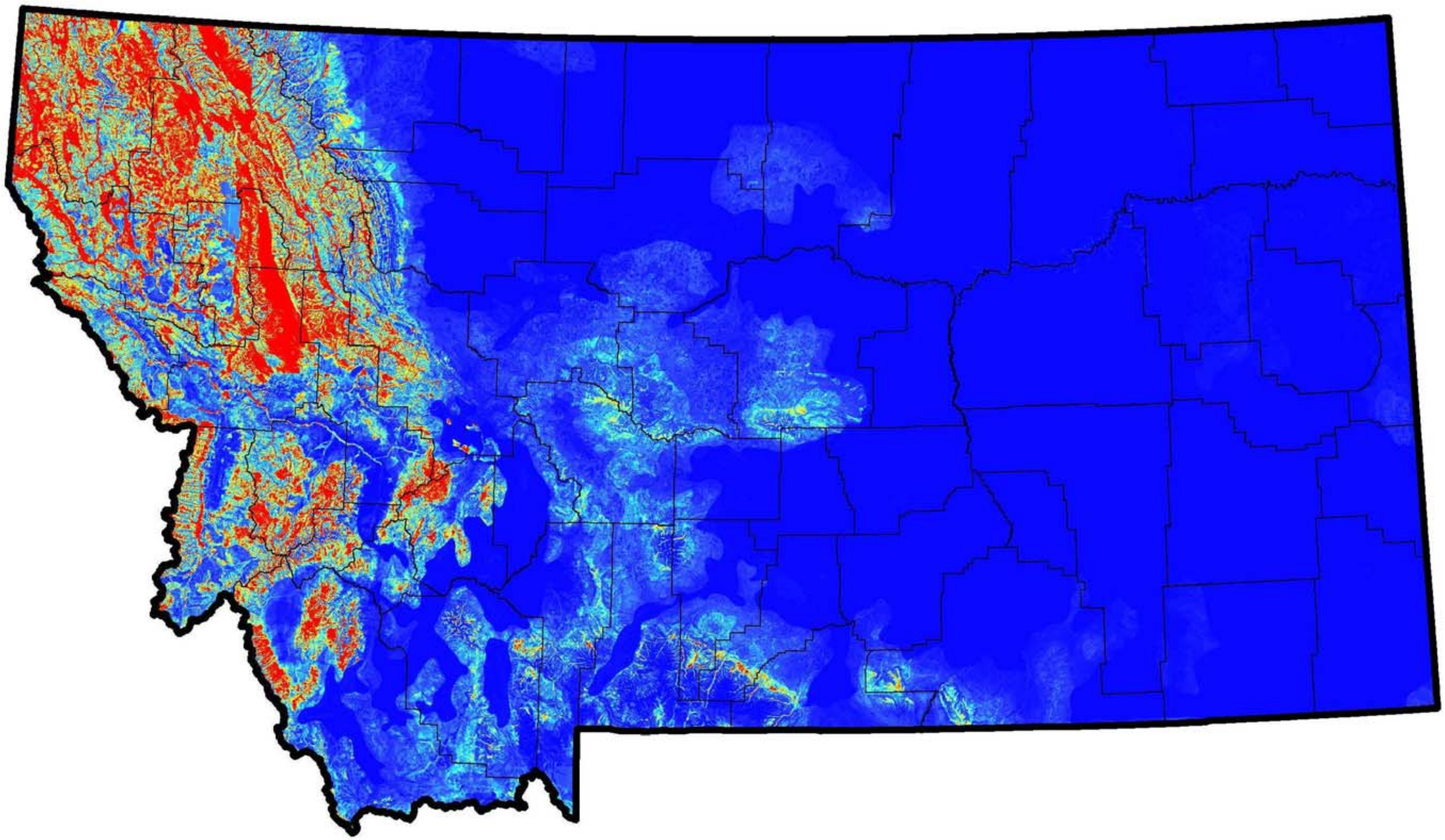


Figure D-1b. Predicted suitable habitat and training (squares) and test (stars) observations for *A. macrodactylum*.

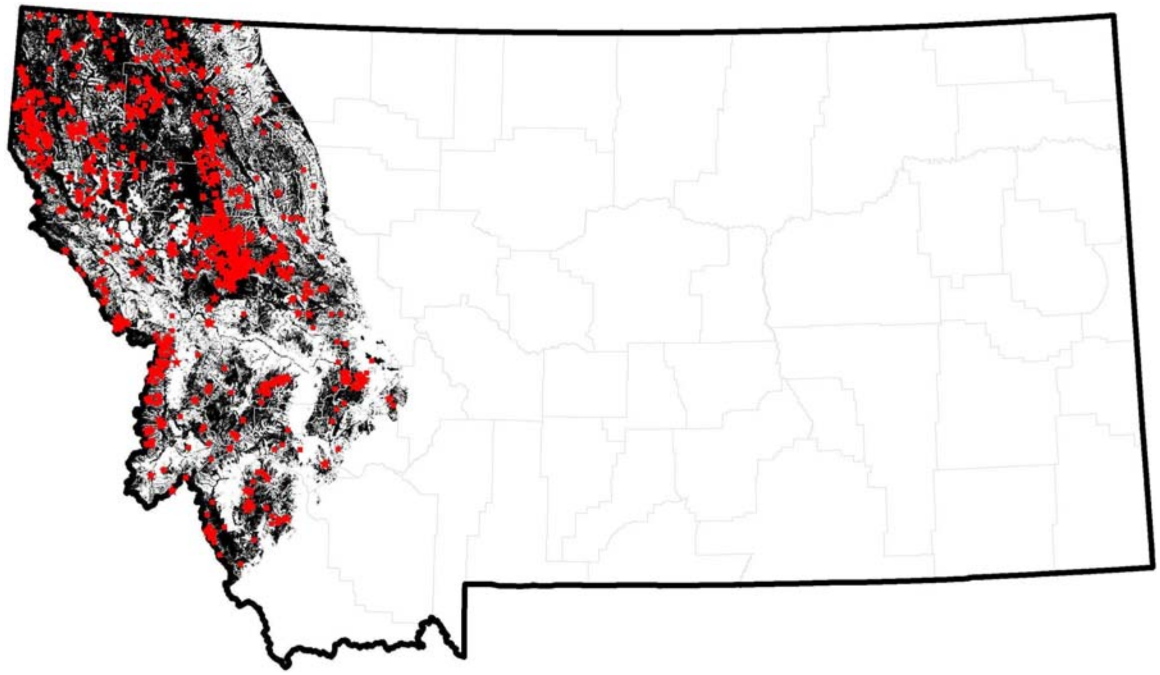
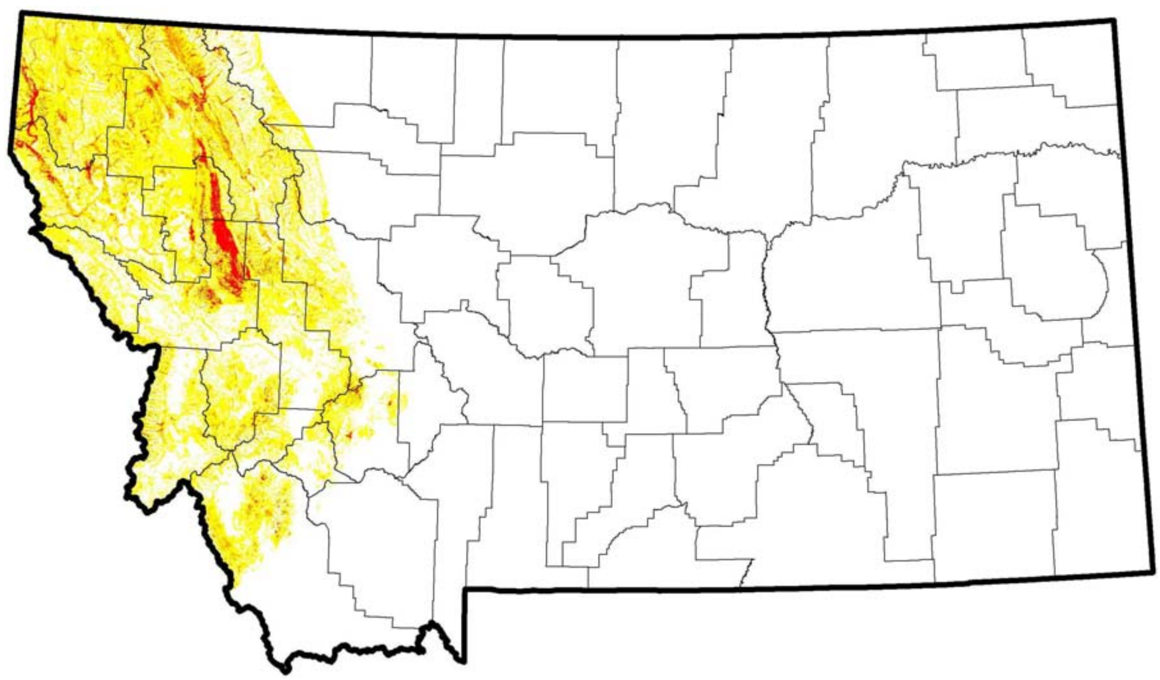


Figure D-1c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *A. macrodactylum*.



Tiger Salamander (*Amybstoma tigrinum*)

The state-wide continuous habitat suitability model (Figure D-2a) indicates that the species' known range may be extended westward on state, BLM and private lands in southwestern Montana east of the Red Rock and Jefferson Rivers (e.g., Blacktail Mountains, Sweetwater Basin, Three Forks to Townsend area, and Wilsall to White Sulfur Springs area). Characteristics of environmental variables of importance include land cover features associated with wetlands, and slopes less than 20 degrees. Solar radiation indices were the only variables that did not help improve models and they actually made them worse in the jackknife plots of test gain. Continuous range model AUC (0.878) was good, but low relative to other species (Table 3.6). AVI for the low cutoff was poorer than GAP (0.93 and 0.98, respectively) and the low cutoff threshold resulted in a 49 percent (139,456 km^2) decrease in area of predicted habitat relative to GAP models (Table 3.7). Omission and commission error rates, map accuracy, and Kappa were similar for the low cutoff and GAP models (Table 3.8). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance. However, the low cutoff threshold for the range model appears to exclude some areas on private lands that lack surveys and are likely to support populations and the low, moderate, and optimal habitat suitability class thresholds appear to be an artifact of survey effort in some poorly surveyed areas dominated by private lands (Figures D-2b-c). Thus, although the Maxent models appear to be an improvement over GAP in areas that have some baseline surveys (e.g., the area between the Tongue and Powder Rivers), they likely under represent the species' distribution in areas that lack some baseline survey effort (e.g., large areas between the Missouri and Yellowstone Rivers). Thus, until more observation data is available in regions that currently have few observations, it would probably be best to use a combination of a deductively based model and the continuous model outputs when considering this species in management plans. Another potential approach is to increase the regularization on the Maxent model to extend the predicted area while maintaining representation for areas of higher suitability.

Figure D-2a. Continuous state-wide habitat suitability index for *A. tigrinum*.

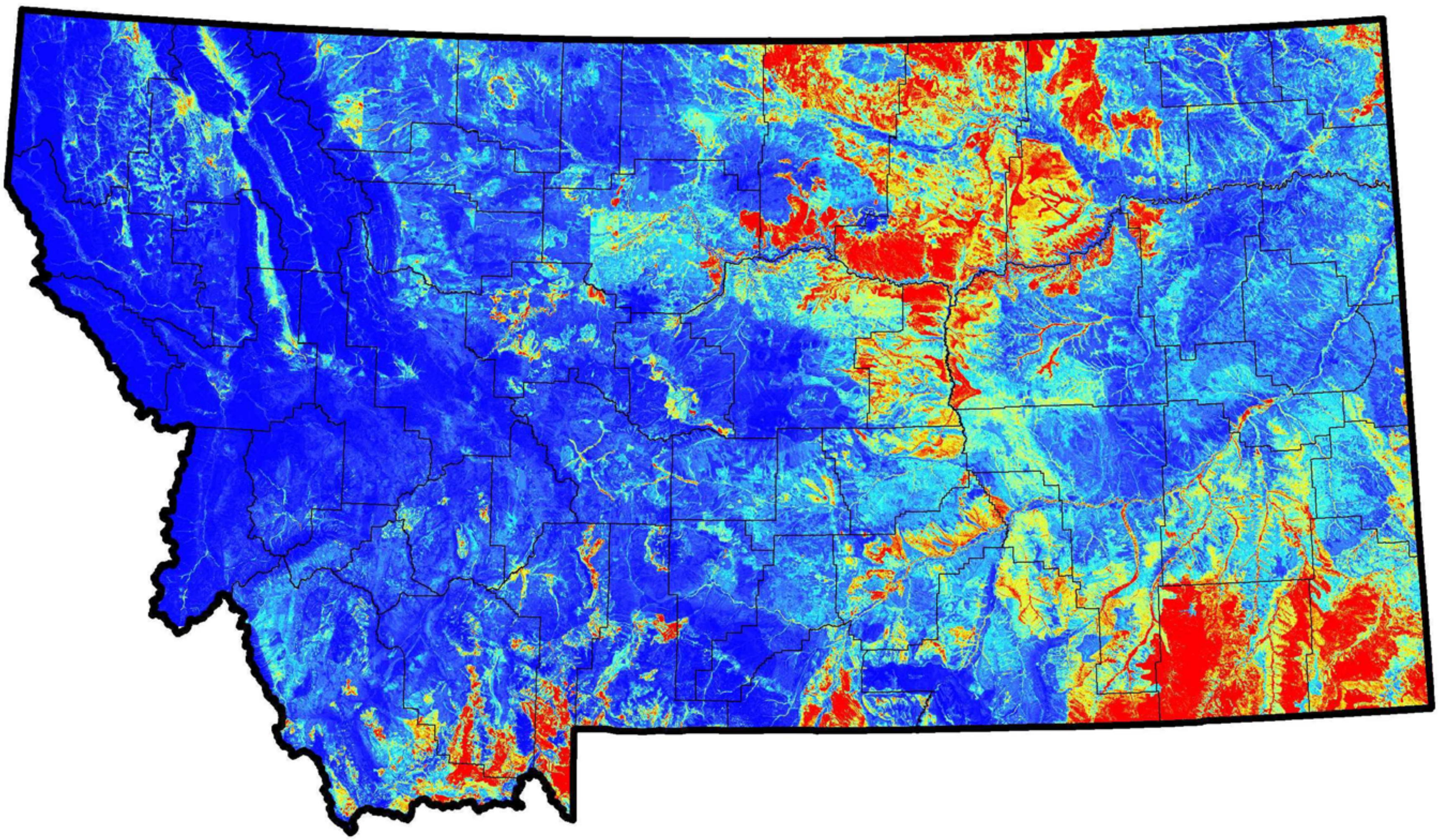


Figure D-2b. Predicted suitable habitat and training (squares) and test (stars) observations for *A. tigrinum*.

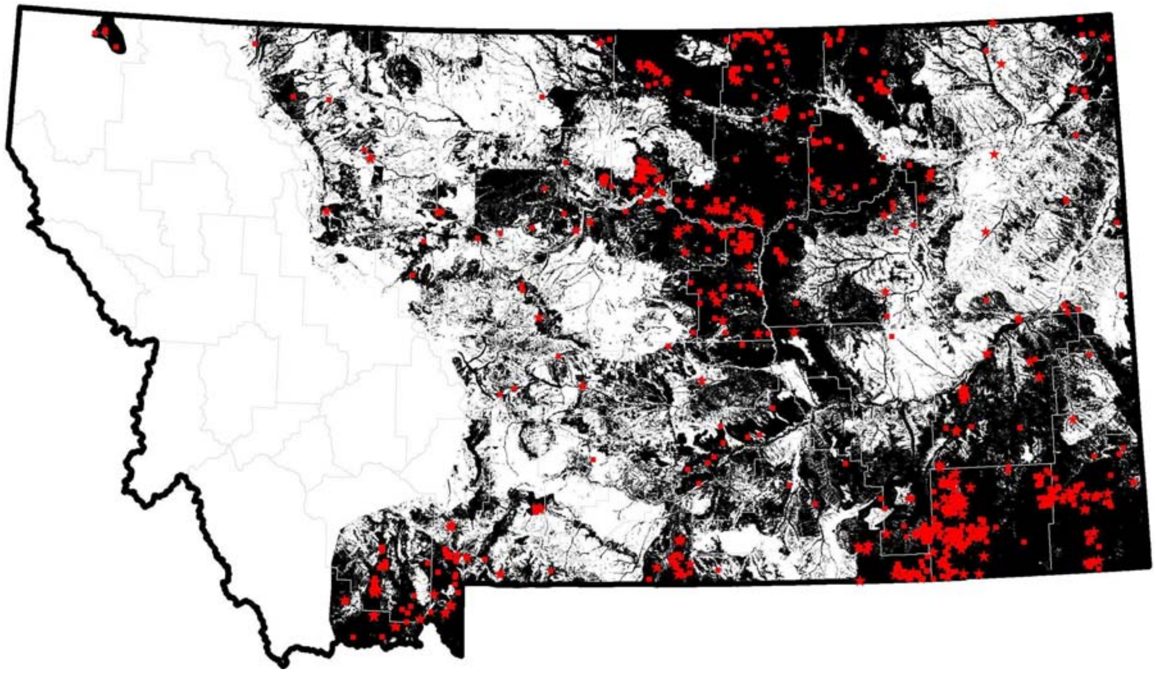
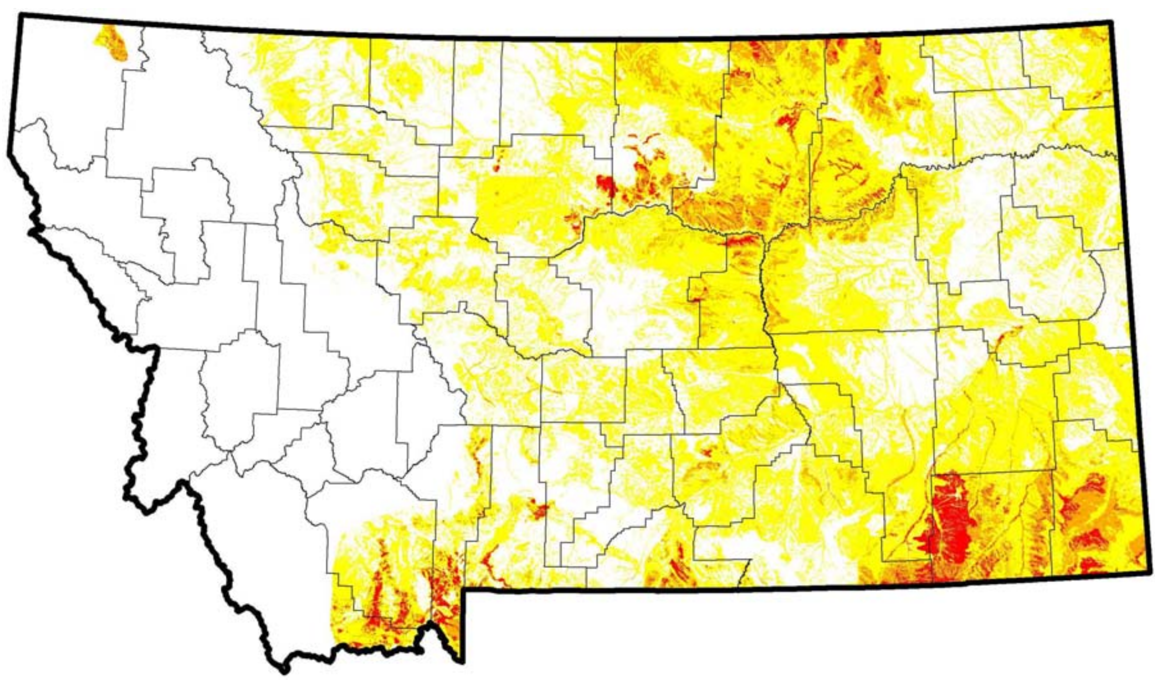


Figure D-2c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *A. tigrinum*.



Idaho Giant Salamander (*Dicamptodon atterimus*)

The state-wide continuous model indicates the species' range may be extended along the Idaho border between Lookout and Lolo passes, a region that is parallel to the species' known range in Idaho (Figures D-3a-b, Jones et al. 2005). However, recent electrofishing surveys throughout this area failed to detect them outside of the area south of Saltese, Haugan, and De Borgia (pers. obs.). The species' known range in Montana is very limited and some environmental variables with high importance ranks such as terrain ruggedness may simply be an artifact of the few areas where they have been documented. However, precipitation of greater than 110 cm and evergreen forest cover are clearly important to the species' distribution and north, northeast, and easterly aspects are probably restrictive in terms of maintaining stream flows and high soil moisture. The range-wide low cutoff threshold distribution appears to be a reasonable representation of potential distribution and suitable habitat in the region they have been documented. GAP AVI was good (0.92), but the Maxent models probably more appropriately represent the distribution of the species than the more general 1 km hydrography buffer within suitable habitat cover types applied by GAP (Hart et al. 1998). The limited distribution of the species in western Montana highlights the importance of implementing conservation measures for documented populations and surveying for additional populations in areas likely to be impacted by management actions such as timber harvest and road development within their predicted distribution.

Figure D-3a. Continuous state-wide habitat suitability index for *D. atterimus*.

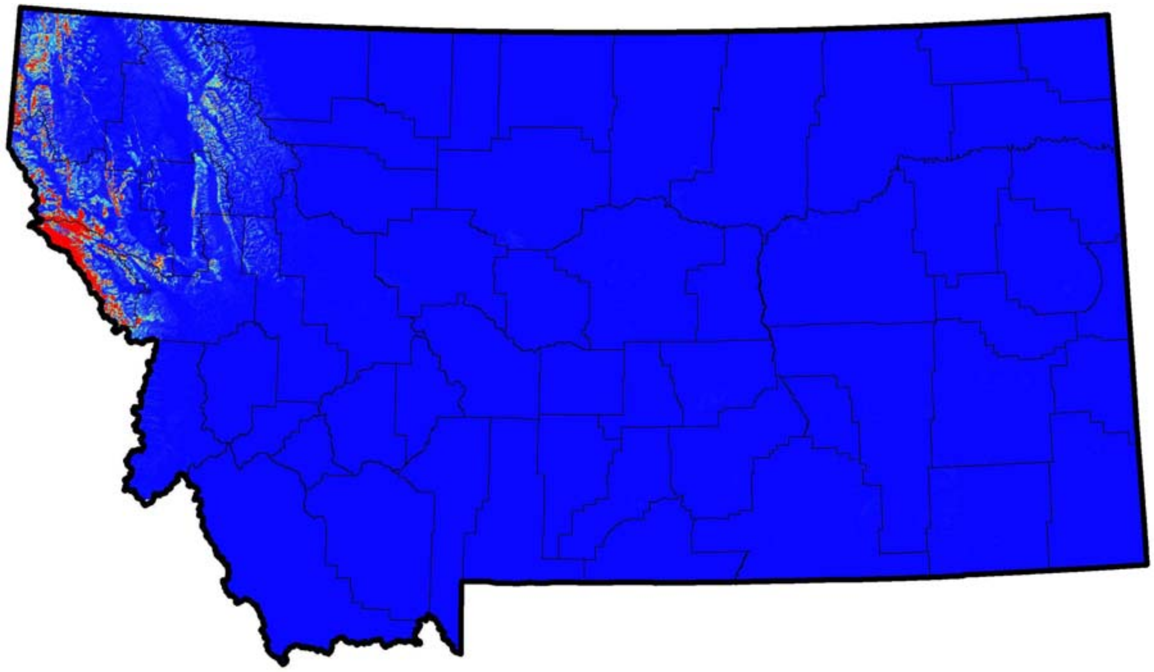
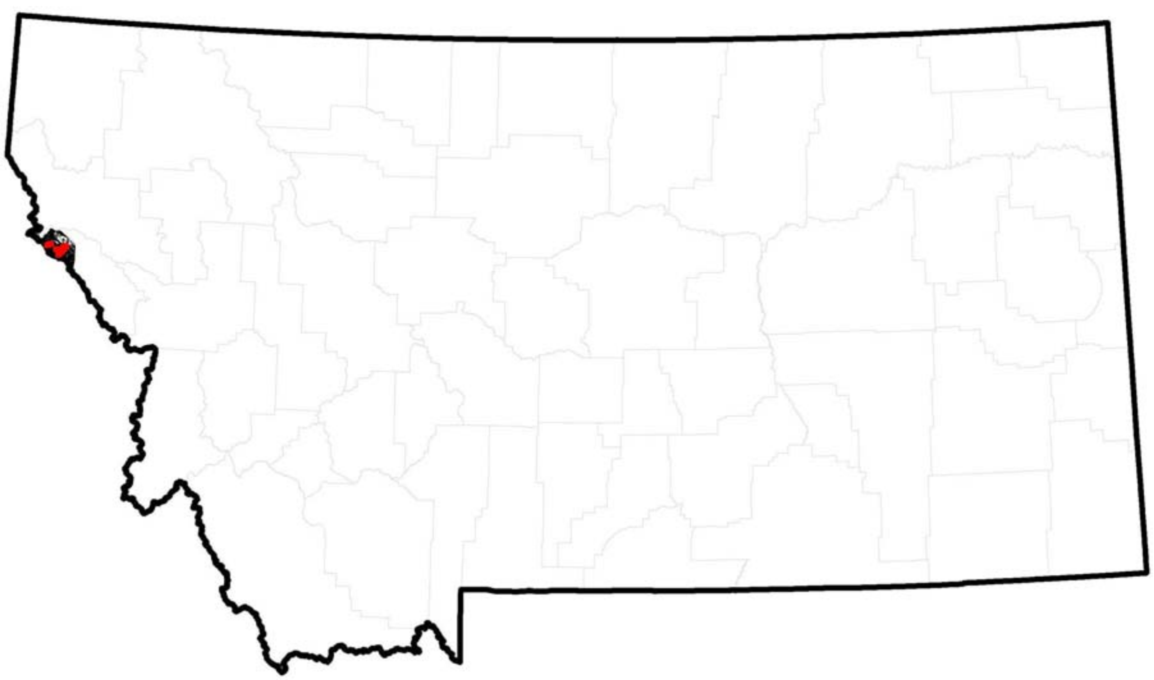


Figure D-3b. Predicted suitable habitat and training observations for *D. atterimus*.



Coeur d'Alene Salamander (*Plethodon idahoensis*)

The state-wide continuous model indicates that the species' known range may be extended a significant distance to the east (Figure D-4a). Similarly, the models indicate the species is likely to be documented in a number of additional watersheds across their known range in Montana. This seems likely given that only limited baseline surveys have been performed to-date and most of these date back to the late 1980s (Wilson 1993). Soil temperature and moisture regime (cool and moist soils) and average minimum January temperature (greater than -11 °C) were ranked as important variables in state-wide models, but were ranked of low importance at the scale of the species' range. Close proximity to streams, slopes greater than 20 degrees, and elevations below 1,000 m were all associated with highest habitat suitability. Similarly terrain ruggedness was ranked as an important variable by both state-wide and range-wide models, apparently as a result of being indicative of the rock fractures the species is dependent on (Wilson et al. 1997, Wilson et al. 1998). GAP AVI was very poor (0.30), but in many cases areas predicted by GAP were only a few pixels away from observations. The low cutoff threshold of the continuous range model resulted in a 572 percent (5,015 km^2) increase in area of predicted suitable habitat relative to GAP models (Table 3.7). Although these more expansive continuous and low cutoff threshold models could not be evaluated with test data visual inspection indicates a good fit with our field experience at both the regional and landscape scale and they seem to more appropriately represent the probable distribution of the species than the simple GAP model of streams within suitable habitat cover types (Figure D-4b, Hart et al. 1998).

Figure D-4a. Continuous state-wide habitat suitability index for *P. idahoensis*.

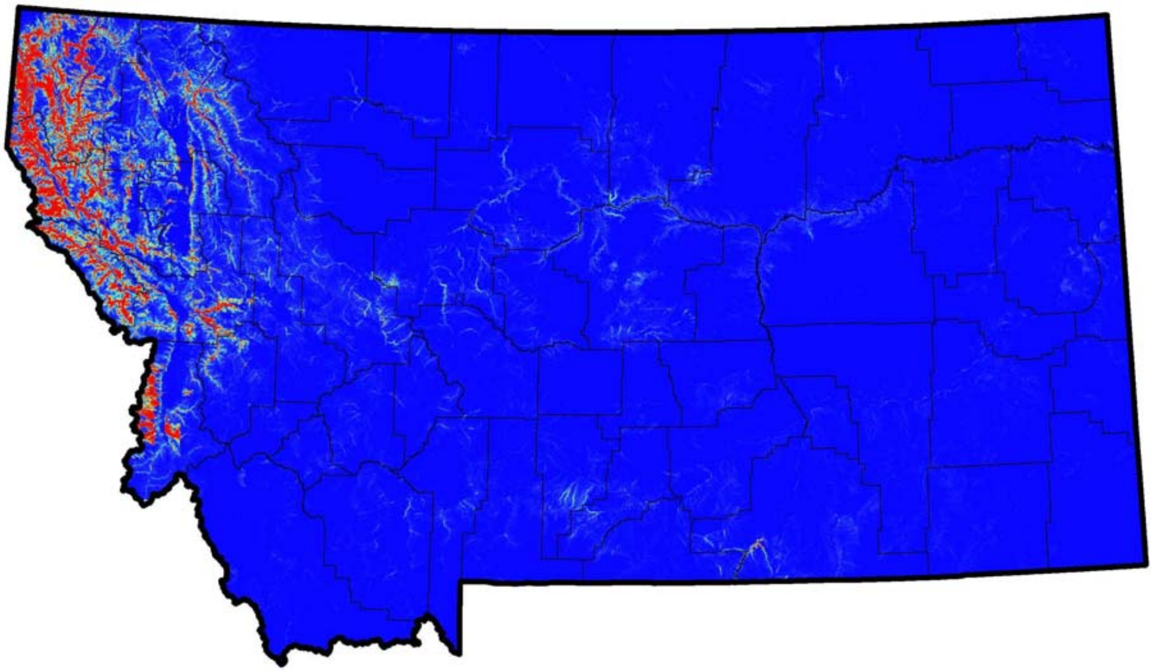
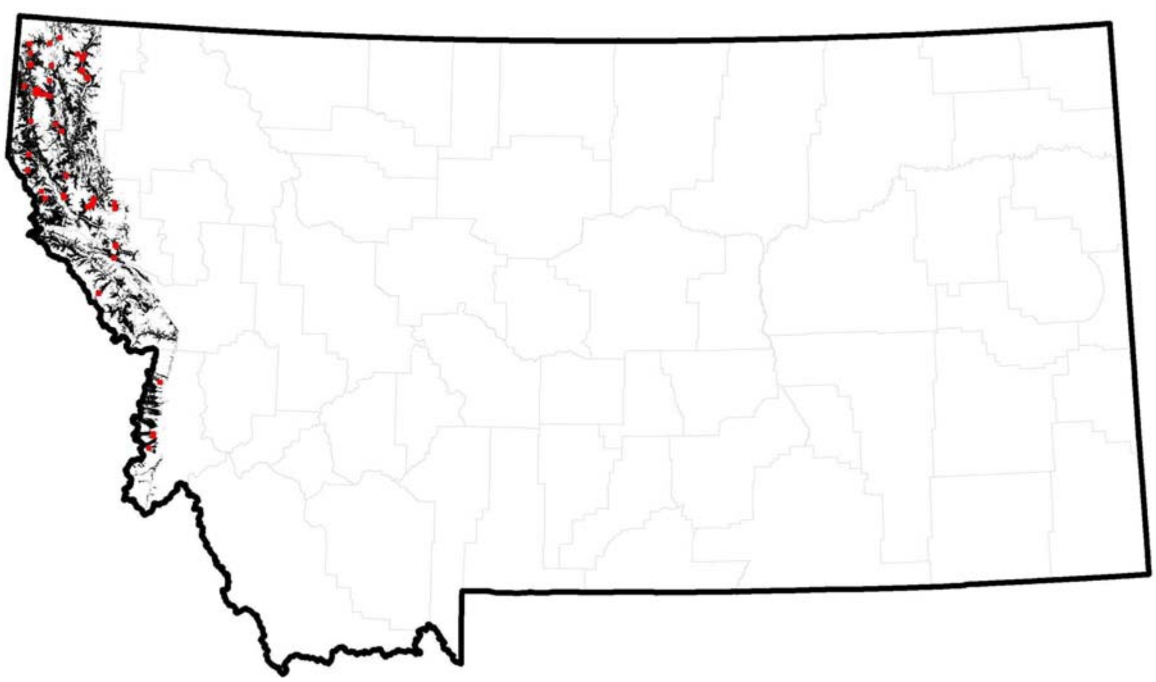


Figure D-4b. Predicted suitable habitat and training observations for *P. idahoensis*.



Rocky Mountain Tailed Frog (*Ascaphus montanus*)

The state-wide continuous model (Figure D-5a) indicates that the species' known range may be extended to: (1) the east along the Beaverhead Mountains to Bloody Dick Creek; (2) the eastern edge of the East Pioneer Mountains; (3) the Flint Creek Range; (4) portions of the Front Range near the Sun River Wildlife Management Area; and (5) the area around Rogers Pass, Flesher Pass, and Stemple Pass. Precipitation was ranked as more important in state-wide than range-wide models with suitability of habitat increasing between 50 and 200 cm per year (Table 3.5). Wetland and evergreen forest cover types were associated with higher suitability and slope and terrain ruggedness were ranked as relatively important, probably as indicators of areas likely to support stream flow or provide shaded habitats. The AUC and AVI for test data evaluating the continuous range-wide and low cutoff threshold models (0.889 and 0.92, respectively) indicated good model fit and much better fit than the AVI for GAP (0.32) (Figure D-5b, Table 3.6), but in many cases areas predicted by GAP were only a few pixels away from observations. Thus, the GAP models do not appear to be as poor as indicated by the test data, but are clearly too restrictive as simply streams within suitable habitat cover types (Hart et al. 1998). The low cutoff threshold of the continuous range model resulted in a 588 percent (28,927 km^2) increase in area of predicted suitable habitat relative to GAP models (Table 3.7). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and the magnitude of deviances is inversely correlated with predicted habitat suitability, indicating that the habitat suitability classifications performed well at the landscape scale. The low, moderate, and optimal cutoff thresholds match with our field observations and appear to be good approximation of relative habitat suitability for the species (Figure D-5c).

Figure D-5a. Continuous state-wide habitat suitability index for *A. montanus*.

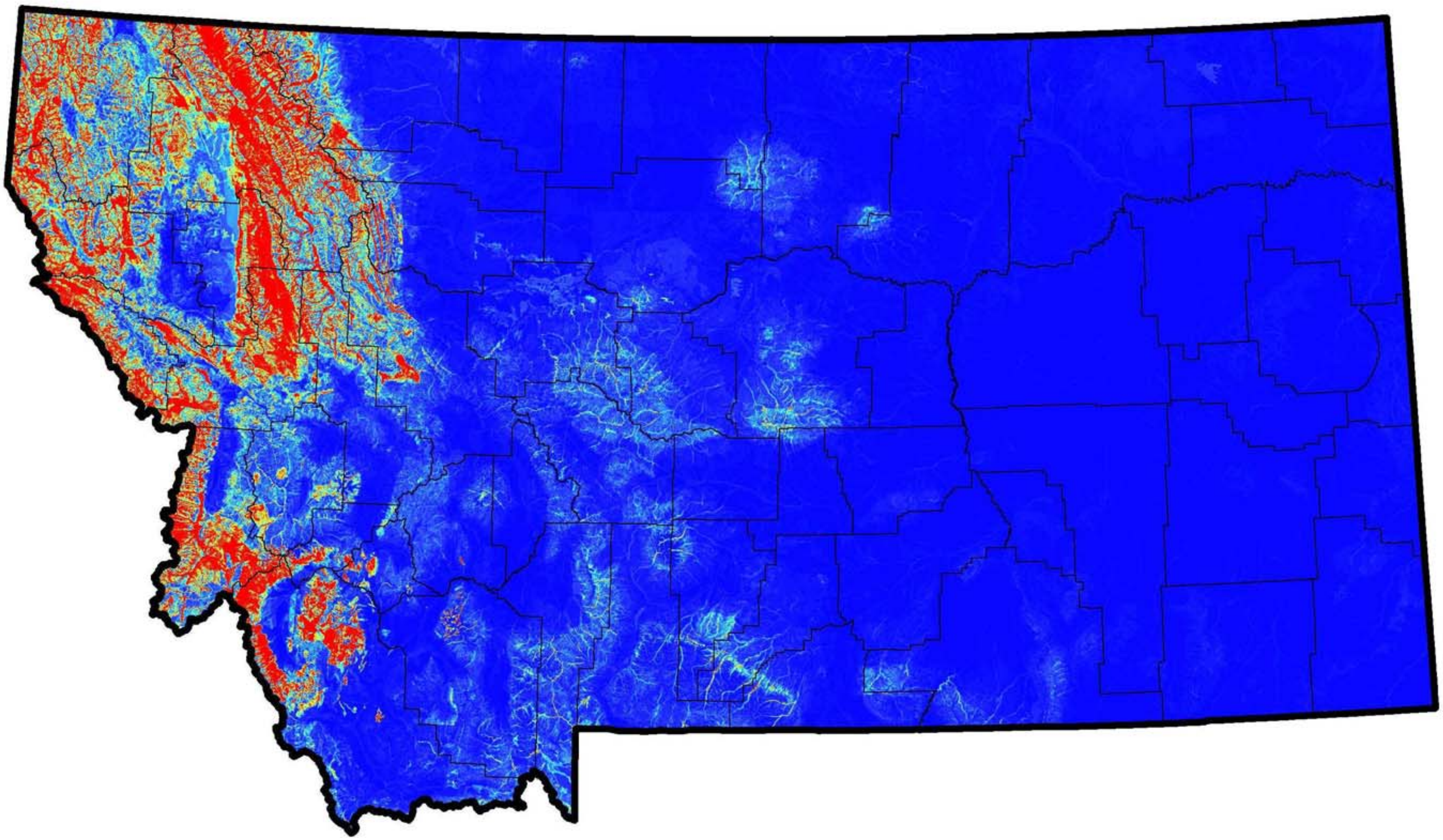


Figure D-5b. Predicted suitable habitat and training (squares) and test (stars) observations for *A. montanus*.

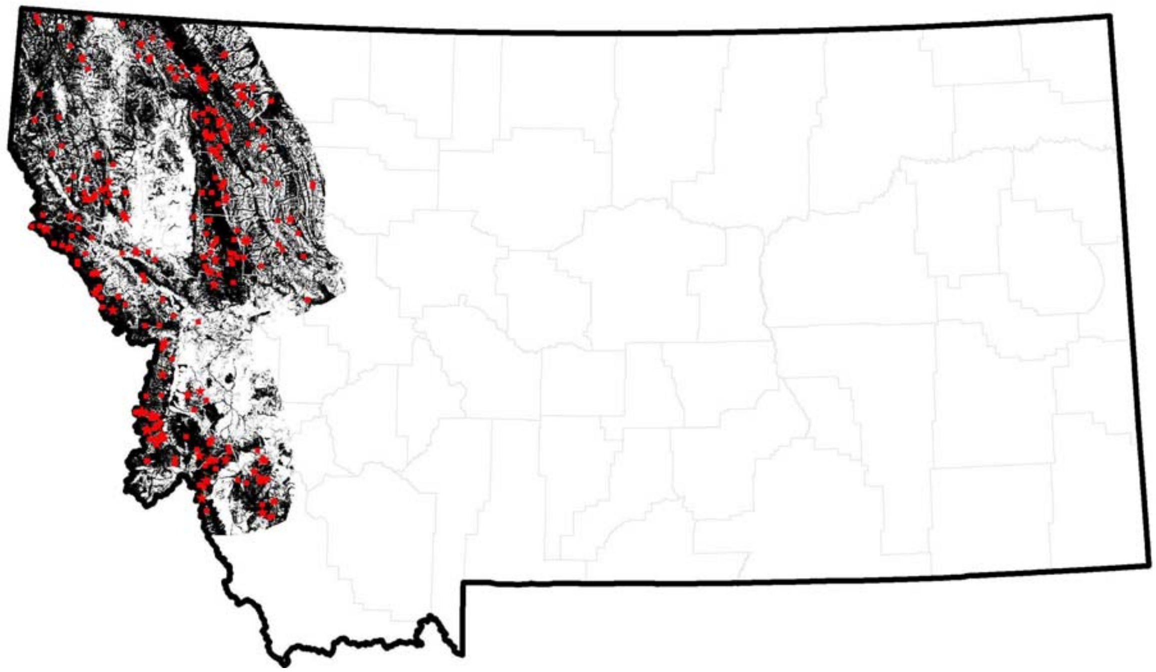
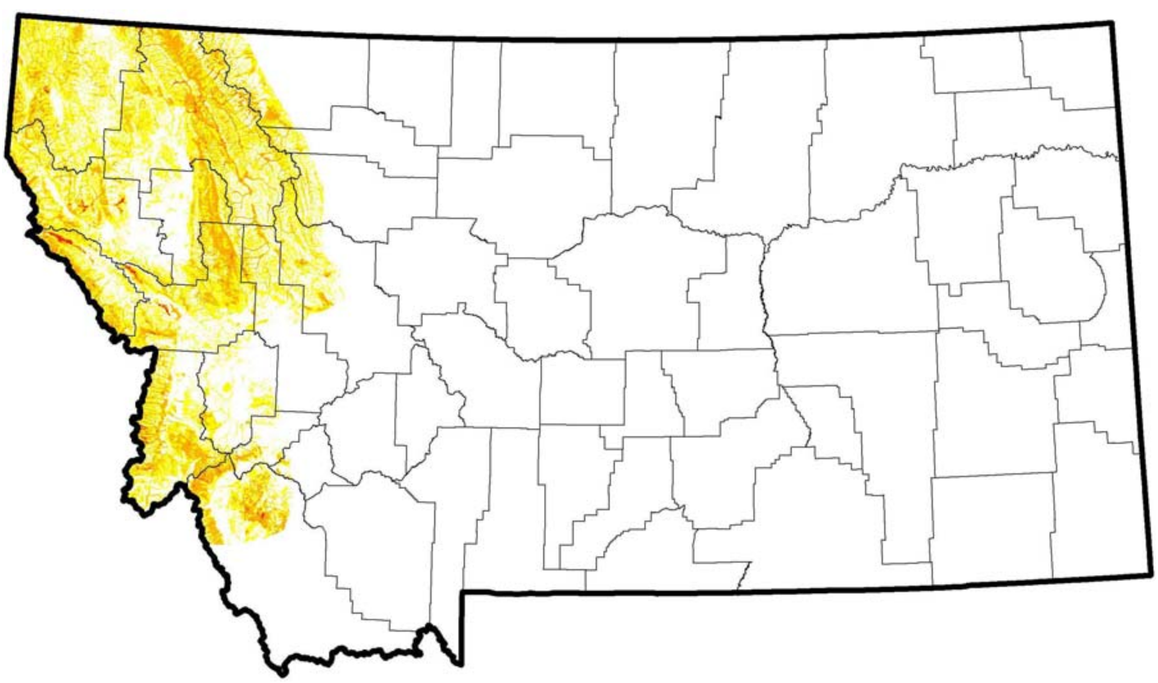


Figure D-5c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *A. montanus*.



Plains Spadefoot (*Spea bombifrons*)

In addition to identifying numerous areas currently lacking documentation within the known range of the species as highly suitable habitat, the state-wide continuous model (Figure D-6a) indicates that the known range may be extended: (1) in the Madison River valley near Ennis; (2) the Ruby River valley to just above the Ruby River Reservoir; (3) Blacktail Deer Creek southeast of Dillon; (4) the Paradise Valley south of Livingston; and (5) on portions of the Blackfeet and Crow Indian Reservations. Soil map units in the STATSGO soil layer are clearly valuable for predicting the distribution of the species, but extracting friable soils from this data is likely to improve the models since they seem to be much more common on these soil types (pers. obs.). Characteristics of other environmental variables of importance (Table 3.5) included warmer drier soils, slopes less than 15 degrees, annual precipitation of 20-50 cm, covertypes associated with riparian areas, and average maximum July temperatures greater than 29 °C. Solar radiation indices were the only variables that did not help improve models and they actually made them worse in the jackknife plots of test gain. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold (0.895 and 0.95, respectively) indicated good model fit and there was a 48 percent (128,305 km^2) reduction in predicted area from that predicted by GAP, which had a similar AVI (Tables 3.6 and 3.7, Figure D-6b). Evaluation with the lentic site survey data indicated low omission errors for both the low cutoff threshold and GAP models, but the low cutoff threshold performed slightly better for commission errors, overall map accuracy, and Kappa (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds generally appear to be appropriate at the scale of the species' range as well as the level of a public land survey section (Figure D-6c).

Figure D-6a. Continuous state-wide habitat suitability index for *S. bombifrons*.

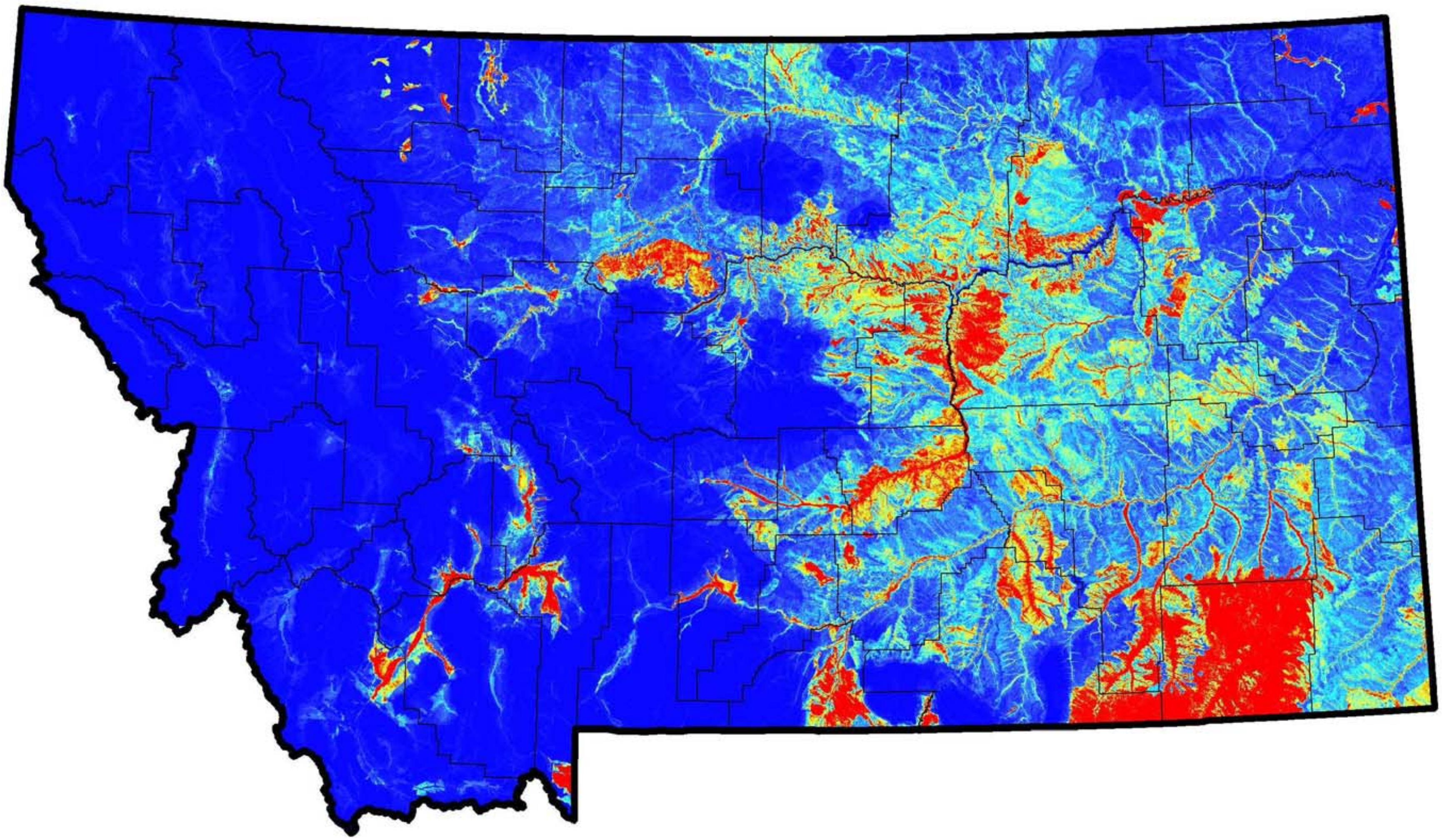


Figure D-6b. Predicted suitable habitat and training (squares) and test (stars) observations for *S. bombifrons*.

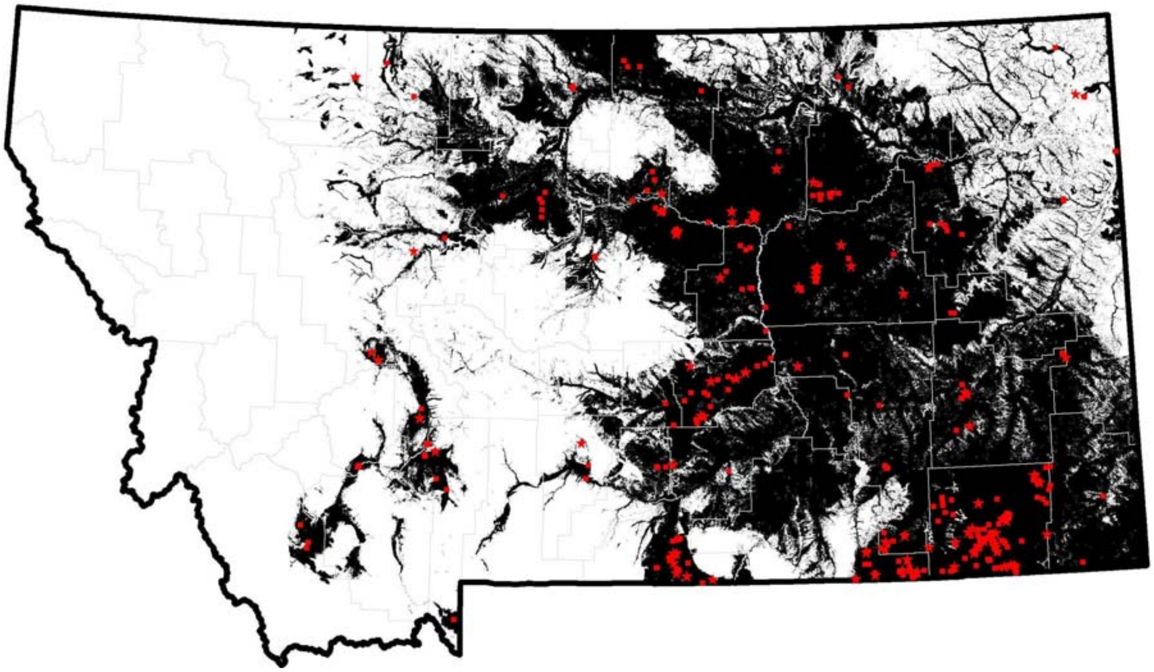
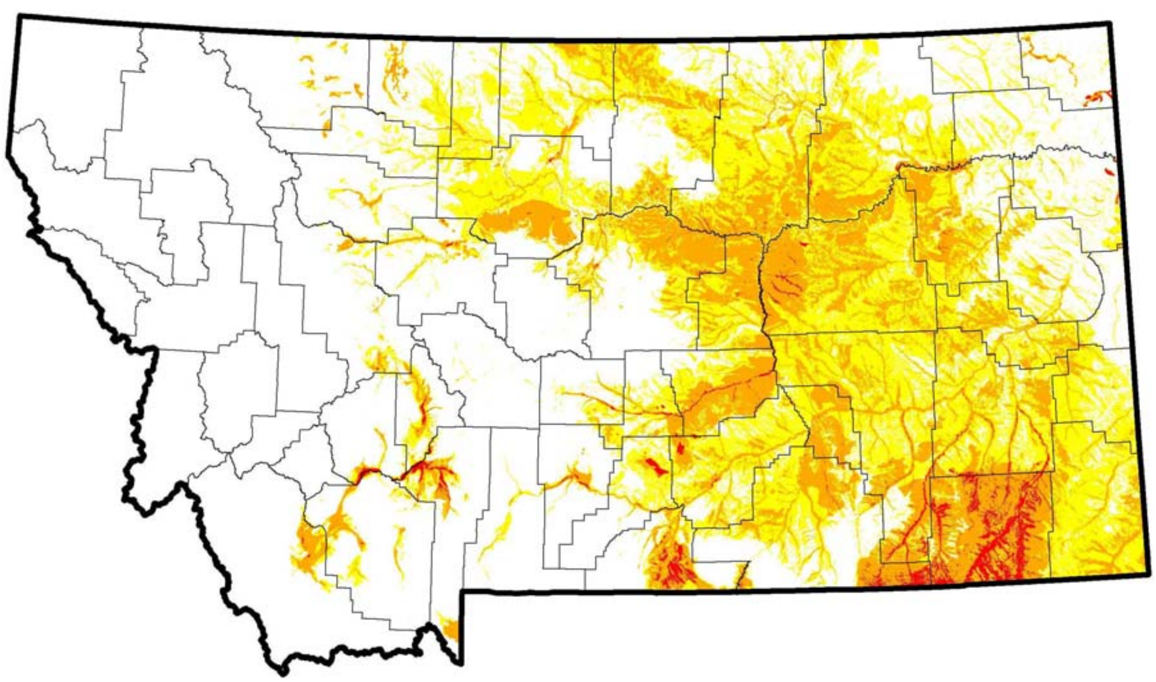


Figure D-6c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *S. bombifrons*.



Western Toad (*Bufo boreas*)

The state-wide continuous model indicates that suitable habitat exists in the Bighorn Mountains on the Crow Indian Reservation and in the Big Snowy Mountains outside the current known range of the species (Figures D-7a-b). While evidence for declines (Maxell et al. 2003) and failure to detect the species during recent surveys at the margins of their known range in these areas (pers. obs.) indicates this is unlikely, surveys focused on these areas are probably warranted. Characteristics of environmental variables of importance (Table 3.5) include cooler moister soil types, annual precipitation ranges of 50-100 cm, slopes of less than 30 degrees, and distances to major streams of under 1,000 m. Solar radiation indices were the only variables that did not help improve models and they actually made them worse in the jackknife plots of test gain. The AUC and AVI for test data evaluating the range-wide continuous model and low cutoff threshold (0.888 and 0.95, respectively) indicated good model fit and there was a 41 percent (58,708 km^2) reduction in predicted area from that predicted by GAP, which had a lower AVI (0.83) (Tables 3.6 and 3.7). Low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section (Figure D-7c). Evaluation with the lentic site survey data indicated low omission errors for both low cutoff threshold and GAP, but the low cutoff threshold model had higher commission error rates and, as a result, overall lower map accuracy than GAP (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance, the fact that single site visits fail to adequately assess true absence, and recent declines, which have left a number of apparently suitable breeding sites unoccupied. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and the magnitude of deviances is inversely correlated with predicted habitat suitability, indicating that the habitat suitability classifications performed well at the landscape scale.

Figure D-7a. Continuous state-wide habitat suitability index for *B. boreas*.

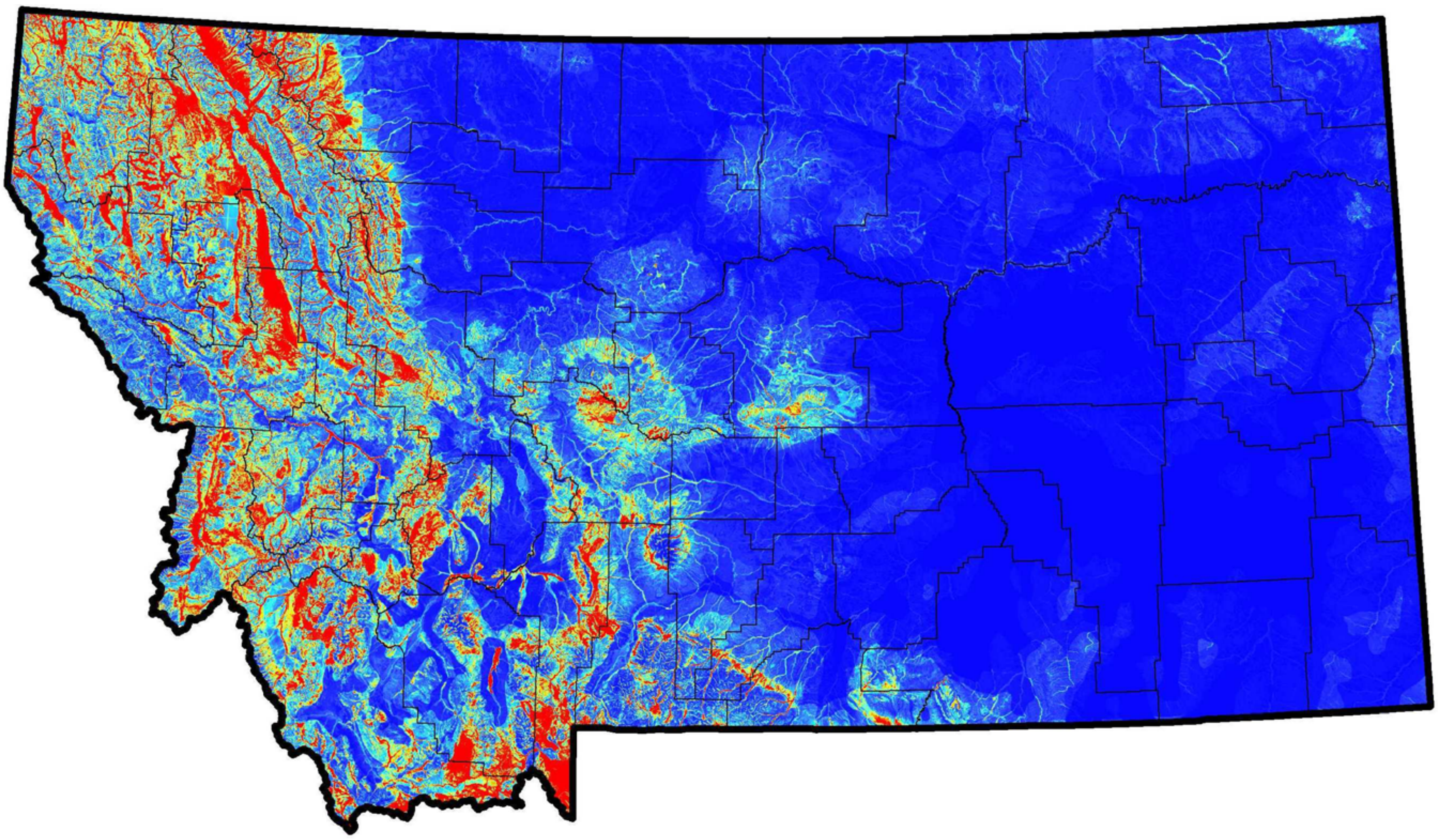


Figure D-7b. Predicted suitable habitat and training (squares) and test (stars) observations for *B. boreas*.

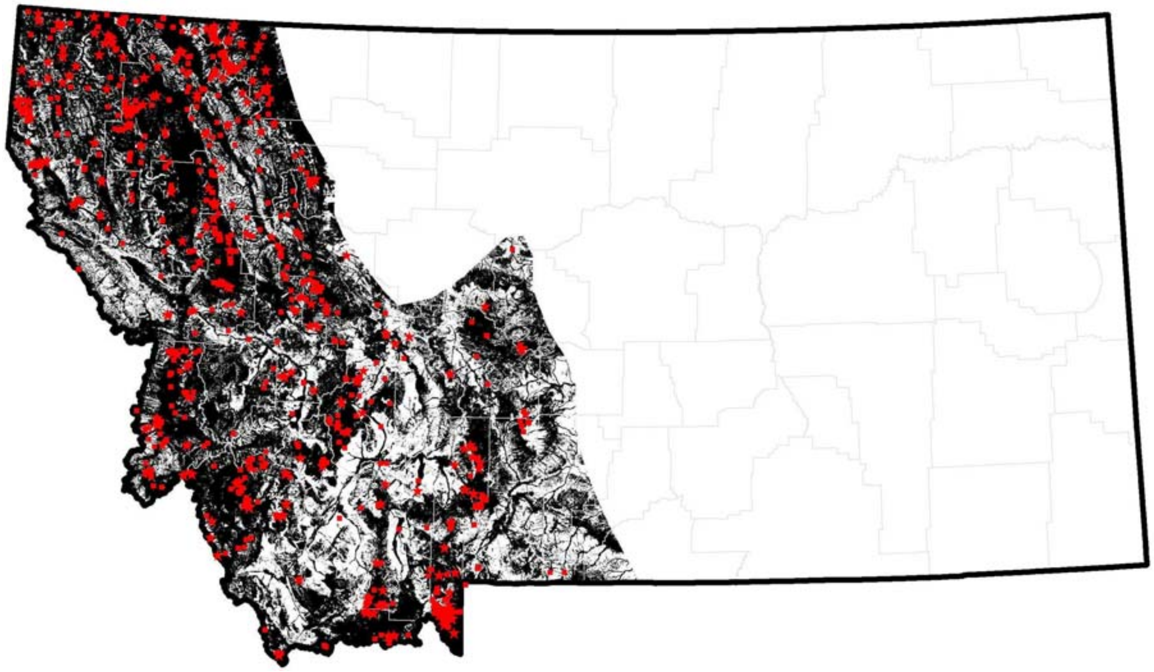
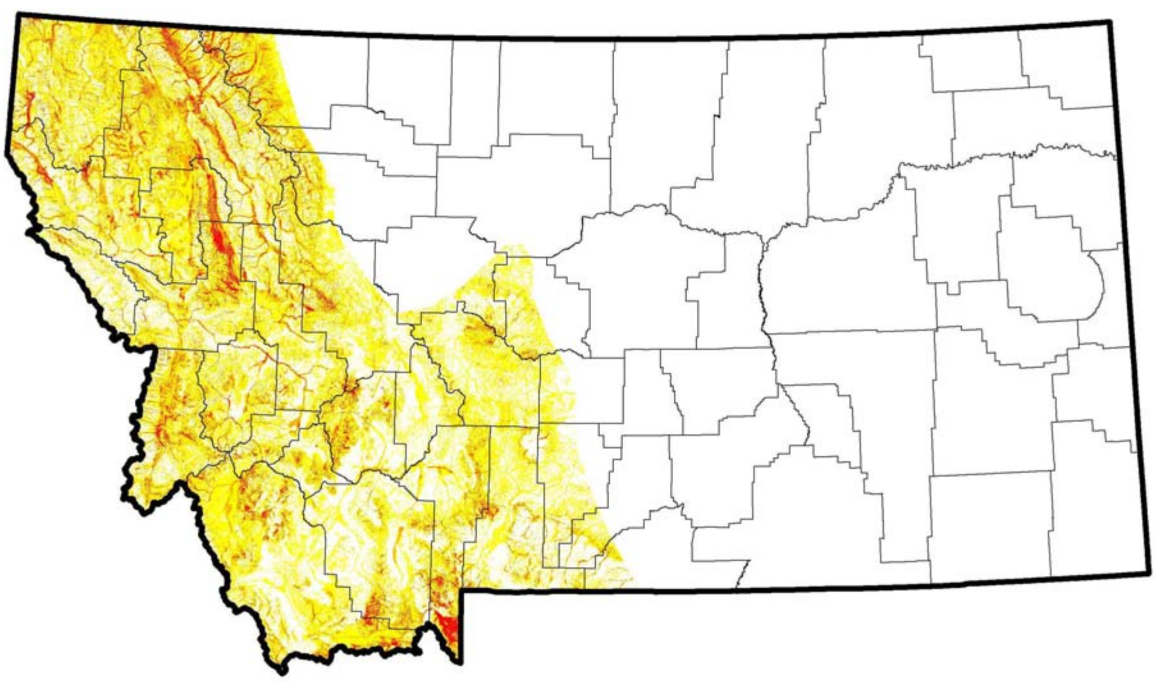


Figure D-7c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *B. boreas*.



Great Plains Toad (*Bufo cognatus*)

The state-wide continuous model indicates a number of areas within the known range of the species that currently lack observations and should be targeted with nocturnal calling surveys to fill in these data gaps (Figures D-8a-b). In addition, the state-wide continuous model indicates that the species' known range may be extended: (1) upstream along the Clark's Fork of the Yellowstone River; (2) upstream along the Little Bighorn River; (3) areas within the Milk and Marias River watersheds; and (4) portions of the upper Missouri River between Great Falls and Three Forks. Characteristics of environmental variables of importance include average maximum temperatures greater than 29 °C, elevations less than 1,250 m, warmer drier soils, and slopes less than 15 degrees. Solar radiation indices were the only variables that did not help improve models and they actually made them worse in the jackknife plots of test gain. Interestingly, while slope was valuable in training the models, it actually made them worse in the jackknife plots of test gain. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold (0.877 and 0.89, respectively) indicated reasonable model fit and there was a 48 percent (84,043 km^2) reduction in predicted area from that predicted by GAP, which had a lower AVI (0.77) (Figure D-8b, Tables 3.6 and 3.7). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, low map accuracy and low Kappa for both the low cutoff threshold and GAP models (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and the magnitude of deviances is inversely correlated with predicted habitat suitability. This correlated well with our field experience and we believe the low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species range as well as the level of a public land survey section in regions with adequate data (Figure D-8c). However, models may under predict distribution in regions with few occurrences.

Figure D-8a. Continuous state-wide habitat suitability index for *B. cognatus*.

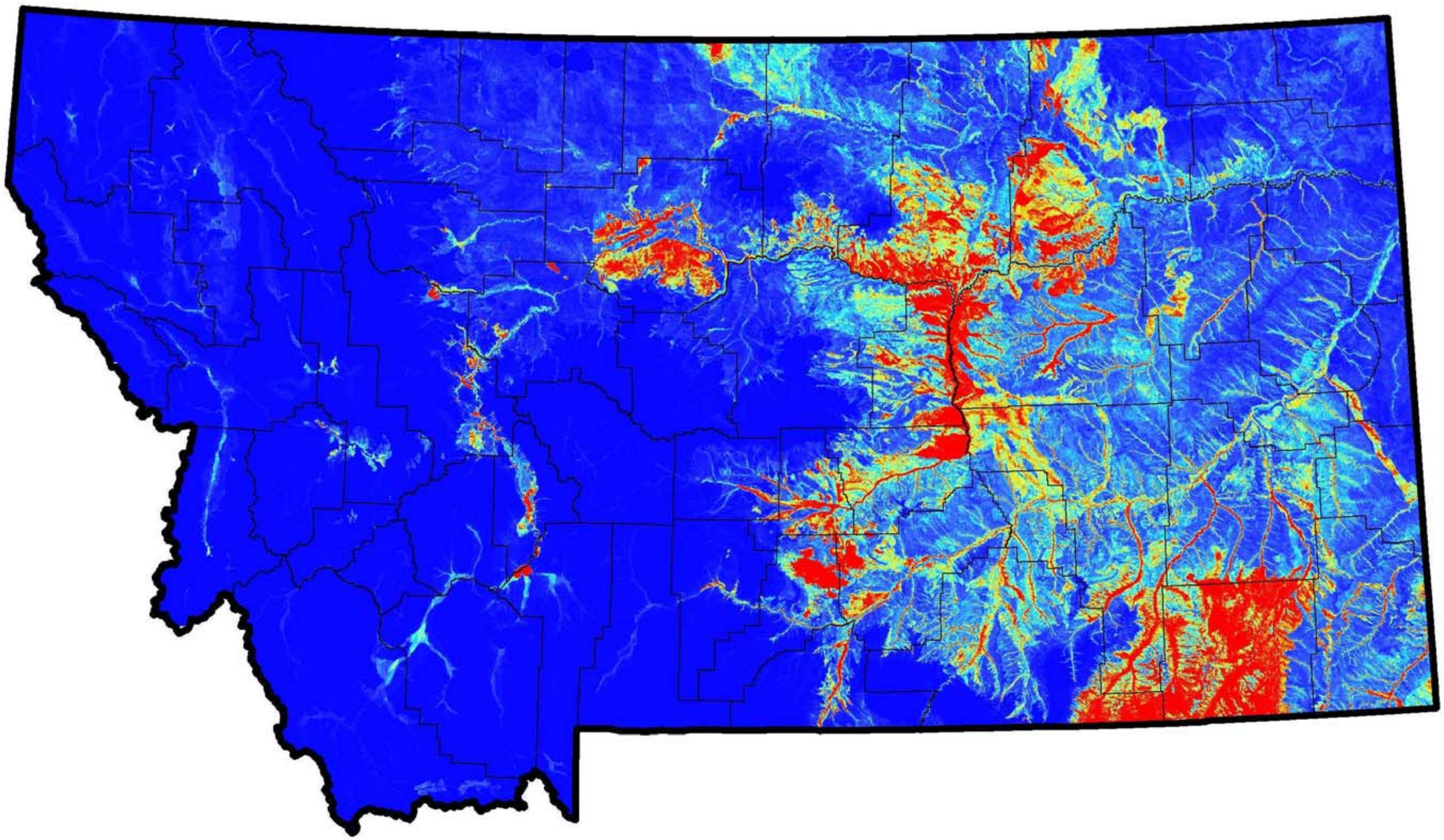


Figure D-8b. Predicted suitable habitat and training (squares) and test (stars) observations for *B. cognatus*.

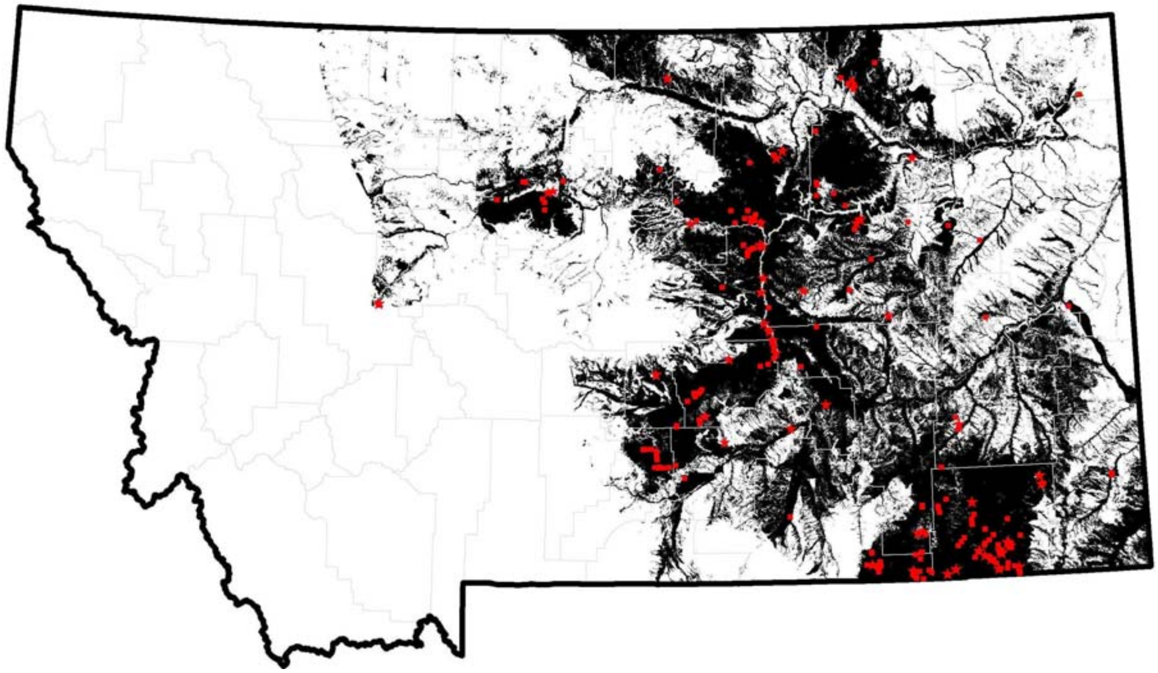
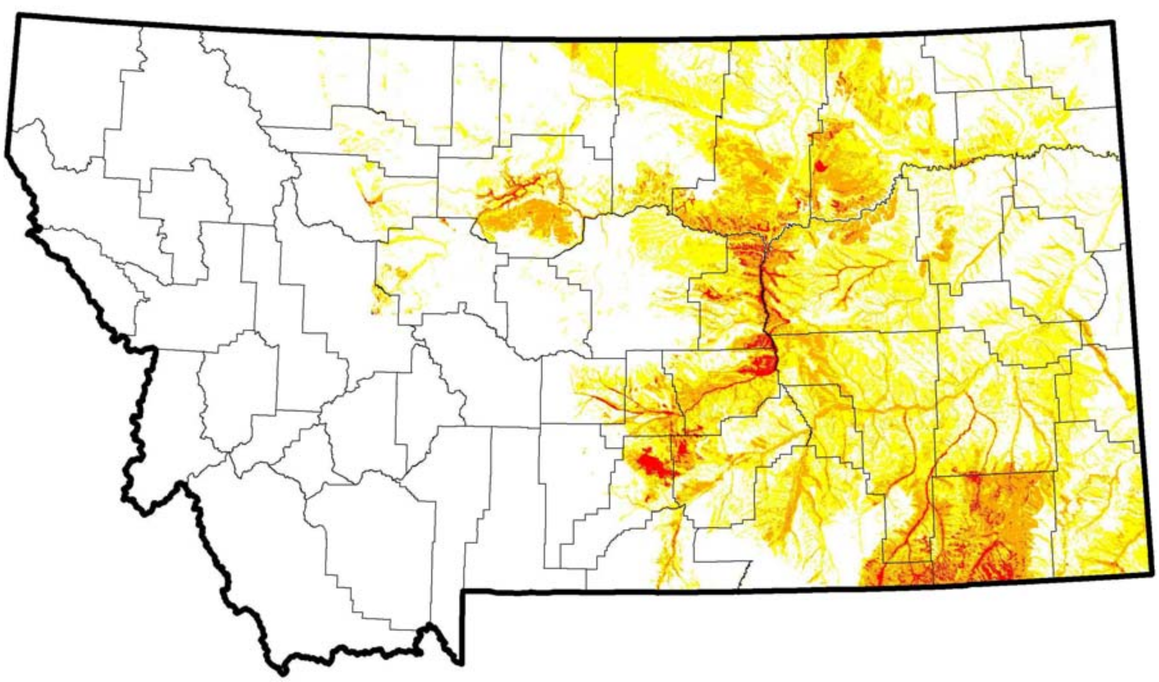


Figure D-8c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *B. cognatus*.



Woodhouse's Toad (*Bufo woodhousii*)

The state-wide continuous model indicates a number of areas within the known range of the species that currently lack observations and should be targeted with nocturnal calling surveys to fill in these data gaps (Figures D-9a-b). In addition, the state-wide continuous model indicates that the species' known range may be extended: (1) northward to the border with Canada where the species has not yet been reported (Russell and Bauer 2000, Secoy and Vincent 1976) along major tributaries to the Milk, Missouri, and Marias Rivers; (2) up the Clark's Fork of the Yellowstone River; and (3) up the Little Bighorn River. Characteristics of environmental variables of importance (Table 3.5) include average maximum temperatures greater than 29 °C, areas close to major streams, and elevations less than 1,200 m. Solar radiation indices and soil temperature and moisture regime did not help improve models and they actually made them worse in the jackknife plots of test gain. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.94 and 0.98, respectively) indicated good model fit and there was a 57 percent (95,611 km^2) reduction in predicted area from that predicted by GAP, which had a lower AVI (0.92) (Figure D-9b, Tables 3.6 and 3.7). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, moderate values for map accuracy and low Kappa for both the low cutoff threshold and GAP models, with low cutoff threshold models outperforming GAP in all situations (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and the magnitude of deviances is inversely correlated with predicted habitat suitability. Thus, low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section in most areas. However, the species' distribution may be slightly under represented in the areas between the Missouri and Yellowstone Rivers that lack some baseline survey effort (Figure D-9b-c).

Figure D-9a. Continuous state-wide habitat suitability index for *B. woodhousii*.

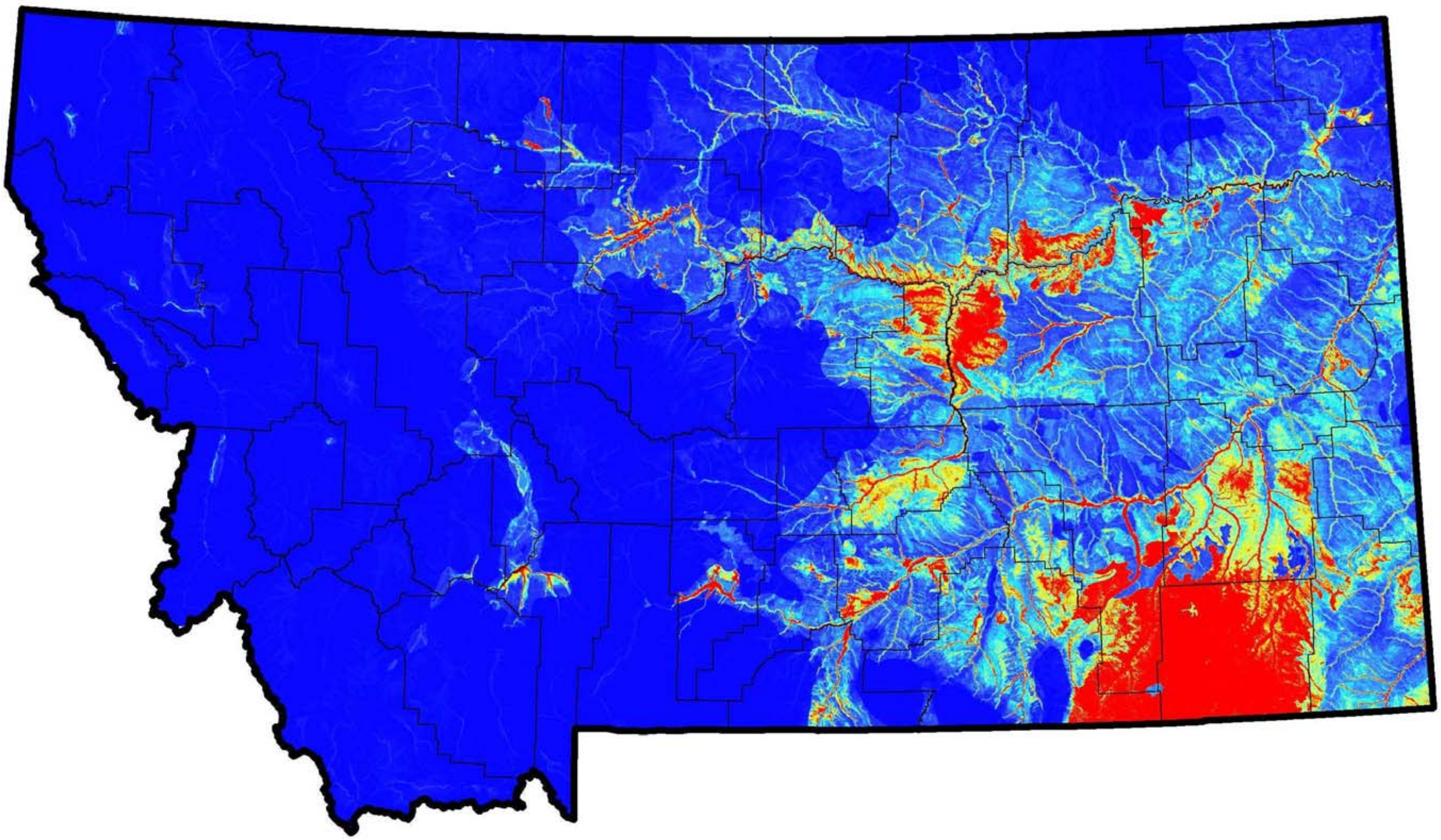


Figure D-9b. Predicted suitable habitat and training (squares) and test (stars) observations for *B. woodhousii*.

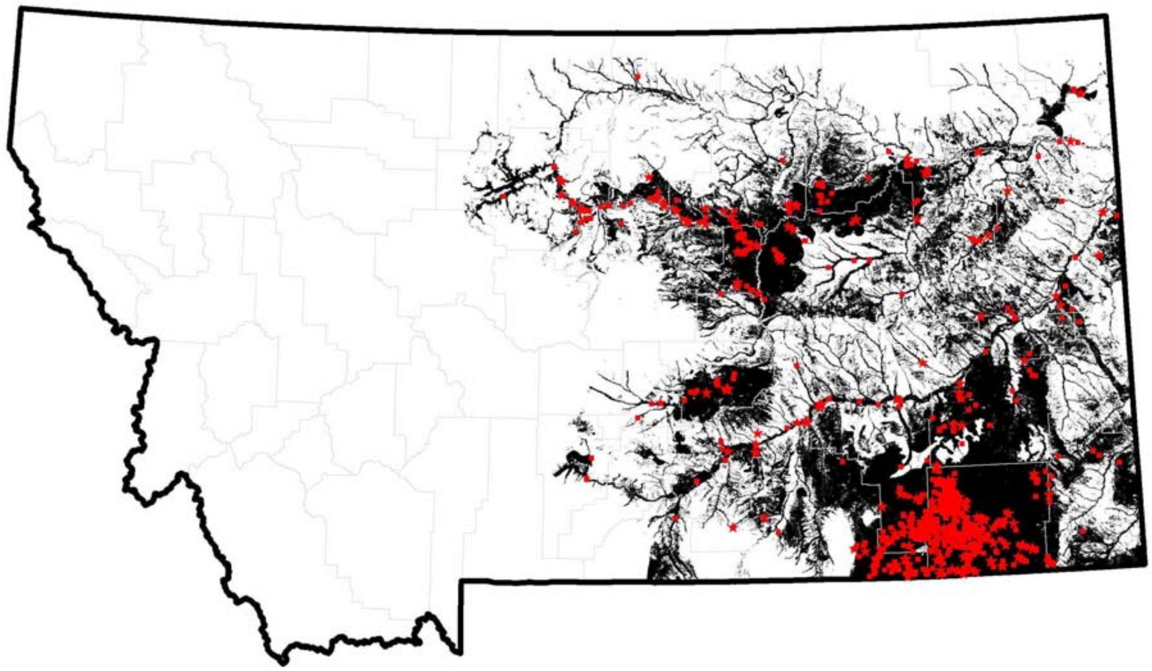
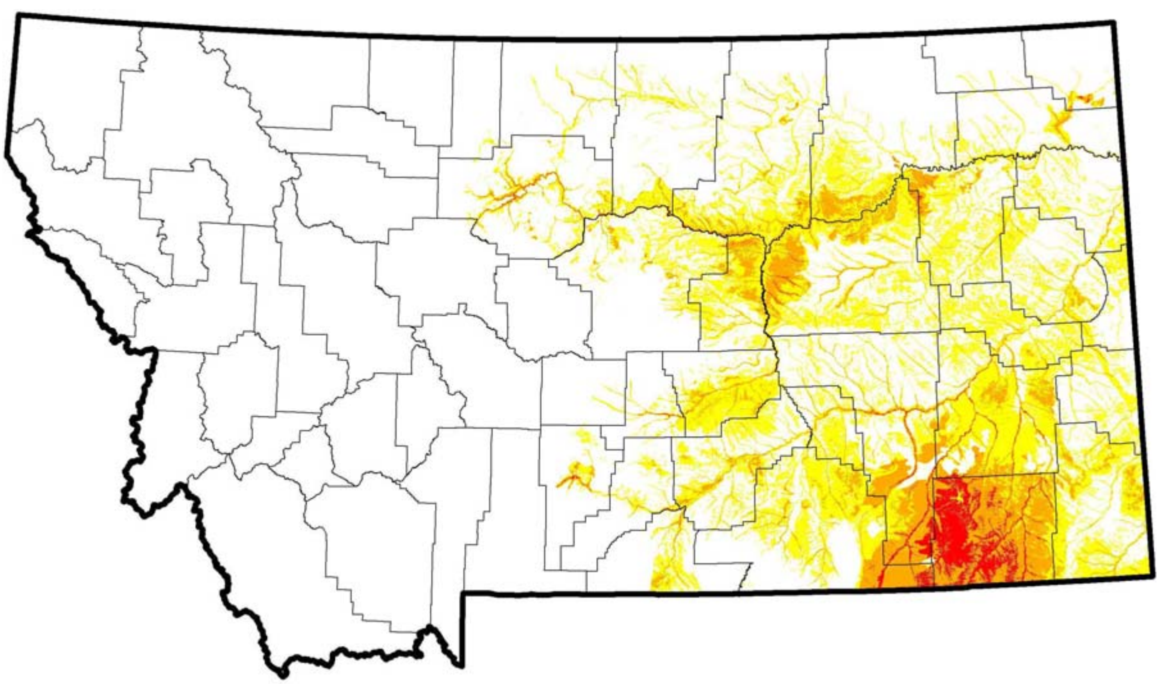


Figure D-9c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *B. woodhousii*.



Boreal Chorus Frog (*Pseudacris maculata*)

The state-wide continuous model indicates a number of areas at the margins of the known range of the species that currently lack observations and should be targeted with nocturnal calling surveys to fill in these data gaps (Figure D-10a). These include: (1) the Paradise Valley; (2) the Madison River Valley; (3) the Ruby Valley; (4) the Jefferson and Missouri Rivers between Helena and Dillon; and (5) the upper Smith River. Characteristics of environmental variables of importance include slopes less than 10 degrees, wetland related land cover types, and areas within 1,000 m of major streams. Solar radiation indices were the only variables that did not help improve models and they actually made them worse in the jackknife plots of test gain. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold model (0.884 and 0.97, respectively) indicate good model fit and there was a 33 percent (92,303 km^2) reduction in predicted area from that predicted by GAP, which had a lower AVI (0.93) (Figure D-10b, Tables 3.6 and 3.7). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, moderate values for map accuracy and low Kappa for both the low cutoff threshold and GAP models, with low cutoff threshold models outperforming GAP for omission errors (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section in most areas (Figure D-10b-c). However, the species' distribution may be slightly under represented in a few areas dominated by private agricultural land that lack some baseline survey effort in eastern Montana.

Figure D-10a. Continuous state-wide habitat suitability index for *P. maculata*.

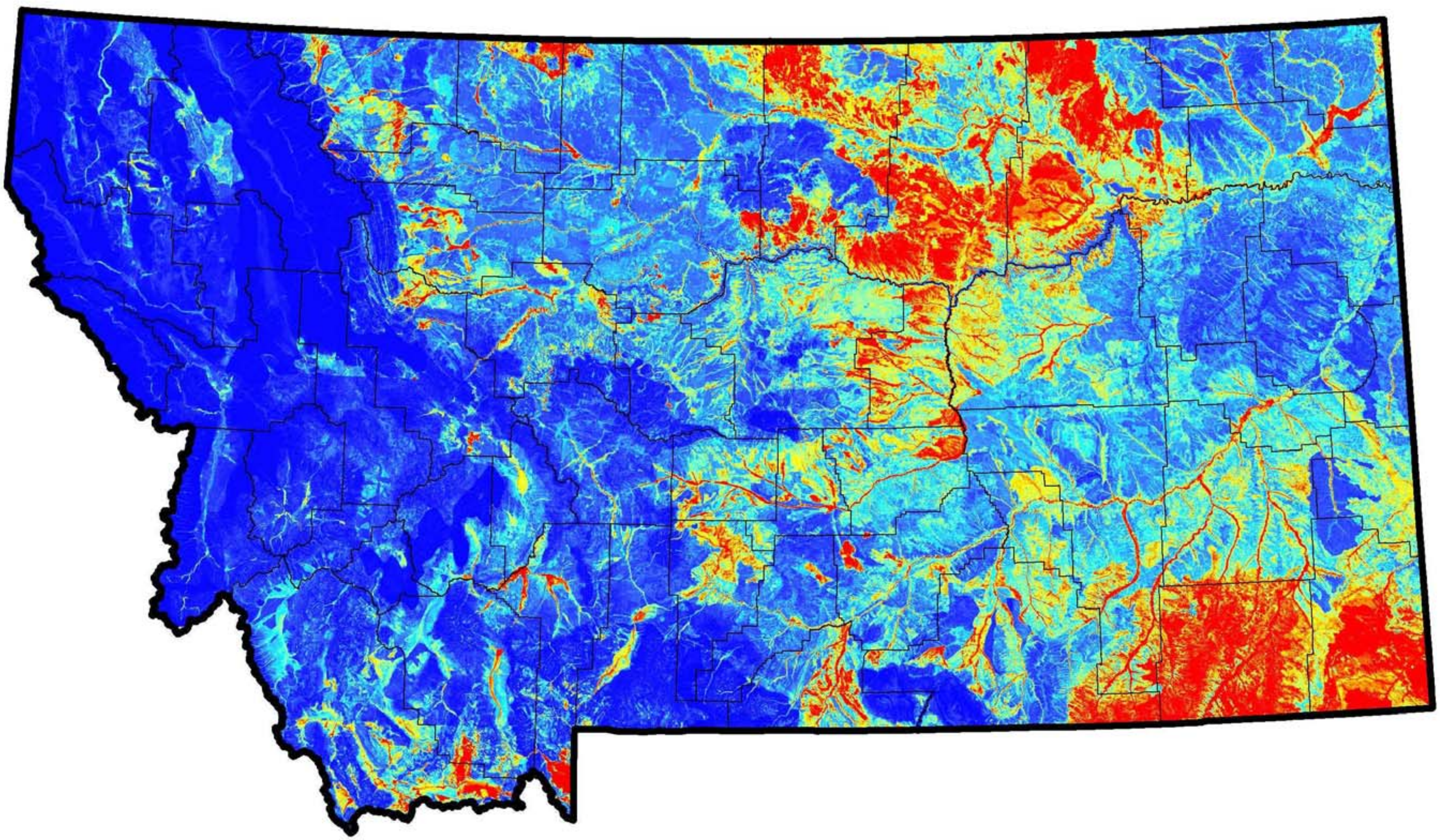


Figure D-10b. Predicted suitable habitat and training (squares) and test (stars) observations for *P. maculata*.

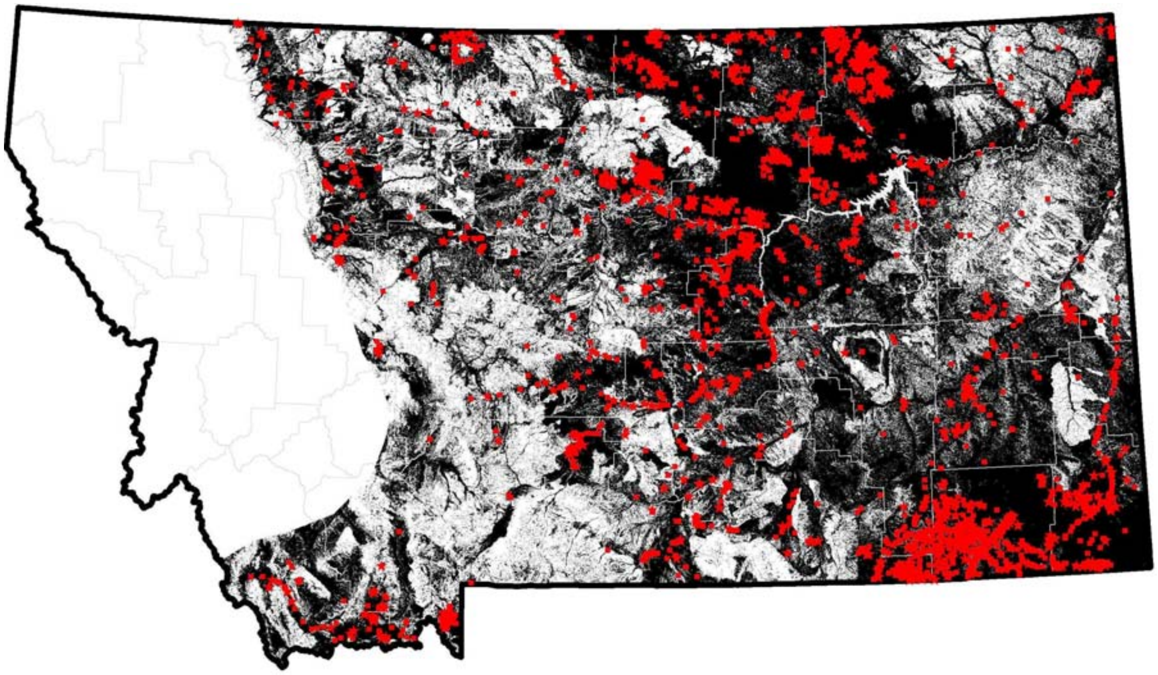
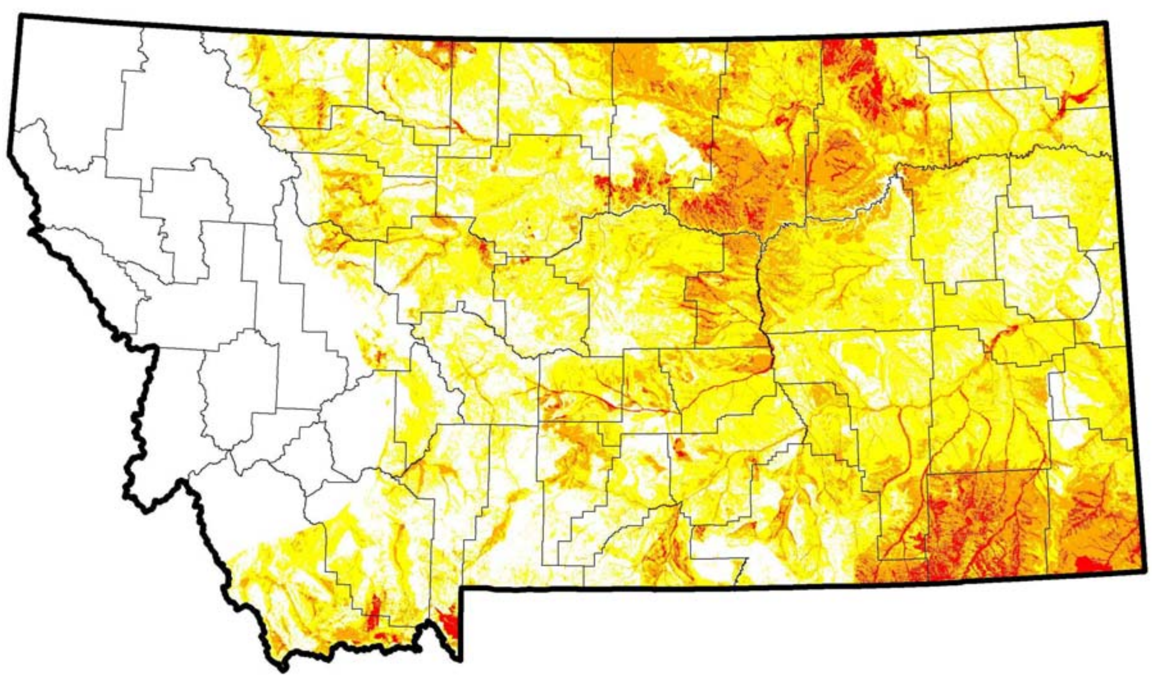


Figure D-10c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *P. maculata*.



Pacific Treefrog (*Pseudacris regilla*)

The state-wide continuous model (Figure D-11a) indicates that the species' range may extend into: (1) the Seeley and Swan Valleys; (2) portions of the lower North and South Forks of the Flathead River drainage; (3) the lower portions of Rock Creek; (4) portions of the lower Clark Fork River drainage between Missoula and Superior; and (5) portions of the Garnet Range. These areas all seem reasonable, but recent calling surveys in these areas have failed to detect them (Maxell et al. 2003, pers. obs.) and it seems likely that these areas have not been occupied in recent decades or were never colonized. Minimum average January temperature (optimally greater than 15 °C) was ranked as the most important variable at the state-wide scale, but was not important in range-wide models. Similarly soil temperature and moisture regime (warmer moister soils) was important at the state-wide scale, but was not important in range-wide models. Other characteristics of environmental variables of importance (Table 3.5) included elevations below 1,600 m, slopes usually less than 10 degrees, annual precipitation greater than 50 cm, and terrain with at least some level of ruggedness. AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold model (0.933 and 0.97, respectively) indicate good model fit (Figure D-11b, Table 3.6). There was a 19 percent (5,101 km^2) reduction in predicted area from that predicted by GAP, which had a lower AVI (0.84) (Tables 3.6 and 3.7). Evaluation with the lentic site survey data indicated low omission error rates and high commission error rates for both the low cutoff threshold and GAP models (Table 3.8). Map accuracy and Kappa for GAP was superior to the low cutoff threshold model, probably as a result of the large number of sites represented by the slightly higher commission error rate for the lower cutoff threshold model (Table 3.5). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section in most areas. However, the species' distribution is likely over predicted in a number of areas south of the Flathead Valley because populations in this region are disjunct and somewhat isolated (Figure D11b-c, Maxell et al. 2003). Models may assist with the location of additional isolated populations in this region.

Figure D-11a. Continuous state-wide habitat suitability index for *P. regilla*.

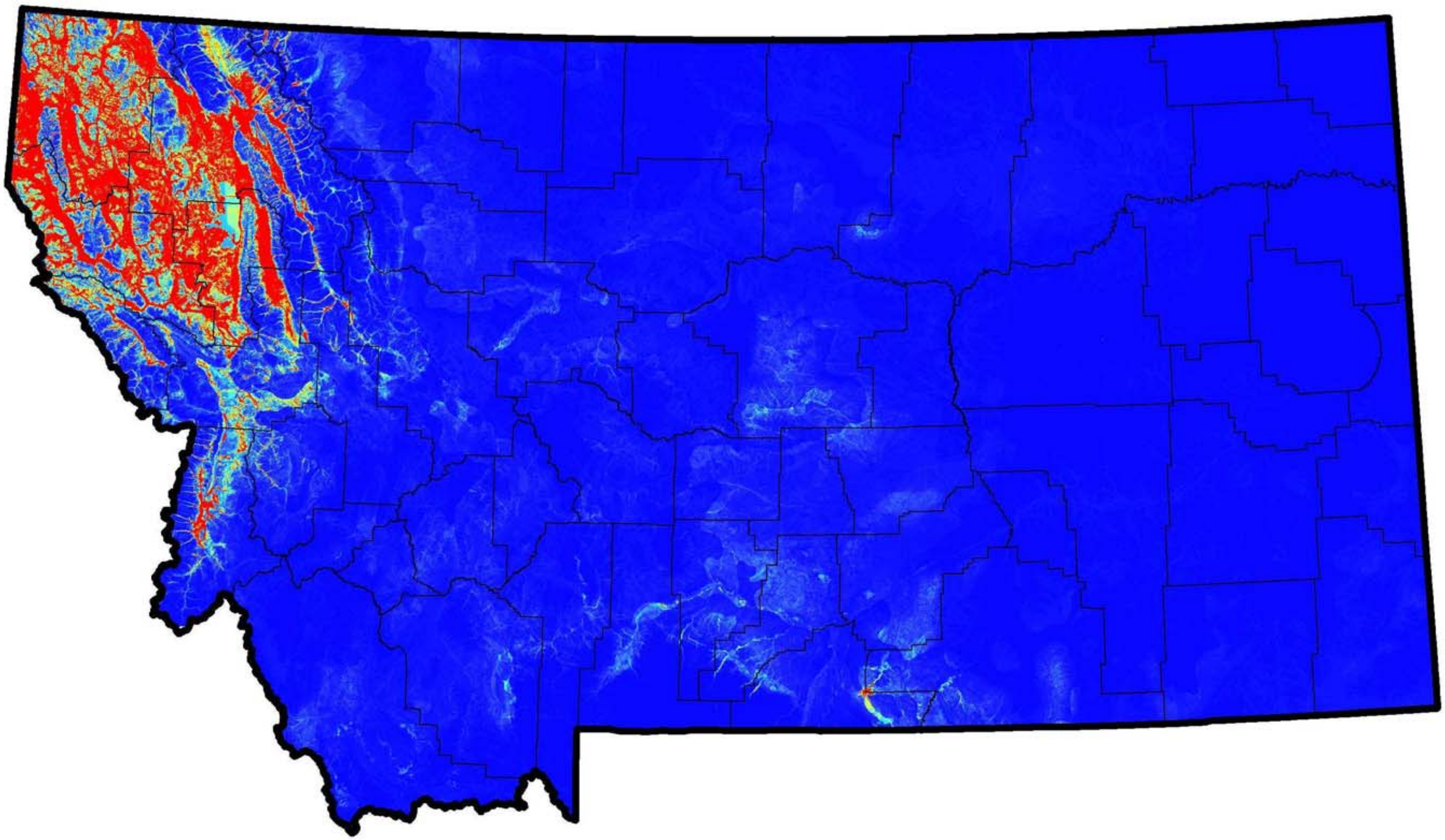


Figure D-11b. Predicted suitable habitat and training (squares) and test (stars) observations for *P. regilla*.

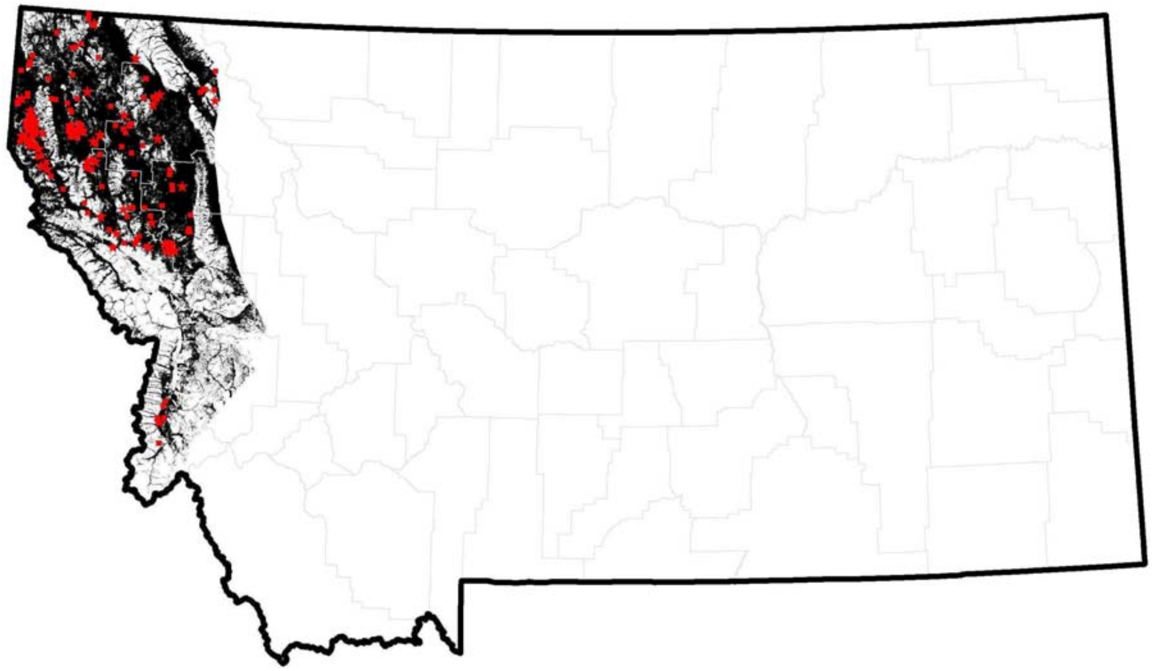
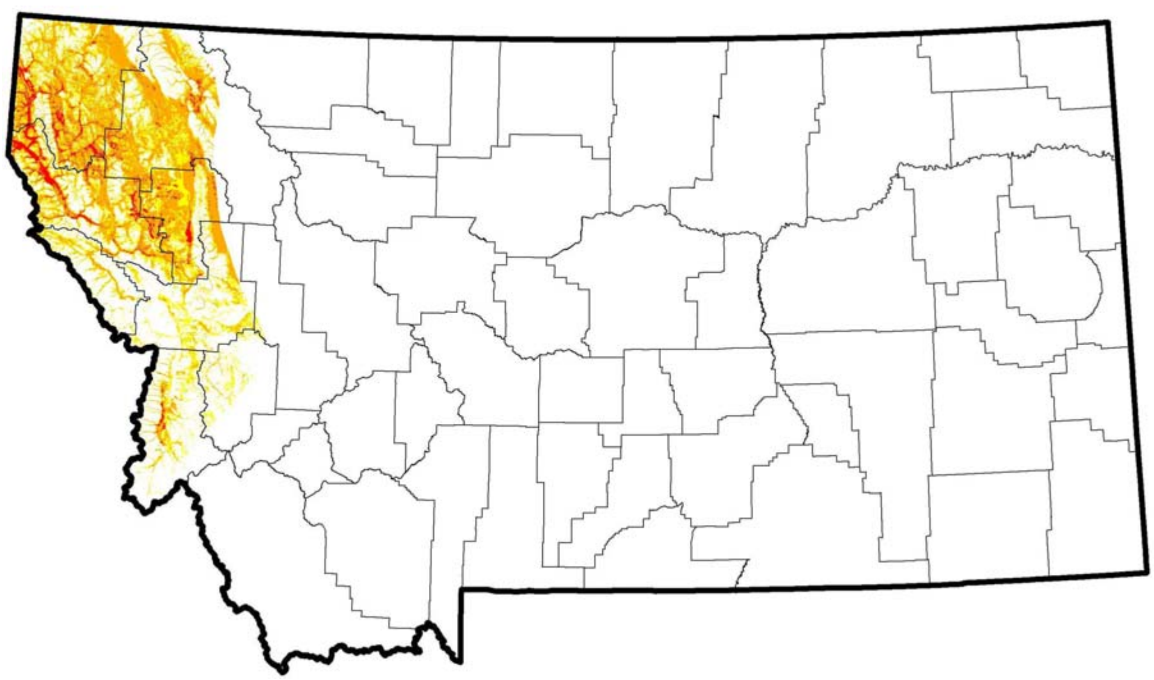


Figure D-11c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *P. regilla*.



American Bullfrog (*Rana catesbeiana*)

Only a state-wide continuous model was run for *R. catesbeiana* because this exotic species has been found in a number of new localities in recent years and while the extent of their range is uncertain, there is a great need to identify areas of potential spread because they represent a major threat to native vertebrate and invertebrate populations (Bury and Whelan 1984, Maxell 2000, Maxell et al. 2003, Werner et al. 2004). The model (Figures D-12a-b) indicates that a number of additional areas are capable of supporting populations, including: (1) the Flathead Valley; (2) a number of major drainages in northwest Montana including the Bull, Fisher, and Thompson Rivers; (3) the Missouri River around Helena; (4) the Paradise Valley; (5) areas south of Billings around Bighorn Lake; and (6) streams around Lewistown. Characteristics of environmental variables of importance (Table 3.5) included average minimum January temperatures greater than -12 °C (ideally greater than -9 °C), areas within 1,000 m of major streams, and flatter slopes. No test data was available for evaluation of the model by AUC or AVI, but GAP AVI was very low (0.03) because GAP was limited to areas around major rivers and lakes resulting in a small predicted area (Table 3.6, Hart et al. 1998). The low cutoff threshold probably greatly over predicts the distribution of the species right now; resulting in a 56,841 percent (34,673 km^2) increase from the GAP prediction (Figure D-12b, Table 3.7). However, the fact that average minimum January temperature appears to be important for limiting the distribution of the species indicates that these areas may be an appropriate representation of potential spread under documented ongoing increases in winter temperatures from climate change (Mote 2003). This highlights the importance of undertaking control measures on currently established populations and educating the public to reduce or eliminate future introductions.

Figure D-12a. Continuous state-wide habitat suitability index for *R. catesbeiana*.

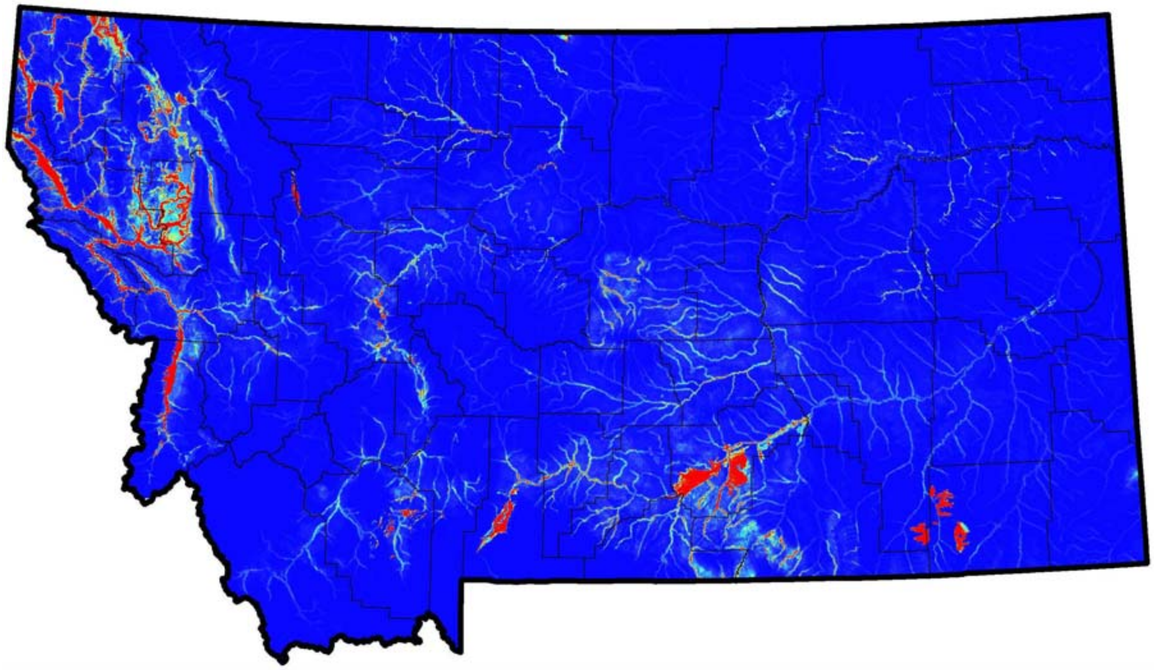


Figure D-12b. Predicted suitable habitat and training observations for *R. catesbeiana*.

Columbia Spotted Frog (*Rana luteiventris*)

The state-wide continuous model indicates that the known range of the species could potentially be extended eastward into: (1) the Bighorn Mountains on the Crow Indian Reservation; and (2) the Big Snowy Mountains (Figure D-13a). It is possible that the species has gone undetected or unreported in both areas, but the Bighorn Mountains are more likely given the presence of populations of *R. luteiventris* in adjacent areas of this mountain range in Wyoming (Dunlap 1977, Funk et al. 2008). Characteristics of environmental variables of importance (Table 3.5) include slopes less than 15 to 20 degrees, annual precipitation ranges of 50-230 cm, cooler moister soil types, average maximum July temperatures less than 27 °C, and wetland, evergreen, or transitional land cover types. Solar radiation indices were the only variables that were essentially of no help in improving models as indicated by jackknife plots of training and test gain and AUC. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.936 and 0.98, respectively) indicated good model fit and the AVI was much better than the GAP model (0.53), which was probably unrealistic in that it simply turned on streams within appropriate habitat types (Table 3.6, Hart et al. 1998). As a result the low cutoff threshold predicted a 155% (50,032 km^2) larger area than GAP (Table 3.7). Evaluation with the lentic site survey data indicated relatively high omission errors for the low cutoff threshold (0.128) and even higher omission error rates for GAP (0.362) (Table 3.8). However, moderate levels of commission errors for both the low cutoff and GAP resulted in relatively high overall map accuracy (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section in most areas. However, the low cutoff threshold probably under predicts suitable habitat in some valley bottoms east of the Continental Divide where few surveys have been performed (Figures D-13a-b).

Figure D-13a. Continuous state-wide habitat suitability index for *R. luteiventris*.

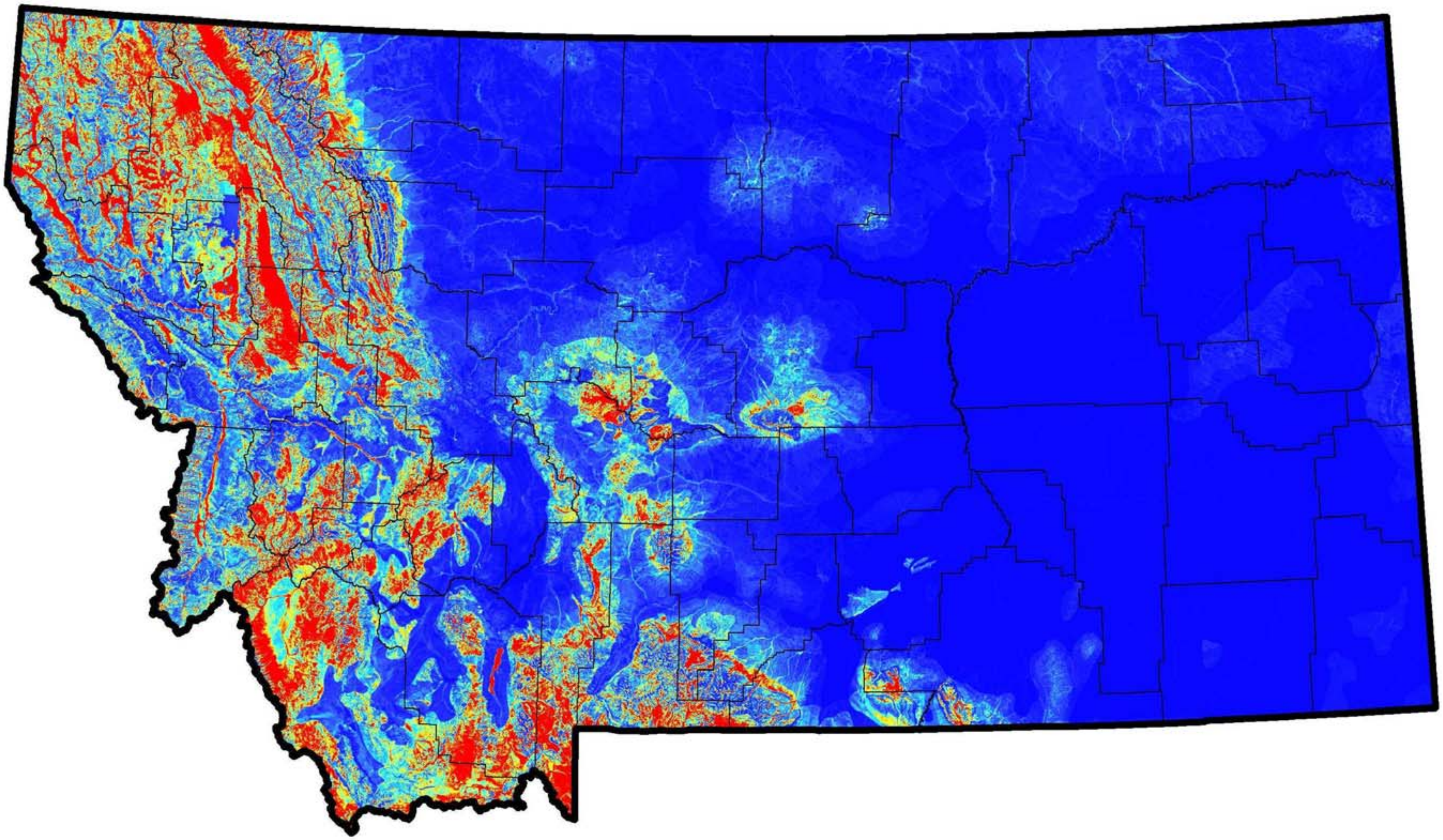


Figure D-13b. Predicted suitable habitat and training (squares) and test (stars) observations for *R. luteiventris*.

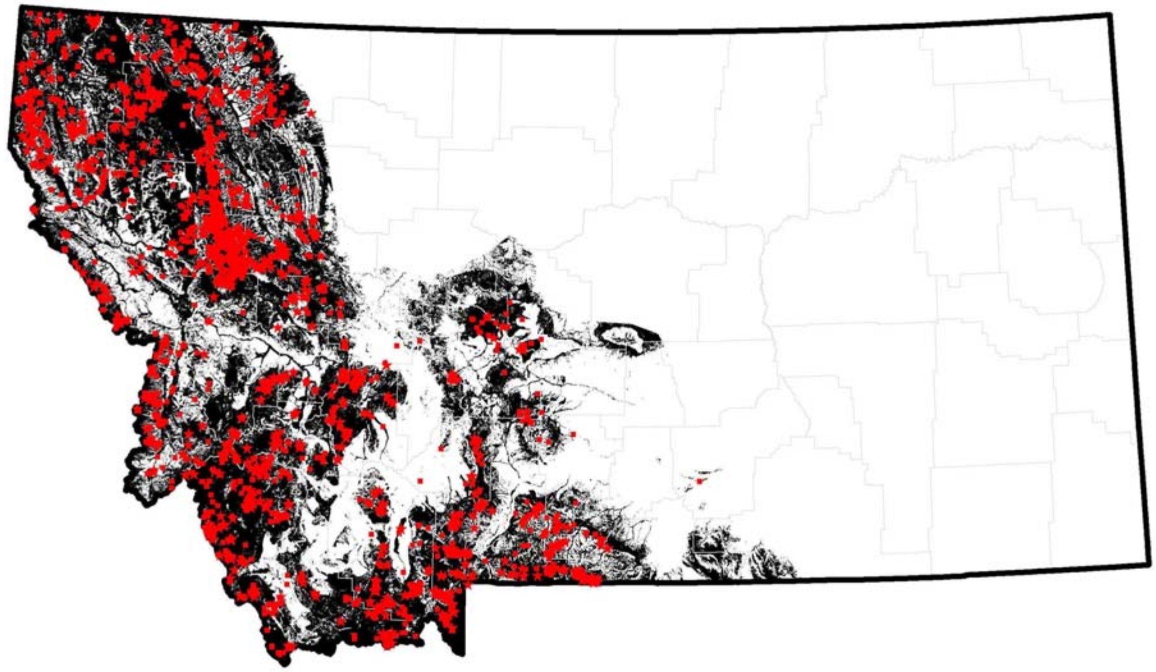
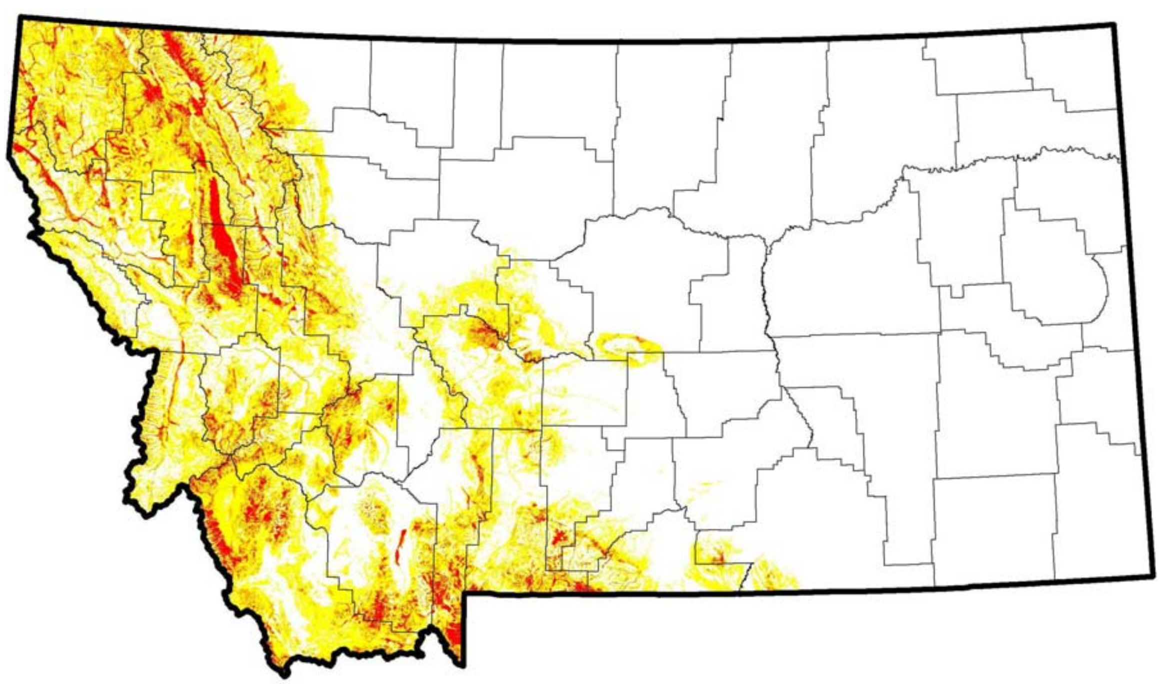


Figure D-13c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *R. luteiventris*.



Northern Leopard Frog (*Rana pipiens*)

The state-wide continuous model supports an historic distribution in western Montana prior to declines (Werner 2003, Werner et al. 2004) that was restricted to major valley bottoms and interestingly, despite a number of records in the lower Flathead Valley, predicts more suitable habitat extending from just north of Flathead Lake up to the Canadian border near Eureka where the last historic populations still exist (Figure D-14a). Characteristics of environmental variables of importance (Table 3.5) include elevations less than 1,500 m, areas within 500 m of major streams, and slopes less than 10 degrees. Solar radiation indices were the only variables that were essentially of no help in improving models as indicated by jackknife plots of training and test gain and AUC. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff models (0.923 and 0.96, respectively) indicated good model fit and much better fit than the AVI for GAP (0.53), which was based on streams within suitable cover types (Figure D-14b, Table 3.6, Hart et al. 1998). The GAP models were essentially correct in focusing on stream habitats, but a 500-1,000 m or larger buffer around streams would have been required to raise the AVI to levels achieved by the low cutoff threshold model which predicted a 131 percent (67,762 km^2) larger area than GAP (Table 3.7). Evaluation with the lentic site survey data indicated low omission errors for the low cutoff threshold, but GAP suffered from higher omission error rates (Table 3.8). Both the low cutoff threshold and GAP models had high commission error rates and low Kappa values, but GAP had higher overall map accuracy. High commission errors and low Kappa values for the low cutoff threshold likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability classes seem to more realistically represent the distribution of the species than the simple GAP model of streams within suitable habitat cover types (Figure D-14c). However, a few areas that lack surveys may be under predicted and a few areas that have received a lot of survey effort may be over predicted into drier areas that are unlikely to support populations.

Figure D-14a. Continuous state-wide habitat suitability index for *R. pipiens*.

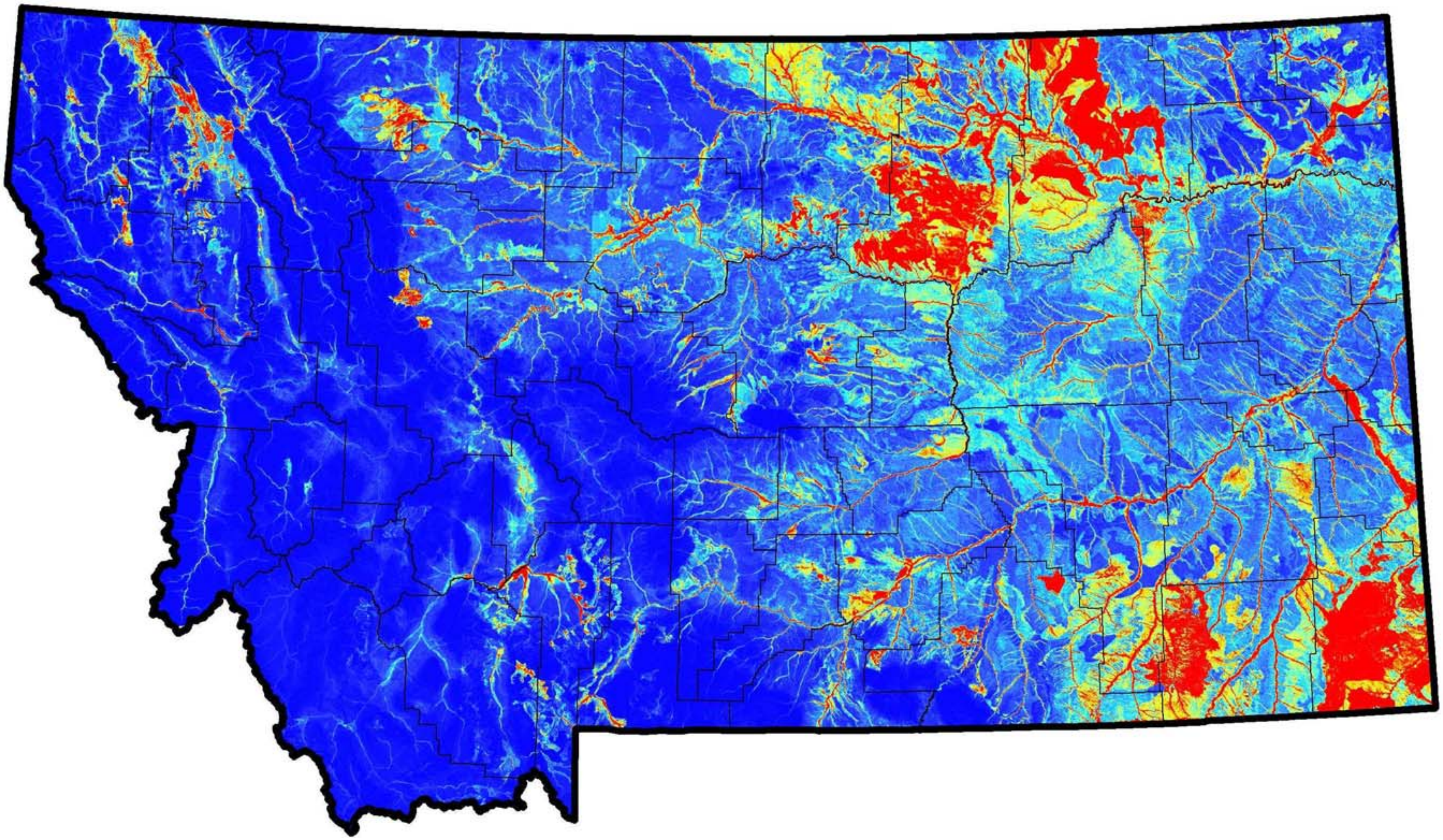


Figure D-14b. Predicted suitable habitat and training (squares) and test (stars) observations for *R. pipiens*. Only current range is shown.

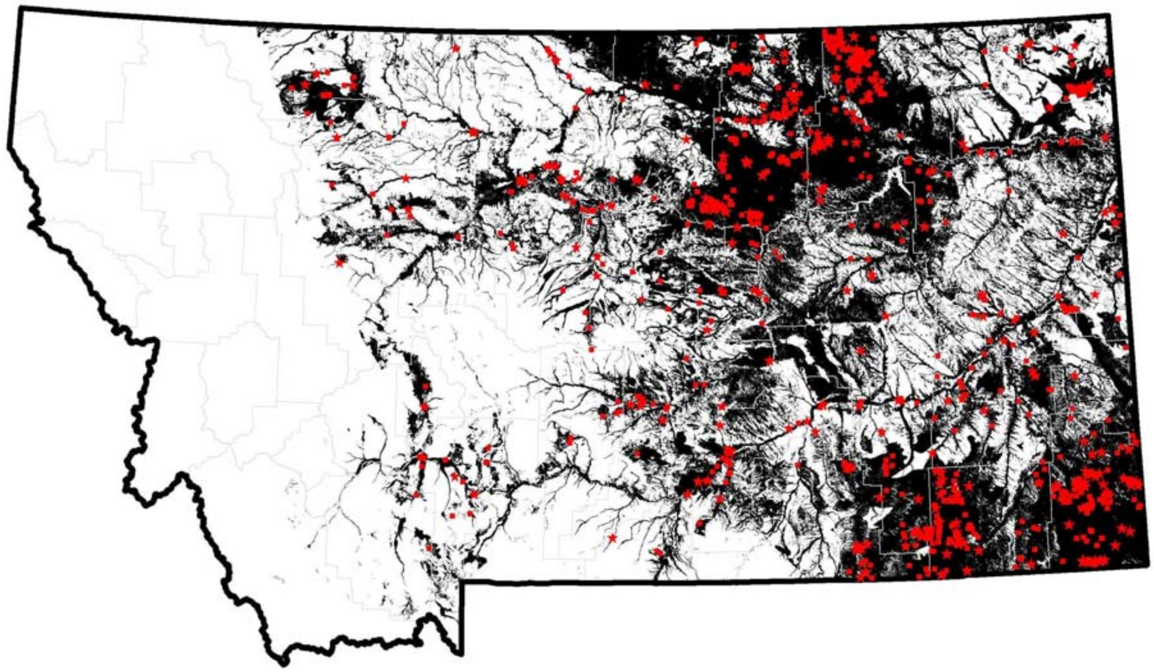
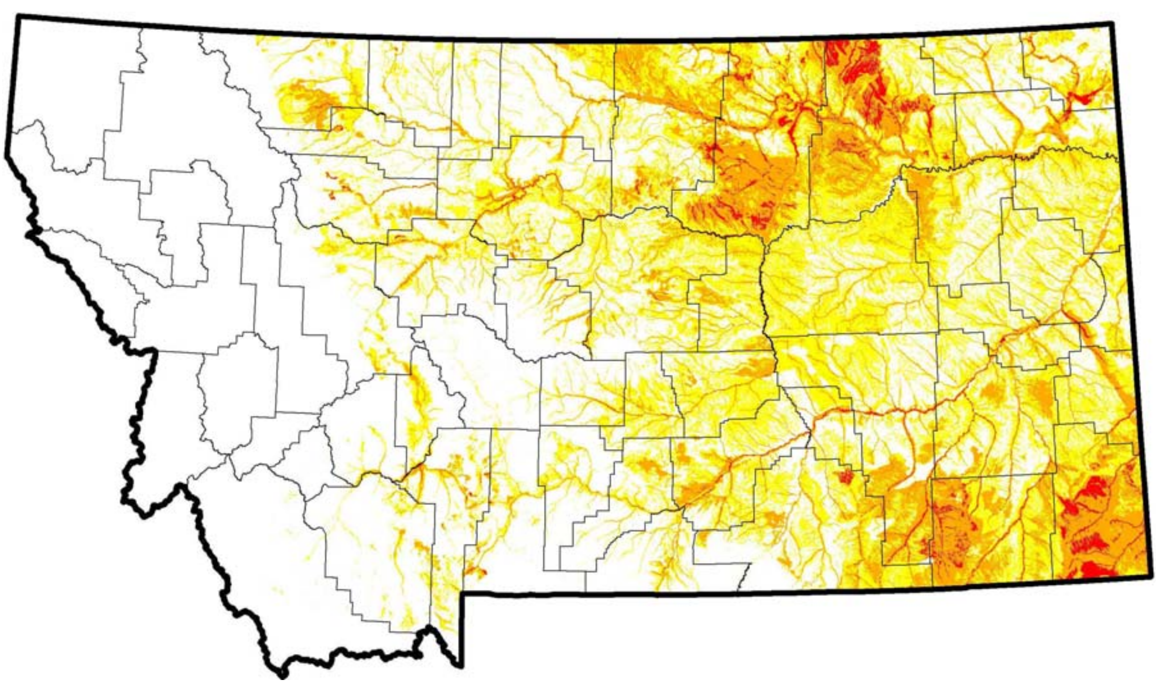


Figure D-14c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *R. pipiens*. Only current range is shown.



Snapping Turtle (*Chelydra serpentina*)

The state-wide continuous model indicates the known range of the species may be extended: (1) into a number of northern tributaries to the Yellowstone River below Billings; (2) the Clark's Fork of the Yellowstone River; and (3) upstream of current observations on the Little Bighorn River (Figure D-15a). Interestingly, two known sites of introduction outside of the native range of the species (portions of the Flathead and Gallatin Valleys) are predicted as having small areas of suitable habitat by the state-wide continuous model (Maxell et al. 2003). Characteristics of environmental variables of importance (Table 3.5) include average maximum July temperatures of greater than 29 °C (state-wide models only) and areas within 200 m of major streams. Solar radiation indices were the only variables that were essentially of no help in improving training gain on the model. GAP AVI was very poor (0.44), but in many cases areas predicted by GAP were only a few pixels away from observations (Table 3.6). The total area predicted by GAP and the low cutoff threshold models were within 8 km^2 of each other (Table 3.7). Although the range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to more appropriately represent the probable distribution of the species than the simple GAP model of streams within suitable habitat cover types (Hart et al. 1998) in most areas. However, in a couple areas it appears that soil map units produce an artifact that creates blocks of predicted habitat too far from major streams and these areas inappropriately represent the species' distribution at the local-landscape scale. The species is one of the most poorly documented vertebrates in the state (Table 3.3, Figure D-15b) and we encourage surveys in areas indicated as suitable habitat by these models which currently lack observations.

Figure D-15a. Continuous state-wide habitat suitability index for *C. serpentina*. Only observations within their native range were used to train the model.

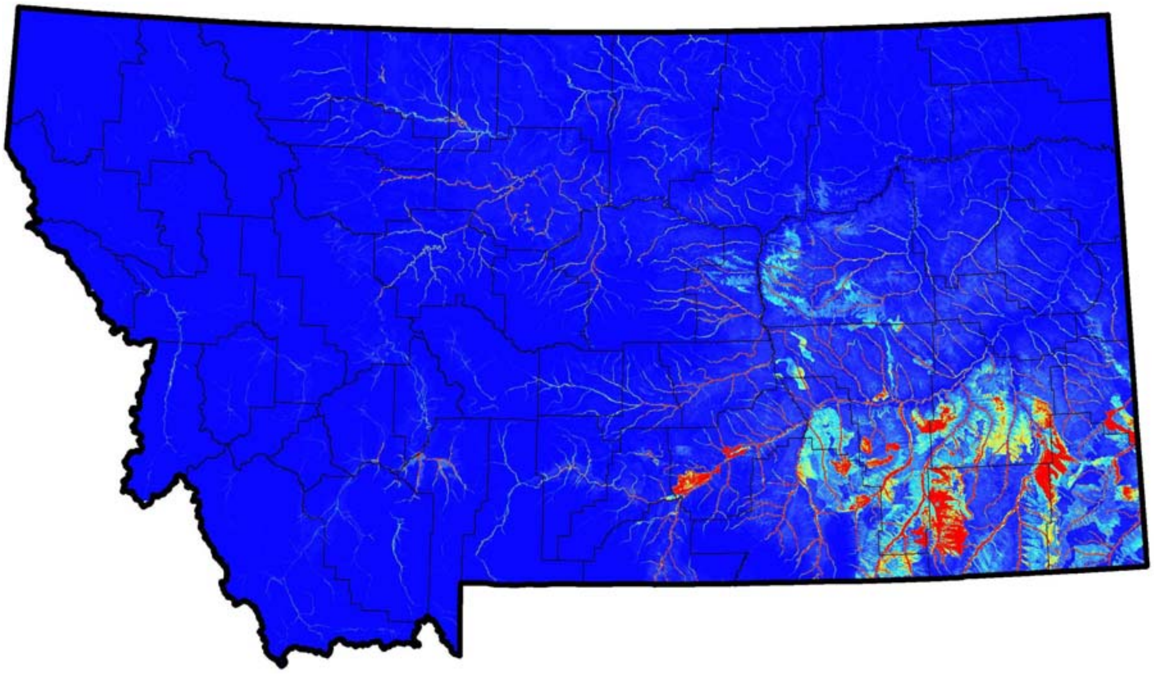
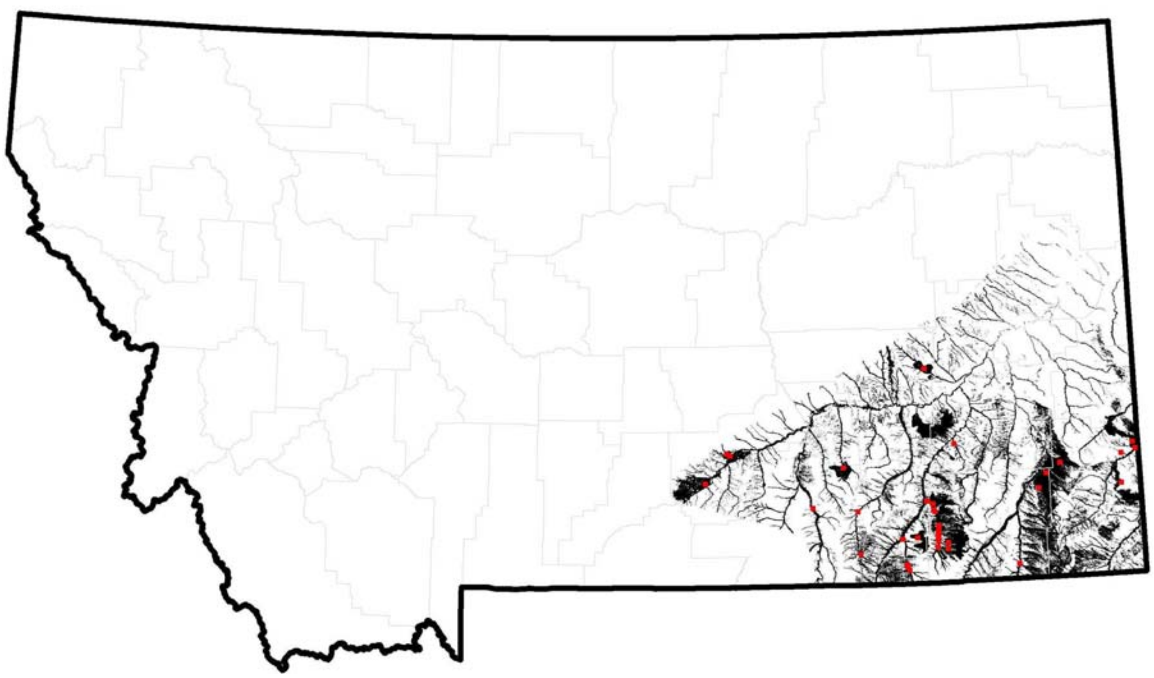


Figure D-15b. Predicted suitable habitat and training observations for *C. serpentina*. Only native range is shown.



Painted Turtle (*Chrysemys picta*)

The state-wide continuous model indicates suitable habitat in several areas across Montana, which currently lack observation records (Figures D-16a-b). These include: (1) the Madison River Valley near Ennis; (2) the Paradise Valley near Livingston; (3) a large area of farmland and Blackfeet Tribal land extending north of Great Falls to the Canadian border; (4) a large region between the Yellowstone and Missouri Rivers east of the Big Snowy Mountains; and (5) lower elevation portions of the Crow Indian Reservation. The state-wide models also indicate stringers of low to moderately suitable habitat that extend up the Clark Fork River to Butte and over Deer Lodge Pass to the Big Hole River that likely provide limited contemporary connectivity over the Continental Divide. Because evidence indicates that the species recolonized the Great Plains after a period of intense aridity that reached its maximum 14,000 years ago (Starkey et al. 2003), it also seems likely that this same route is the most probable path by which the species colonized the Pacific Northwest (see overview of range in Stebbins 2003). Characteristics of environmental variables of importance (Table 3.5) include wetland and deciduous forest cover types, elevations less than 1,400 m, slopes less than 20 degrees, and areas in close proximity to major streams. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.929 and 0.97, respectively) indicated good model fit and the AVI was much better than the GAP model (0.22) (Table 3.6). The GAP model was probably unrealistic in that it simply turned on streams within appropriate habitat types (Hart et al. 1998). As a result the low cutoff threshold model predicted a 370 percent (121,305 km^2) larger area than GAP (Table 3.7). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, and low Kappa values for both the low cutoff threshold and GAP models (Table 3.8). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section in western Montana and most areas of eastern Montana (Figure D-16c). However, some areas east of the Continental Divide that lack surveys are probably under predicted by the low cutoff threshold.

Figure D-16a. Continuous state-wide habitat suitability index for *C. picta*.

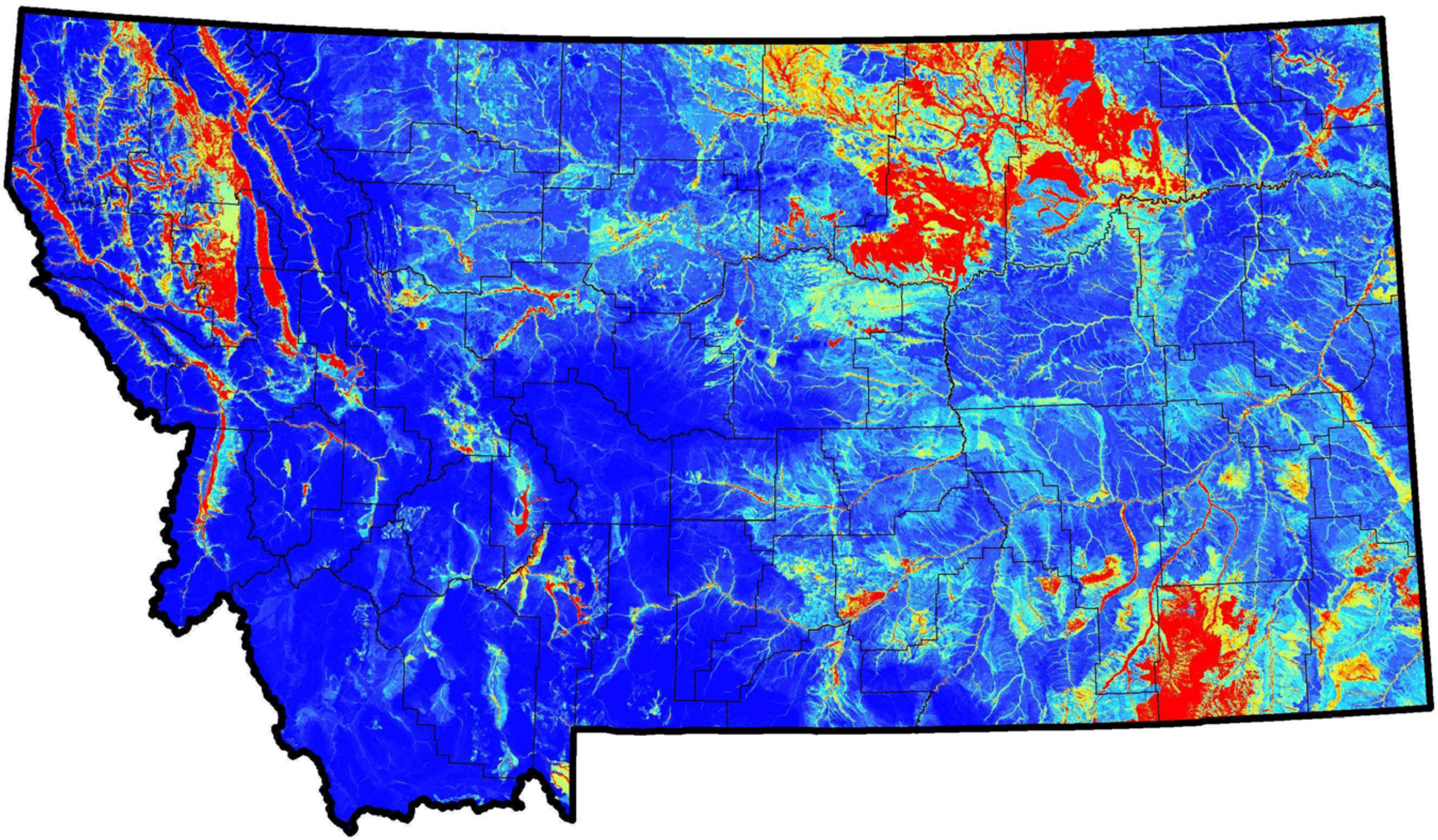


Figure D-16b. Predicted suitable habitat and training (squares) and test (stars) observations for *C. picta*.

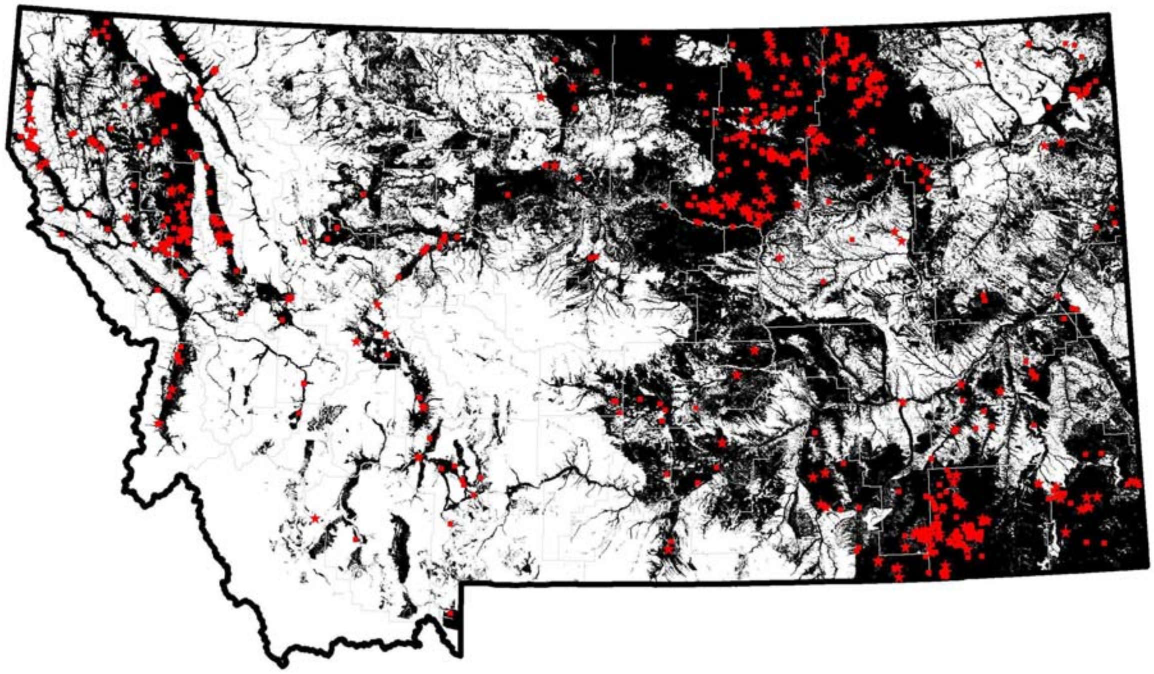
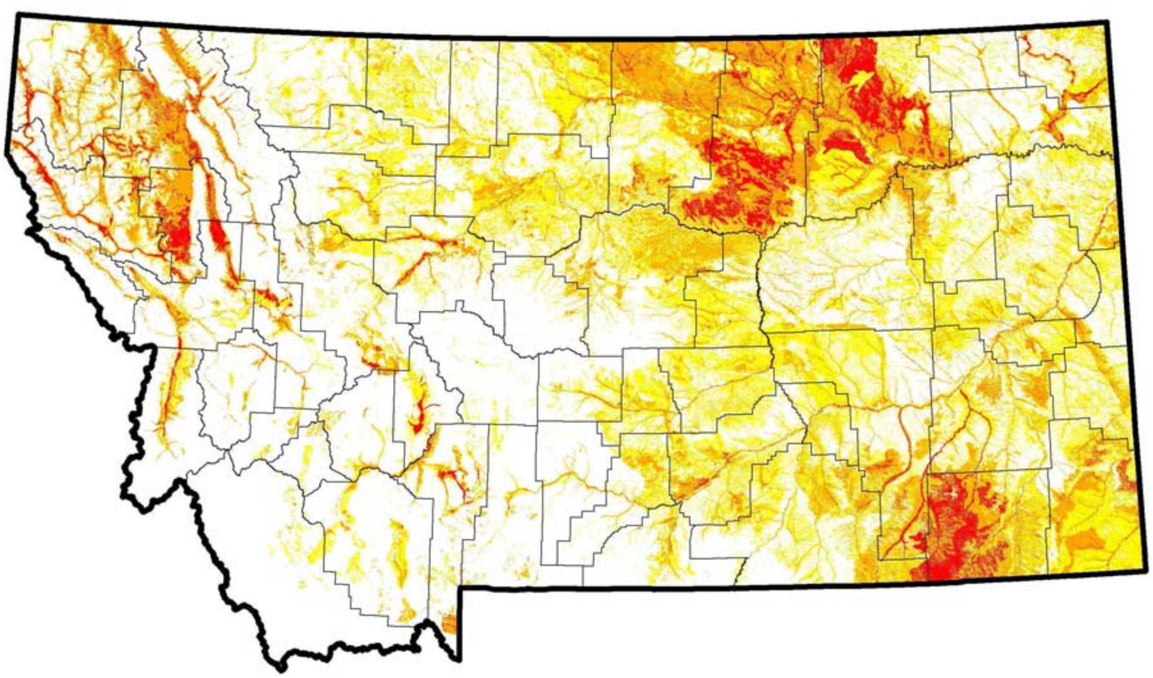


Figure D-16c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *C. picta*.



Spiny Softshell (*Apalone spinifera*)

The state-wide continuous model probably represents the likely historic range of the species prior to the establishment of Tiber Reservoir and Fort Peck Reservoir (Figure D-17a). That is, it seems likely that the species was previously distributed further upstream in the Marias River above Tiber Reservoir, portions of the Milk River, and in the area of the Missouri River now covered by Fort Peck Lake when these rivers were free-flowing. Characteristics of environmental variables of importance (Table 3.5) include areas within 200 m of major streams, elevations below 1,100 m, and wetland and water cover types. GAP AVI was very poor (0.28) because the major river habitats predicted by GAP failed to encompass small backwater habitats and larger tributaries the species uses (Table 3.6, Hart et al. 1998). As a result, the low cutoff threshold model predicted a 1,746 percent (7,926 km^2) larger area than GAP (Table 3.7). Although range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to appropriately represent the distribution of the species and seem to be better than rule based GAP model of major streams (Figure D-17b).

Figure D-17a. Continuous state-wide habitat suitability index for *A. spinifera*.

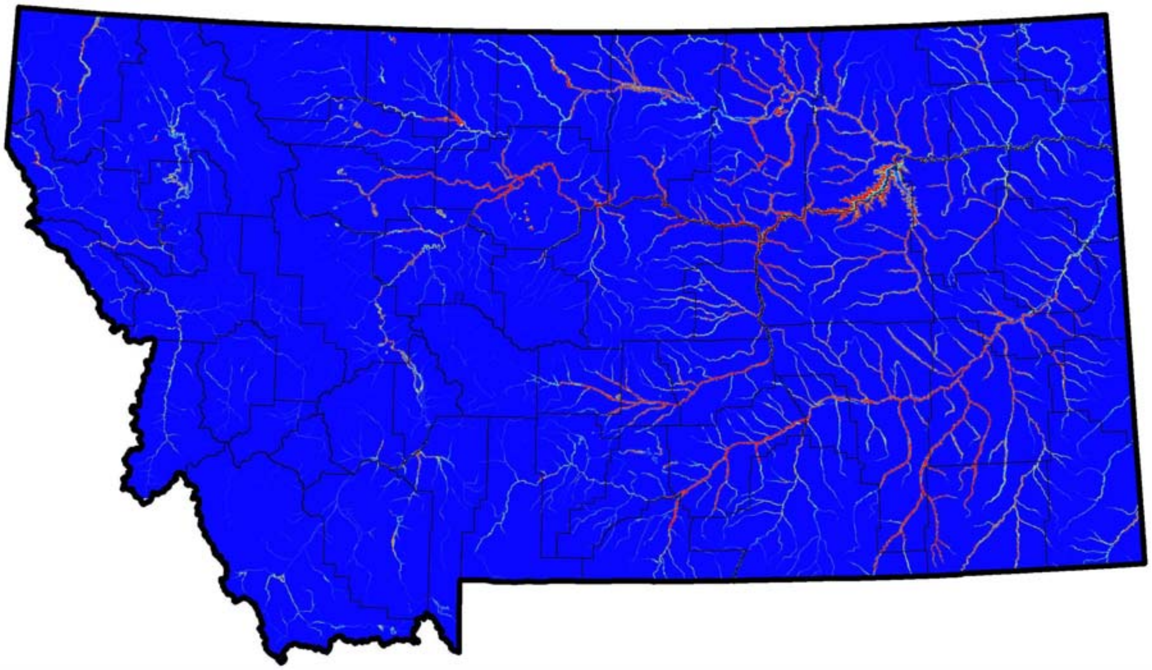
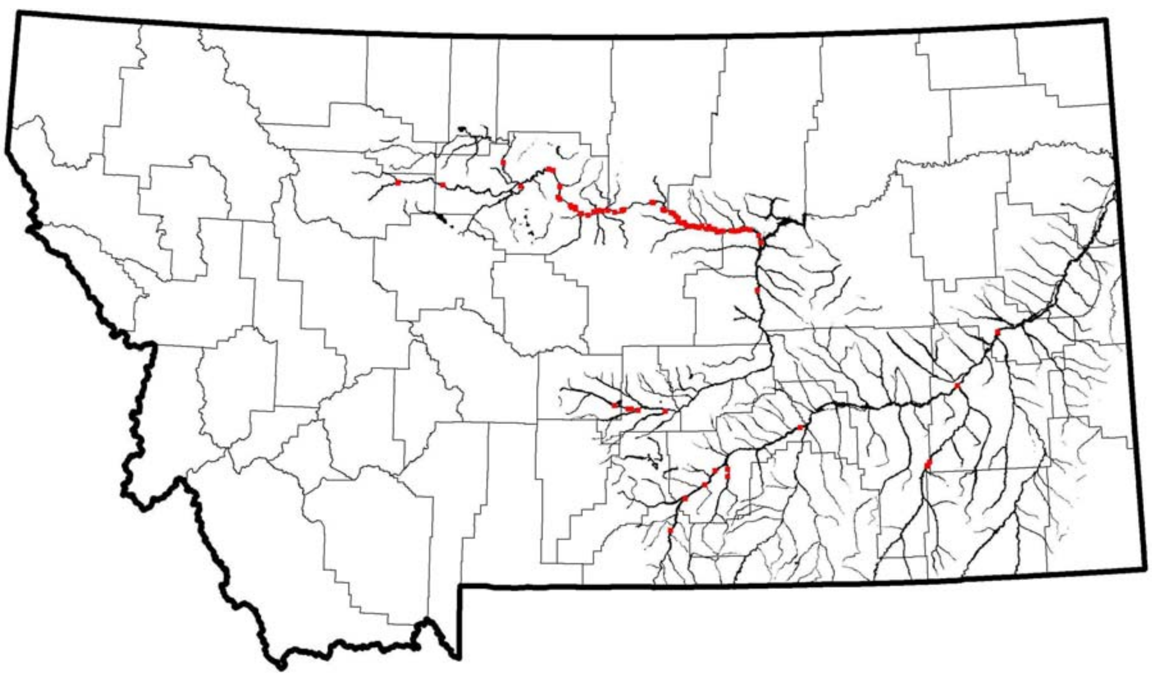


Figure D-17b. Predicted suitable habitat and training observations for *A. spinifera*.



Northern Alligator Lizard (*Elgaria coerulea*)

The state-wide continuous model indicates the species' known range may be extended into previously undocumented areas in: (1) the Sapphire Mountains; (2) the Garnet Mountains; (3) the Bitterroot Mountains between Missoula and Lookout Pass; (4) lower portions of the Swan River drainage; and (5) lower portions of the South Fork, Middle Fork, and North Fork of the Flathead River drainages (Figure D-18a). Furthermore, a large area centered around the Fisher River drainage currently lacks observations, but is predicted to have suitable habitat (Figure D-18b). Soil temperature and moisture regime (cool and moist soils) and average minimum January temperature (greater than -9 to -12 °C) were ranked as important variables in state-wide models, but were ranked of low importance at the scale of the species' range. Elevation was ranked as more important within the species' range model (less than 1,200 m) than it was in the state-wide model. Southeast, south, and southwest aspects were favored and high terrain ruggedness was ranked as the most and second most important variable at the state-wide and range-wide scales, respectively, apparently as a result of being indicative of the rocky slopes the species is associated with (Werner et al. 2004). Solar radiation indices did not improve training gain and may not be of use in modeling the species' distribution in their current form. GAP AVI was poor (0.67), but in many cases areas predicted by GAP were only a few pixels away from observations and the GAP model seemed to be a reasonable approximation of the species distribution (Table 3.6). Even though the range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to more appropriately represent a likely continuous range of habitat suitability than the deductively based GAP model of buffering both sides of forest edges at elevations below 2,100 m (Hart et al. 1998). The low cutoff threshold model predicted a 32 percent (5,379 km^2) smaller area than the GAP model (Table 3.7).

Figure D-18a. Continuous state-wide habitat suitability index for *E. coerulea*.

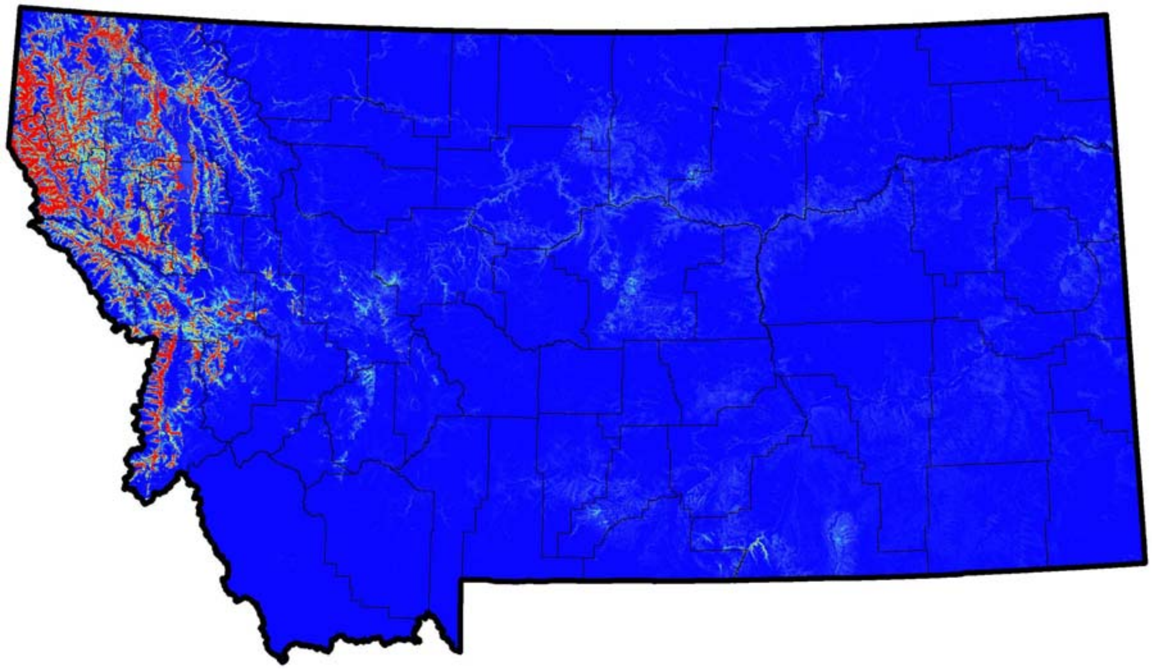
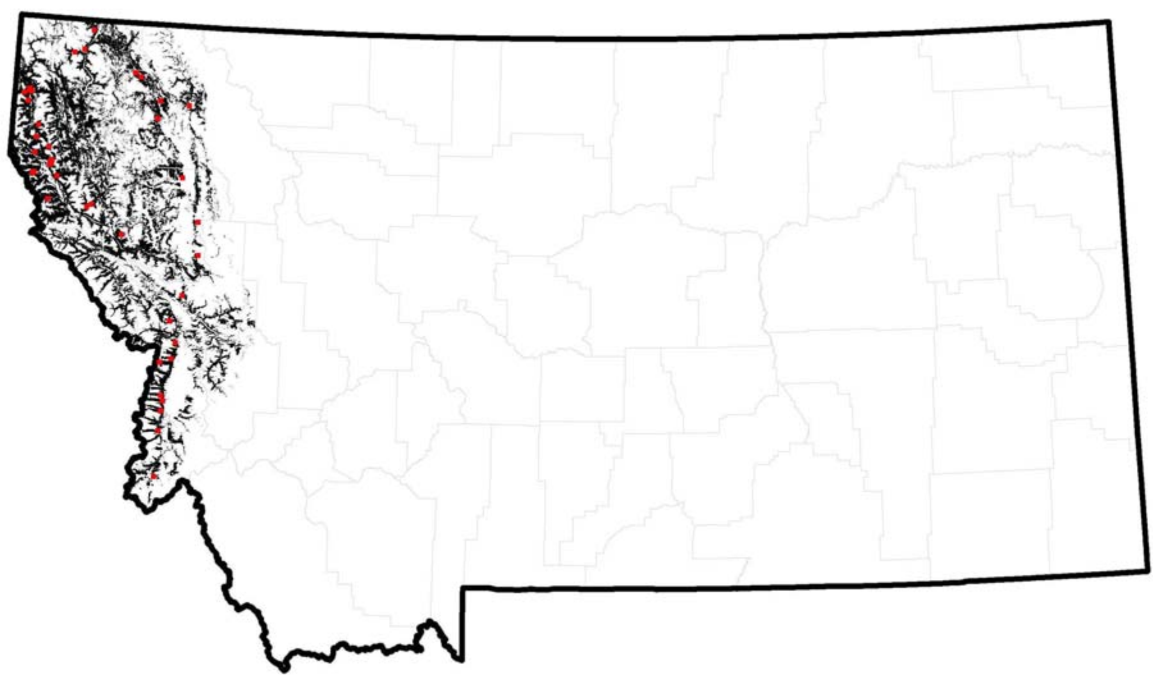


Figure D-18b. Predicted suitable habitat and training observations for *E. coerulea*.



Greater Short-horned Lizard (*Phrynosoma hernandesi*)

The species is one of the most poorly documented vertebrates in the state and museum and observation records for large regions are very old or too spatially imprecise to be used for modeling (Table 3.3, Maxell et al. 2003). The state-wide models may, therefore, be valuable in identifying local areas that should be targeted for surveys within regions where historic observations have poor spatial precision. For example, *P. hernandesi* has been collected around the Three Forks area on three occasions between 1888 and 1953, but all of the collection localities have poor spatial precision. The state-wide and range-wide continuous models constructed without these records predict suitable habitat in several areas around Three Forks which should be targeted for survey to see if populations are still present in the area (Figures D-19a-b). Furthermore, these models indicate suitable habitat in several areas along the Missouri River, supporting the hypothesis that populations of this and other primarily Great Plains distributed species reported in the Three Forks area colonized the area via the Missouri River corridor; an area which has now been greatly altered with hydrologic dams. Other areas which the models predict as possible extensions of known range include: (1) areas around the lower Big Hole River; (2) Beartrap Canyon; (3) some of the bluffs along the west side of the Smith River Canyon; (4) and bluffs around Tiber Reservoir and the upper Marias River (Figures D-19a-b). Characteristics of environmental variables of importance include barren, shrubland, and grassland cover types, low terrain ruggedness, cooler drier soils, average maximum July temperatures greater than 24 °C, and areas with high winter solar radiation index values. GAP AVI was 0.87, but the GAP predicted area was 56 percent (100,407 km^2) greater than the low cutoff threshold model because it simply turned on suitable cover types under 2,100 m elevation (Tables 3.6 and 3.7, Hart et al. 1998). Although range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to more appropriately represent suitable habitat than the deductively based GAP models in most areas. However, these models probably under predict the distribution of the species in a number of areas that lack observations.

Figure D-19a. Continuous state-wide habitat suitability index for *P. hernandesi*.

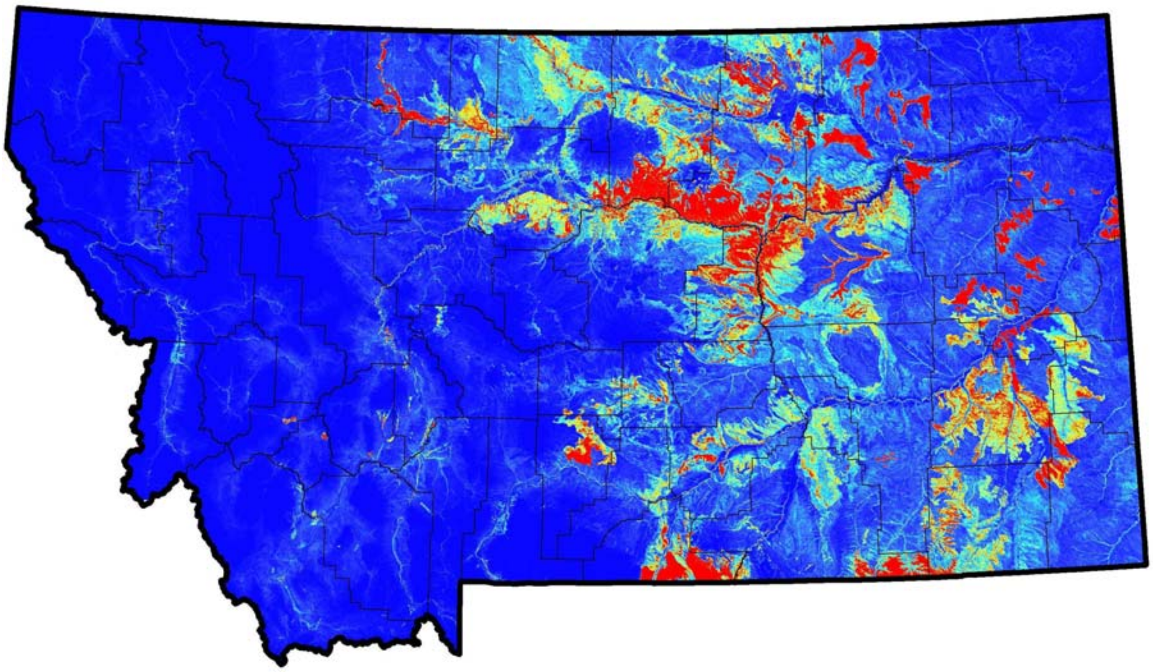
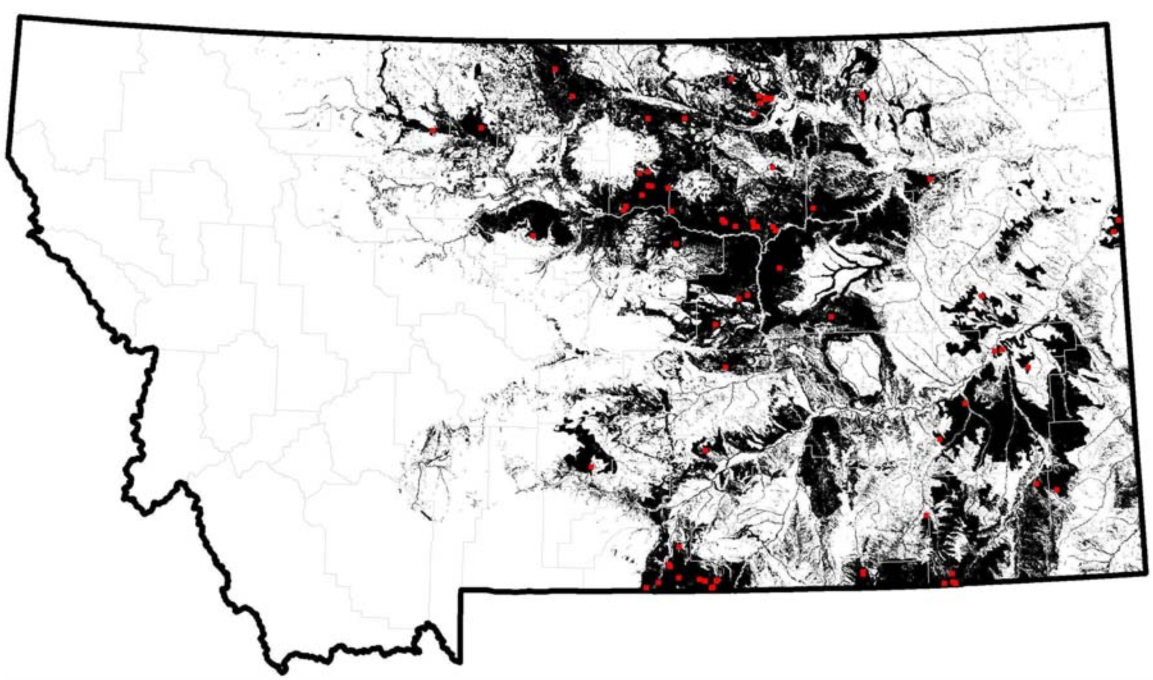


Figure D-19b. Predicted suitable habitat and training observations for *P. hernandesi*.



Common Sagebrush Lizard (*Sceloporus graciosus*)

The state-wide continuous model indicates the species' known range may be extended into previously undocumented areas including: (1) the area along the Yellowstone River between Columbus and Gardiner; (2) the western edge of the Bull Mountains; (3) lower portions of the Musselshell River; and (4) lower portions of the Marias River (Figure D-20a). Characteristics of environmental variables of importance (Table 3.5) include annual precipitation of less than 40 cm, warmer and drier soils, and elevations below 1,750 m. GAP AVI was 0.94, but the GAP predicted area was 75 percent (92,958 km^2) greater than the low cutoff threshold model because it simply turned on suitable cover types under 2,250 m elevation (Tables 3.6 and 3.7, Hart et al. 1998). Although range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to more appropriately represent suitable habitat at regional and local-landscape scales than the deductively based GAP models in areas that have even a small number of observations. In areas currently lacking observations, it seems likely that the continuous models under predict the distribution of the species. This highlights the importance of conducting systematic surveys for the species (Figures D-20a-b).

Figure D-20a. Continuous state-wide habitat suitability index for *S. graciosus*.

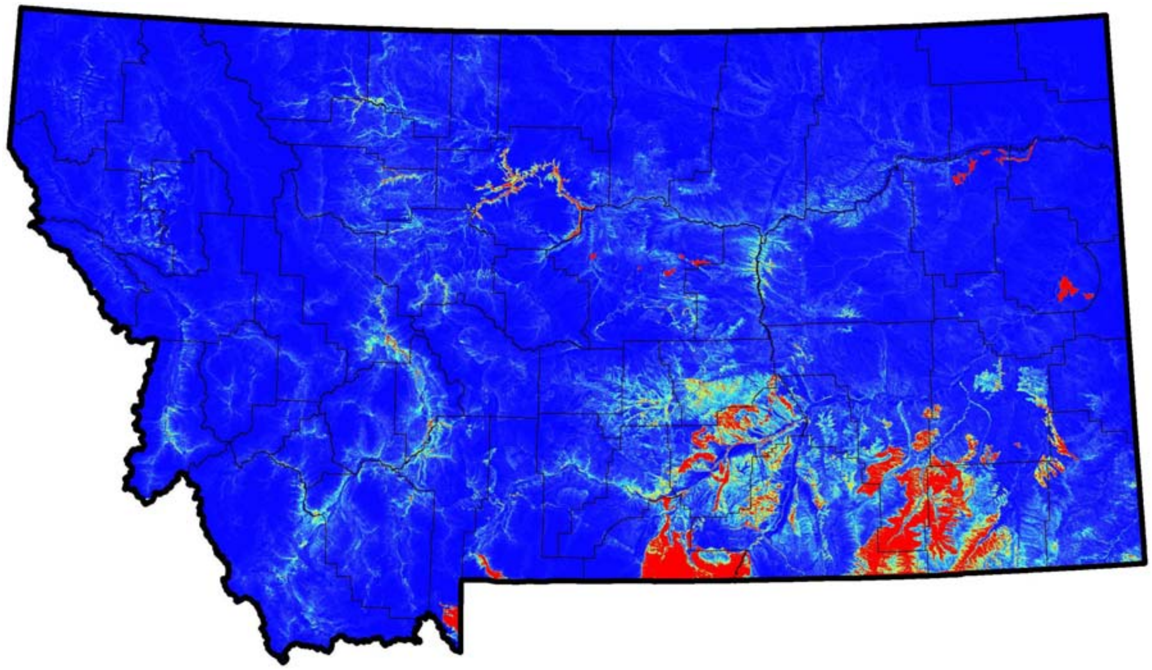
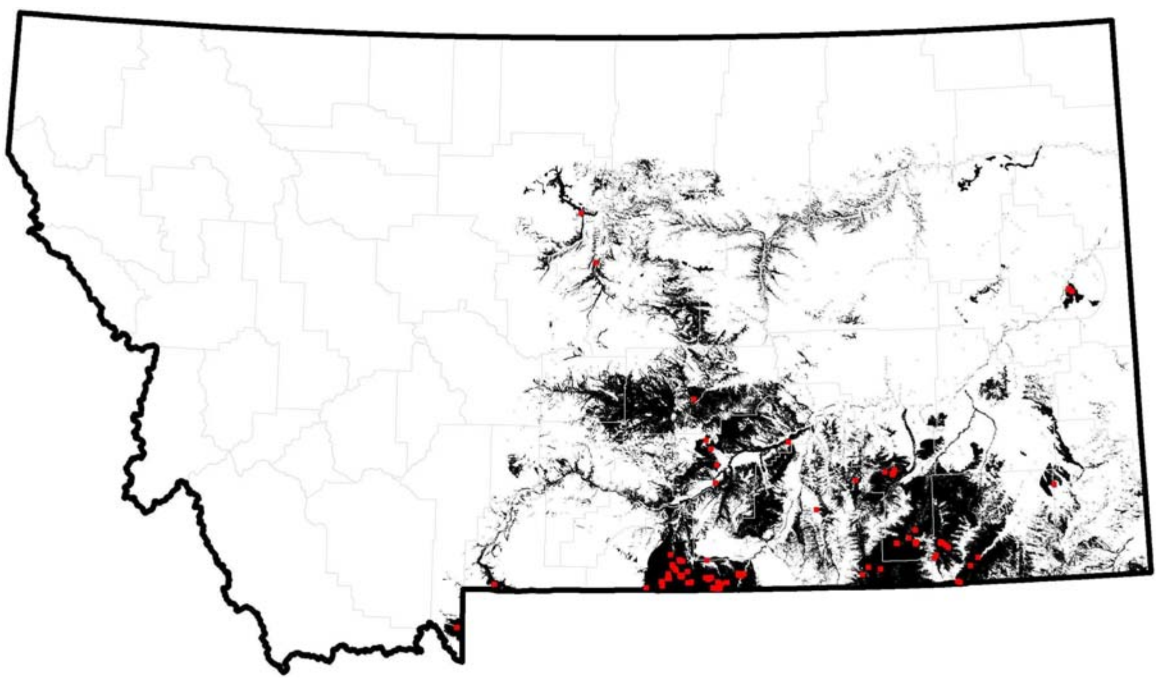


Figure D-20b. Predicted suitable habitat and training observations for *S. graciosus*.



Western Skink (*Eumeces skiltonianus*)

The species is one of the most poorly documented vertebrates in the state and a number of observation records have spatial precisions that are too poor to be used for modeling (Table 3.3, Maxell et al. 2003). Models may, therefore, be valuable in identifying local areas that should be targeted for survey with pitfall traps (Ortega and Pearson 2001). The state-wide continuous model indicates the species' known range may be extended into previously undocumented areas including: (1) lower portions of the Blackfoot River drainage; (2) portions of the upper Clark Fork between Missoula and Garrison; (3) western portions of the Flathead Indian Reservation; and (4) potentially lower portions of the South, Middle, and North Forks of the Flathead River (Figures D-21a-b). Minimum average January temperature (greater than -15 to -16 °C) was ranked as the most important variable at a state-wide scale, but was significantly less important within the species' range. Other characteristics of environmental variables of importance (Table 3.5) include a strong preference for southern aspects, habitat cover types that are grassland, shrubland, or transitional from these to open forest, and elevations below 2,000 m. GAP AVI was 0.79 and the GAP predicted area was 36 percent greater (4,949 km^2) than the low cutoff threshold model because it simply turned on suitable cover types below 2,100 m elevation (Tables 3.6 and 3.7, Hart et al. 1998). Although the continuous and low cutoff threshold models could not be evaluated with test data, they seem to more appropriately represent suitable habitat than the deductively based GAP models in areas that have even a small number of observations (Figure D-21b). Indeed, observations with poor spatial precision not used for training the models almost always fell within areas predicted as suitable habitat by the low cutoff threshold model. In areas currently lacking observations, it is either possible that the range-wide continuous models under predict the distribution of the species or that they are truly rare or absent.

Figure D-21a. Continuous state-wide habitat suitability index for *E. skiltonianus*.

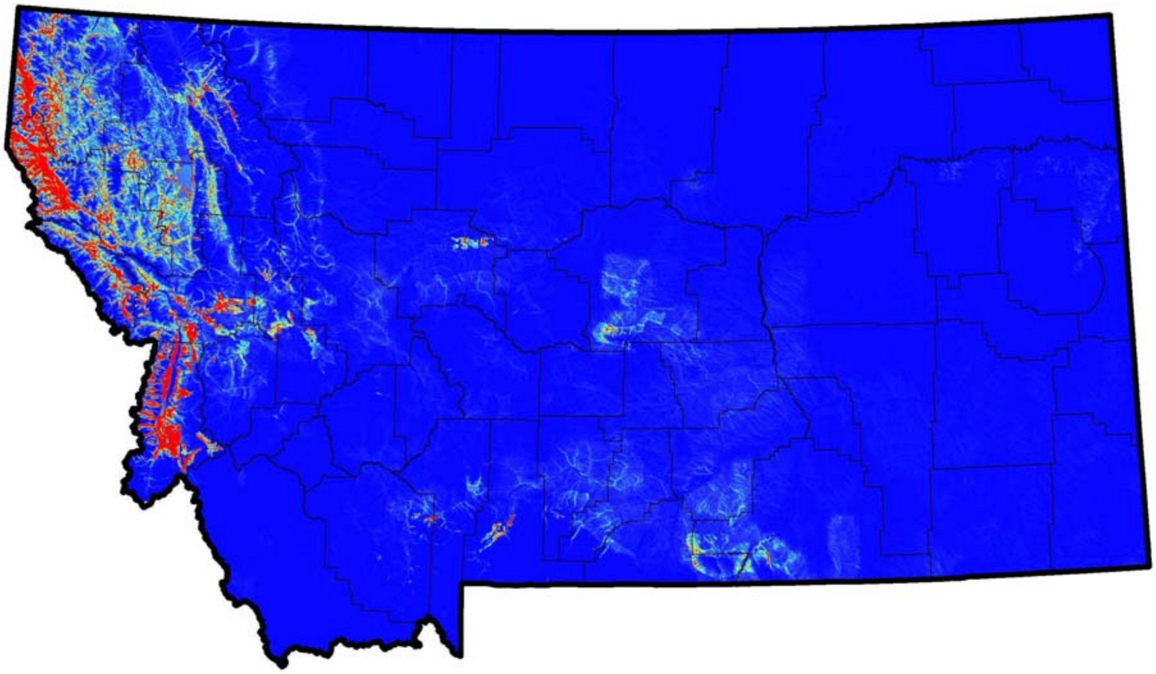
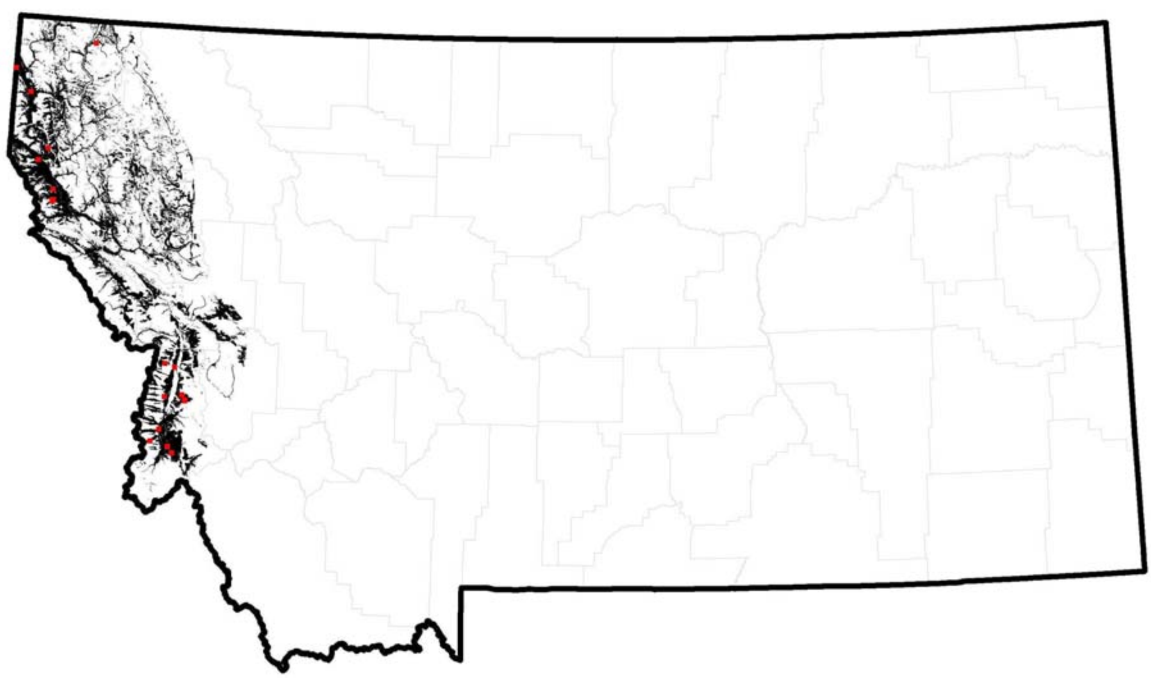


Figure D-21b. Predicted suitable habitat and training observations for *E. skiltonianus*.



Rubber Boa (*Charina bottae*)

The state-wide continuous model indicates the species' known range may be extended into previously undocumented areas including: (1) the eastern edge of the Beartooth Plateau; (2) the Pryor Mountains; (3) the Big Horn Mountains on the Crow Indian Reservation; (4) the Wolf Mountains on the Northern Cheyenne Indian Reservation; (5) the eastern edge of the Little Belt Mountains and the Big Snowy Mountains; (6) the Judith and Moccasin Mountains; portions of the Beaverhead and Centennial Mountains; and (7) portions of the East Front of the Rocky Mountains from the Canadian border southward to Wolf Creek (Figures D-22a-b). Indeed, the state-wide model correctly predicted a recent 185 km range extension for the species that was not used in the modeling effort; an observation on a section of state land near the headwaters of Rosebud Creek in southeastern Bighorn County (Lisa Wilson and David Stagliano, Montana Natural Heritage Program, pers. comm.). Characteristics of environmental variables of importance (Table 3.5) include terrain with at least some level of ruggedness, areas usually within 1,000 m of major streams, elevations less than 2,000 m (usually less than 1,500 m), and evergreen forest, deciduous forest, or woody wetland habitat cover types. Solar radiation indices did not improve training gain and may not be of use in modeling the species' distribution in their current form. GAP AVI was 0.9 and the GAP predicted area was 61 percent (62,009 km^2) greater than the low cutoff model because it turned on suitable cover types within a 500-m distance of all streams at elevations below 2,850 m (Tables 3.6 and 3.7, Hart et al. 1998). Although continuous and low cutoff threshold models could not be evaluated with test data, they seem to appropriately represent suitable habitat in areas that have observation data. Unfortunately, they probably under predict the distribution of the species in areas that lack data and because the species is very cryptic, many areas lack observation data (Werner et al. 2004). Thus, within regions that currently have few observations, it would probably be best to use deductive models in tandem with these continuous inductive models when considering this species in management plans.

Figure D-22a. Continuous state-wide habitat suitability index for *C. bottae*.

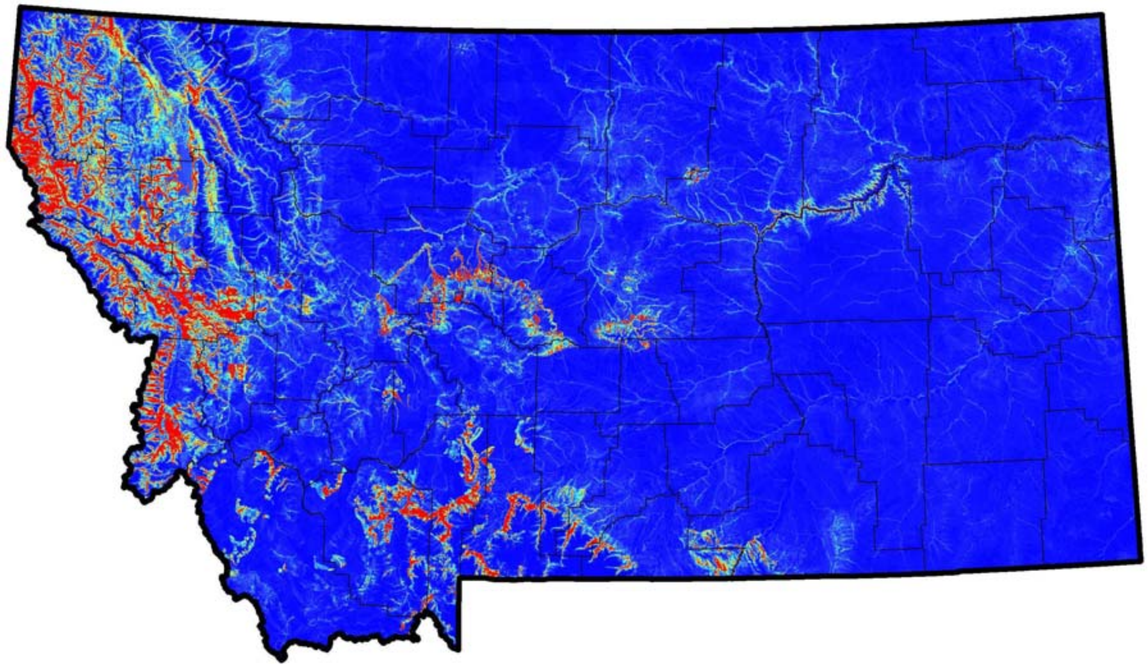
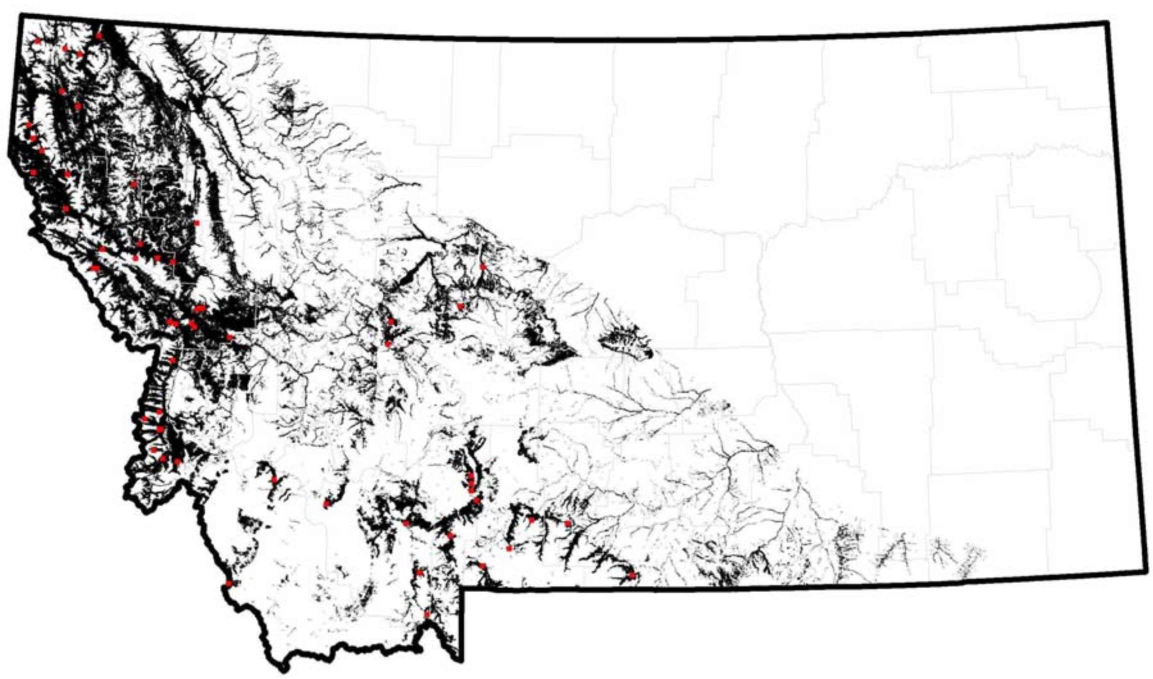


Figure D-22b. Predicted suitable habitat and training observations for *C. bottae*.



Eastern Racer (*Coluber constrictor*)

The state-wide continuous models predict separation between eastern and western populations of the species based on availability of suitable habitat (Figures D-23a-c). However, there are faint stringers of habitat of low predicted suitability that extend up the Clark Fork River to Butte and over Deer Lodge Pass to the Big Hole River that may provide limited connectivity (Figure D-23a). Surveys and collection of specimens in the Melrose, Dillon, and upper Clark Fork region would help improve predictive models and the understanding of potential levels of gene flow between populations east (currently speculated to be *C. constrictor flaviventris*) and west (currently speculated to be *C. constrictor mormon*) of the Continental Divide (Corn and Bury 1986, Maxell et al. 2003). Characteristics of environmental variables of importance (Table 3.5) include average maximum July temperatures greater than 27 °C, elevations less than 1,600 m, and annual precipitation less than 50 cm. Ruggedness, aspect, and the solar radiation indices did not improve the models and the inclusion of these variables actually hurt the model as indicated by the jackknife plots of test gain. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.858 and 0.91, respectively) indicated good model fit and the AVI was the same for the GAP model (0.22), which predicted a 39 percent (100,832 km^2) larger area than the low cutoff threshold model (Tables 3.6 and 3.7). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance. However, the average deviance of test occurrences was high relative to other species and a number of grassland habitats that would likely support the species in various portions of its range are not recognized as suitable habitat by the model. Where they overlap, the low, moderate, and optimal habitat suitability class thresholds appear to be superior to GAP in that they more appropriately reflect the relative suitability of habitat. However, the low cutoff threshold seems to under predict the overall area of suitable habitat in some portions of eastern and western Montana, usually in regions where there are no, or relatively few, observations (Figure D-23b). Until more observations are available in these grassland habitats, it would probably be best to use deductively based models in tandem with the continuous model.

Figure D-23a. Continuous state-wide habitat suitability index for *C. constrictor*.

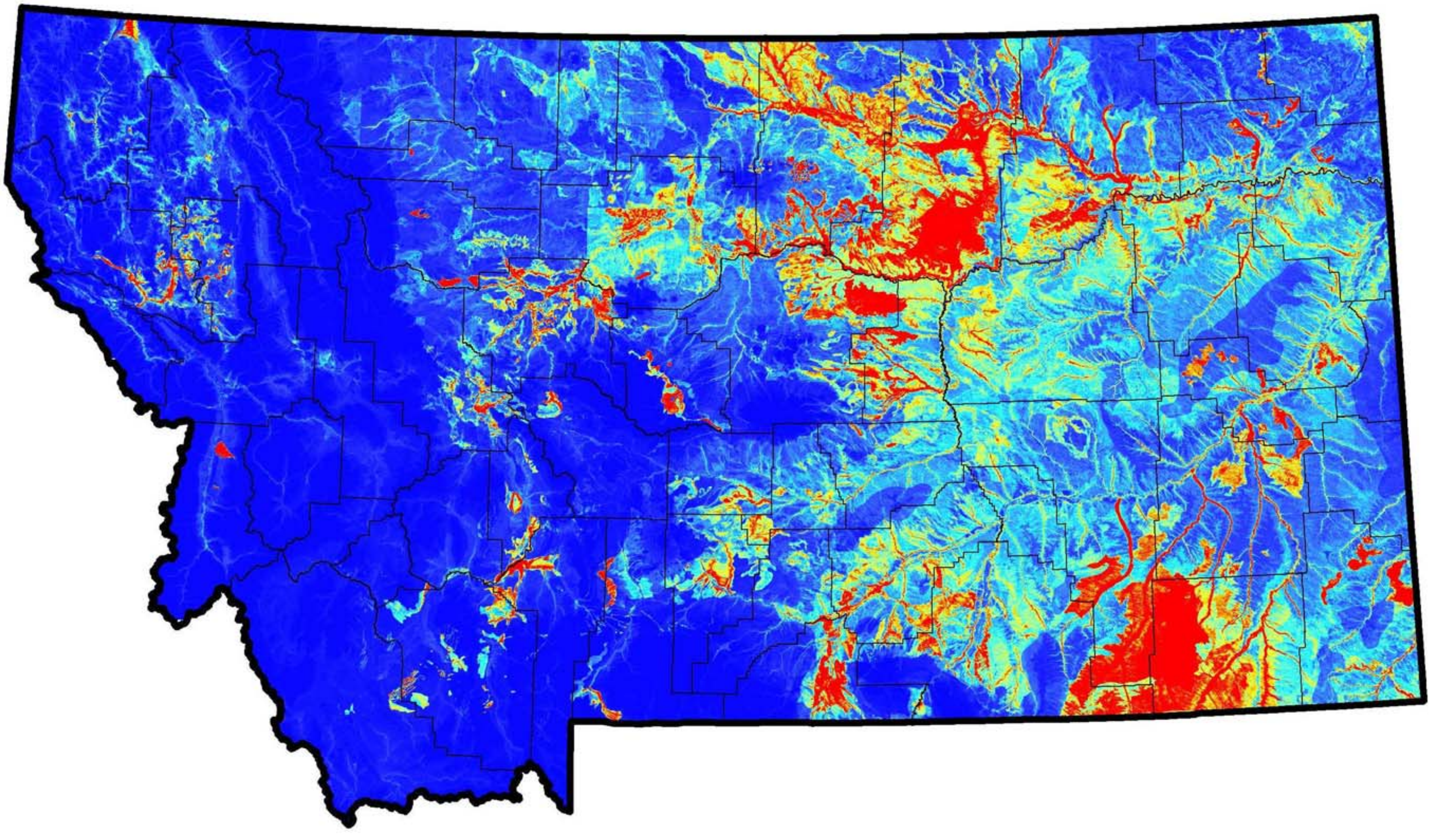


Figure D-23b. Predicted suitable habitat and training (squares) and test (stars) observations for *C. constrictor*.

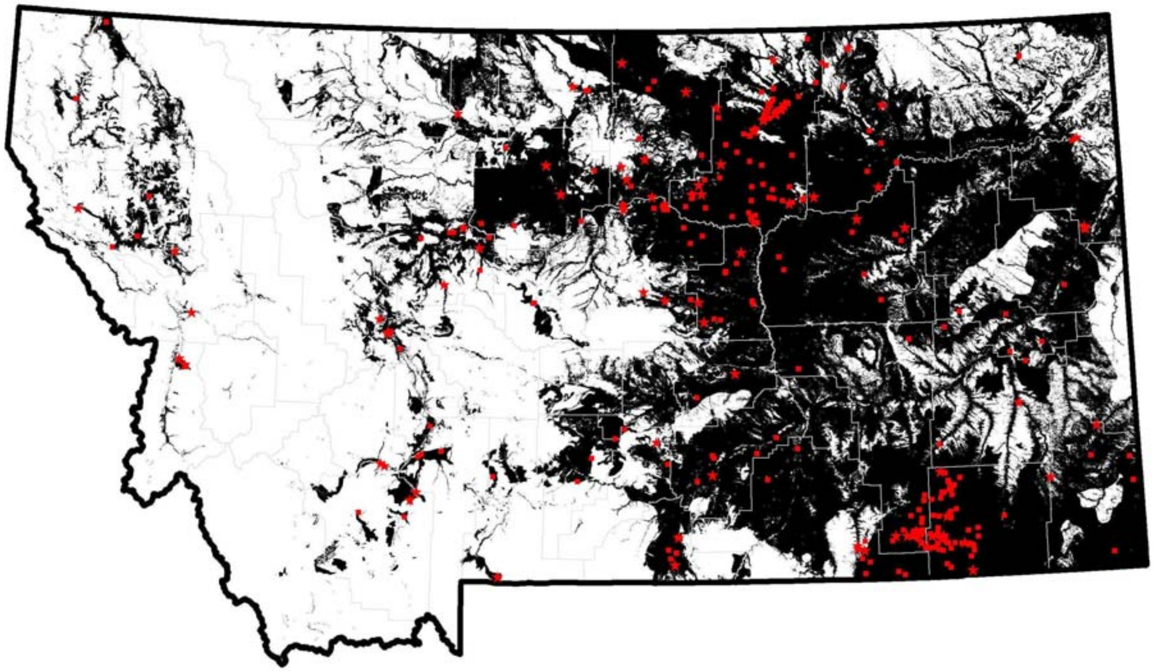
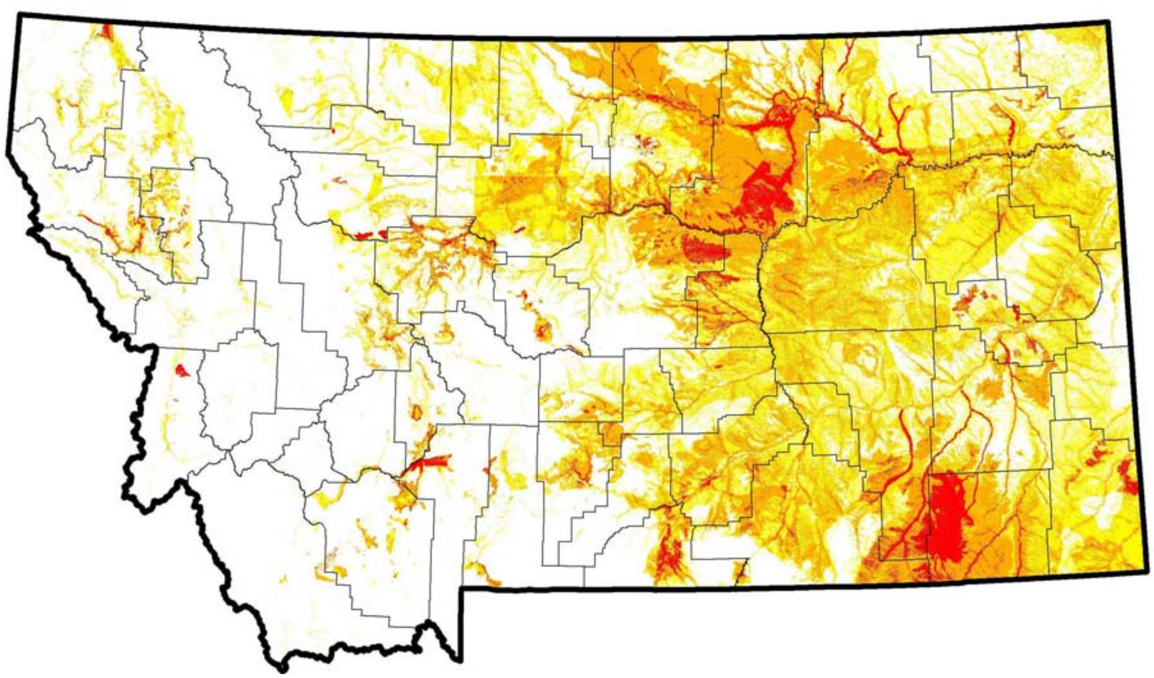


Figure D-23c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *C. constrictor*.



Western Hog-nosed Snake (*Heterodon nasicus*)

The state-wide continuous model indicates the known range of the species may be extended: (1) upstream on the Clark's Fork of the Yellowstone River; (2) upstream on Big Otter Creek from the Missouri River to Stanford; and (3) up the Musselshell River to Ryegate (Figures D-24a-b). Characteristics of environmental variables of importance (Table 3.5) include average maximum July temperatures greater than 29 °C, soils saturated with ground water or drier soils nearby, areas typically within 1,500 m of major streams, and elevations less than 1,500 m. GAP AVI was 0.76 and the GAP predicted area was 63 percent (125,245 km^2) greater than the low cutoff threshold model because it simply turned on suitable cover types at elevations below 1,650 m (Tables 3.6 and 3.7, Hart et al. 1998). Although range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to more appropriately represent suitable habitat than the simple deductively based GAP models in most areas. However, they likely under represent the species distribution in areas that lack observations. In some areas, soil map units produce what appears to be an unrealistically sharp delineation between suitable and non-suitable habitat. However, it is possible that this could adequately represent the species' distribution since they are highly adapted for digging in sandy or other friable soils (Platt 1969, 1983). As with other species, this highlights the need for several state-wide layers summarizing a variety of soil characteristics. More observations are also needed for this species and until data is available in regions that currently have few records, it would probably be best to rely on a combination of deductive and continuous model outputs when considering this species in management plans. Riparian areas with friable soils (Platt 1969, 1983) in the range of the species should be surveyed prior to undertaking management actions that would alter habitat.

Figure D-24a. Continuous state-wide habitat suitability index for *H. nasicus*.

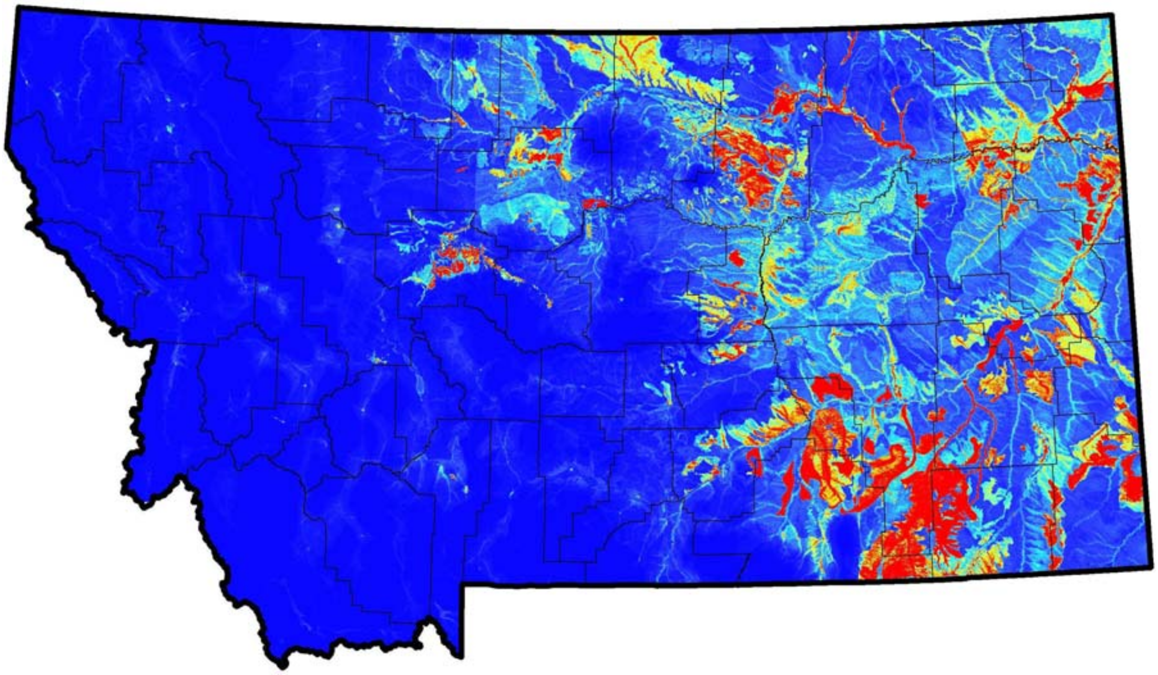
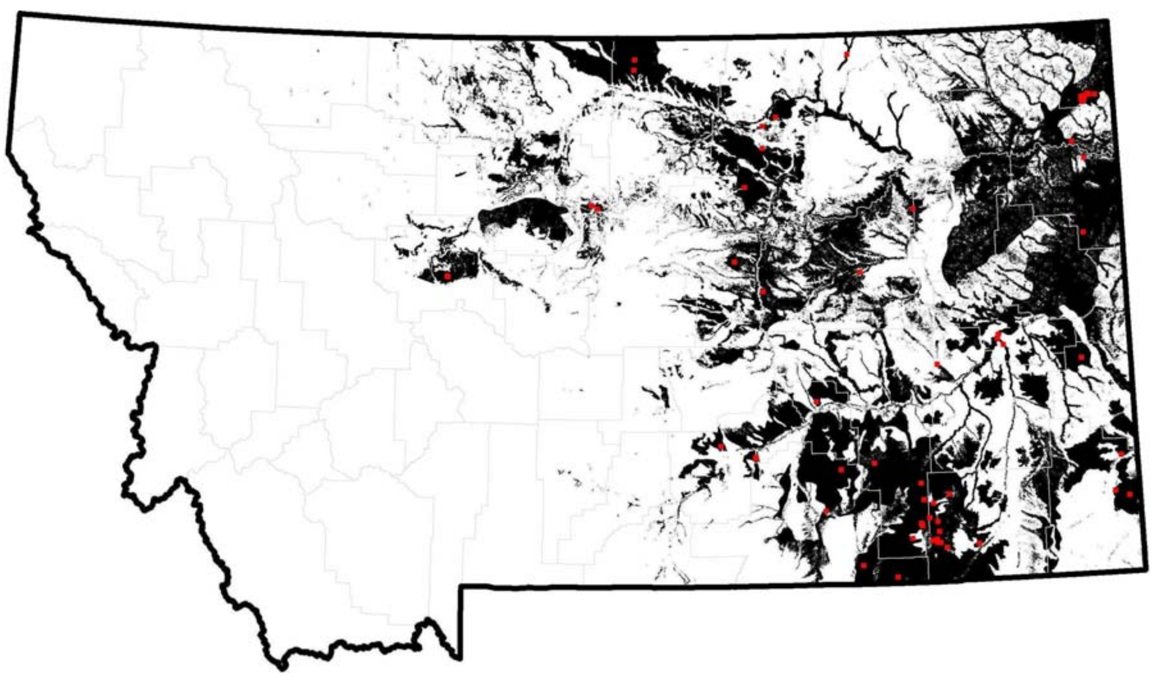


Figure D-24b. Predicted suitable habitat and training observations for *H. nasicus*.



Smooth Greensnake (*Opheodrys vernalis*)

The species is one of the most poorly documented vertebrates in the state and several observation records have spatial precisions too poor to be used for modeling (Table 3.3, Maxell et al. 2003). The state-wide continuous model indicates the species' known distribution may be extended: (1) up the lower portion of the Yellowstone River from the state line; (2) along the lower portion of the Missouri River below Fort Peck Reservoir; (3) in the vicinity of the Poplar River; and (4) possibly the lower portion of the Milk River in Valley and Phillips Counties (Figures D-25a-b). Surveys should be encouraged in each of these areas. Characteristics of environmental variables of importance (Table 3.5) include elevations less than 700 m, average minimum January temperatures less than -17 °C (important for the state-wide, but not the range-wide model), and soils saturated with groundwater. GAP AVI was poor (0.22) because the species has been detected on a number of occasions outside the 500-m hydrography buffer applied within suitable habitats (Table 3.6, Hart et al. 1998). The low cutoff threshold model predicted a 155 percent (3,679 km^2) larger area of suitable habitat (Table 3.7). Although range-wide continuous and low cutoff threshold models could not be evaluated with test data, they seem to be very appropriate and probably do a much better job of representing suitable habitat than the simple deductively based GAP model.

Figure D-25a. Continuous state-wide habitat suitability index for *O. vernalis*.

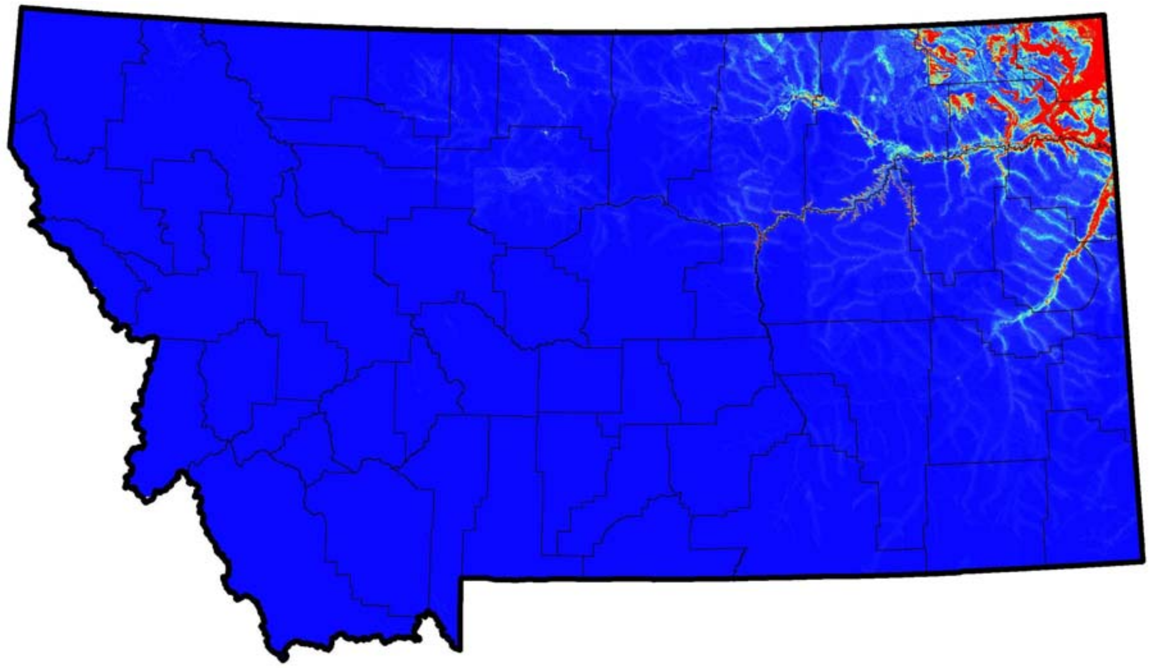
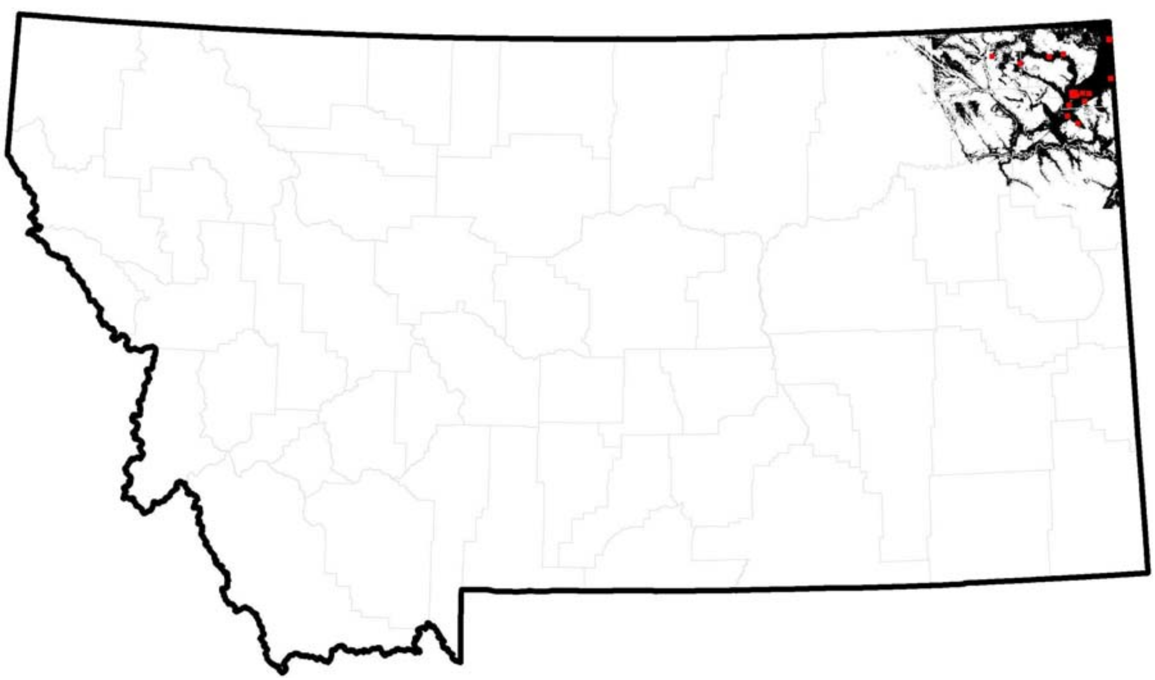


Figure D-25b. Predicted suitable habitat and training observations for *O. vernalis*.



Milksnake (*Lampropeltis triangulum*)

The species is one of the most poorly documented vertebrates in the state and a large percentage of observation and museum records have spatial precisions too large to be used for modeling (Table 3.3, Maxell et al. 2003). The state-wide continuous model indicates the species' known distribution may be extended: (1) southeastward toward the Ekalaka Hills; (2) upstream on the Yellowstone River potentially into the Paradise Valley; (3) southeastern Fergus County; and (4) upstream on the Missouri River potentially as far as the area around Three Forks, an area where several questionable records exist (Figures D-26a-b, Maxell et al. 2003). Characteristics of environmental variables of importance (Table 3.5) include average maximum July temperatures greater than 29 °C, and barren and shrubland cover types; other environmental variable response curves are uninterpretable. GAP AVI was poor (0.68) because despite predicting a very high percentage of areas below 1,950 m within the species potential range, many observations still fell within small patches predicted to be unsuitable (Table 3.6, Figure D-26b). Low cutoff threshold models predicted a 73 percent (95,824 km^2) smaller area of suitable habitat than GAP (Table 3.7). Although range-wide continuous and low cutoff threshold models could not be evaluated with test data, they appear to more appropriately represent suitable habitat in areas that have observations than the simple deductively based GAP models. Unfortunately, they probably under predict the distribution of the species in areas that lack data and because the species is very cryptic, many areas lack observation data (Werner et al. 2004). Thus, until more observation data is available, it would probably be best to use deductive and continuous model outputs in tandem when considering this species in management plans within regions that currently have few observations.

Figure D-26a. Continuous state-wide habitat suitability index for *L. triangulum*.

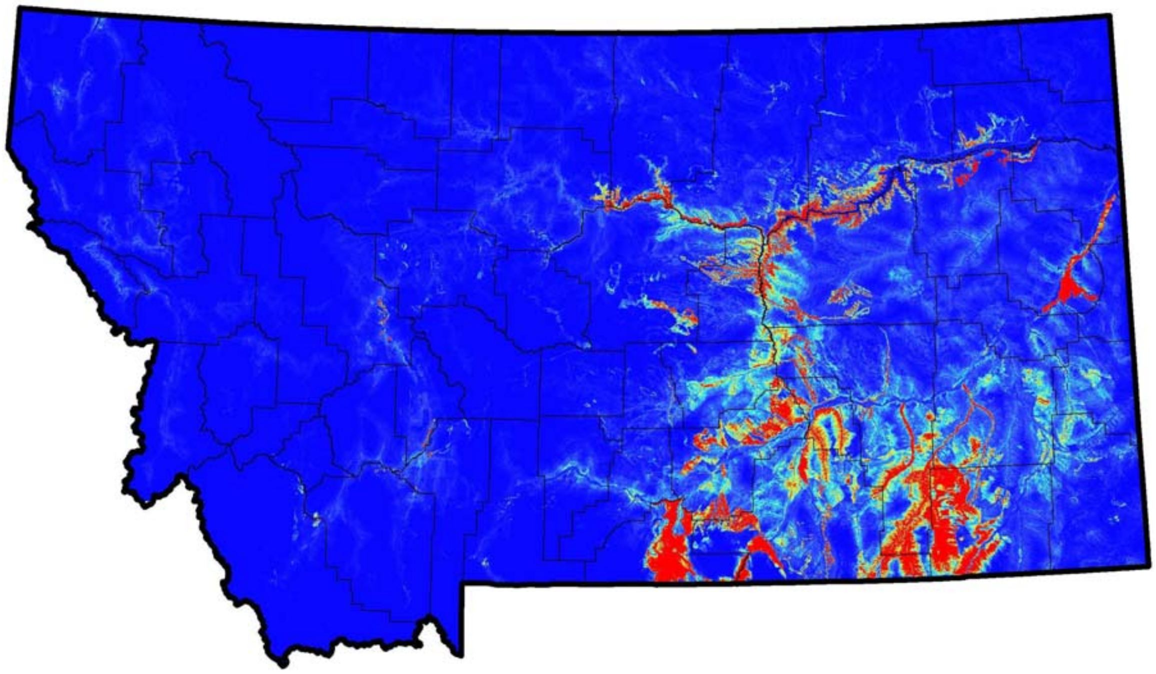
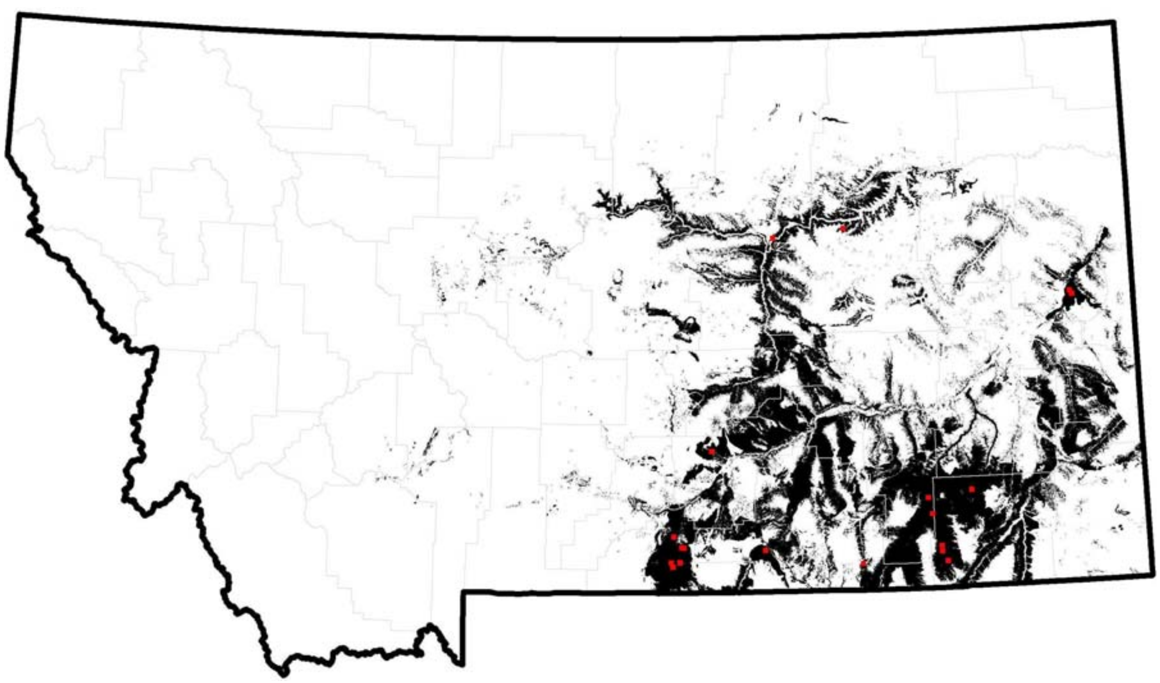


Figure D-26b. Predicted suitable habitat and training observations for *L. triangulum.*



Gophersnake (*Pituophis catenifer*)

The state-wide continuous model predicts suitable habitat for the species in several regions that currently lack observations (Figures D-27a-b). These include: (1) the area between Great Falls and the Canadian border within the Blackfeet Indian Reservation; (2) scattered areas between the Yellowstone and Missouri Rivers; (3) the Bighorn Canyon National Recreation Area; and (4) regions of the Blackfoot, upper Clark Fork and Little Blackfoot Rivers. Similar to *C. constrictor*, the continuous and low cutoff threshold models predict separation between eastern and western populations of the species based on availability of suitable habitat (Figures D-27a-c). However, as compared to *C. constrictor*, there are larger areas of predicted suitable habitat that extend up the Clark Fork River to Butte and over Deer Lodge Pass to the Big Hole River that may provide limited connectivity. Characteristics of environmental variables of importance (Table 3.5) include average maximum July temperatures greater than 28 °C, areas within 2,500 m of major streams, elevations less than 1,500 m, deciduous forest and shrubland cover types, and annual precipitation less than 50 cm. Aspect and the solar radiation indices did not improve the models as indicated by the jackknife plots of training gain, test gain, and AUC. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.897 and 0.96, respectively) indicated good model fit and the AVI was better than the GAP model (0.89) (Table 3.6). GAP predicted a 39 percent (110,378 km^2) larger area than the low cutoff (Table 3.7). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance. However, the average deviance of test occurrences was high relative to many species and a number of grassland habitats that would likely support the species in various portions of its range are not recognized as suitable habitat by the model (Figures D27b-c). Where they overlap, the low, moderate, and optimal habitat suitability class thresholds appear to be superior to GAP in that they more appropriately reflect the relative suitability of habitat. Until more observations are available in these grassland habitats, it would probably be best to use deductively based models in tandem with the continuous model.

Figure D-27a. Continuous state-wide habitat suitability index for *P. catenifer*.

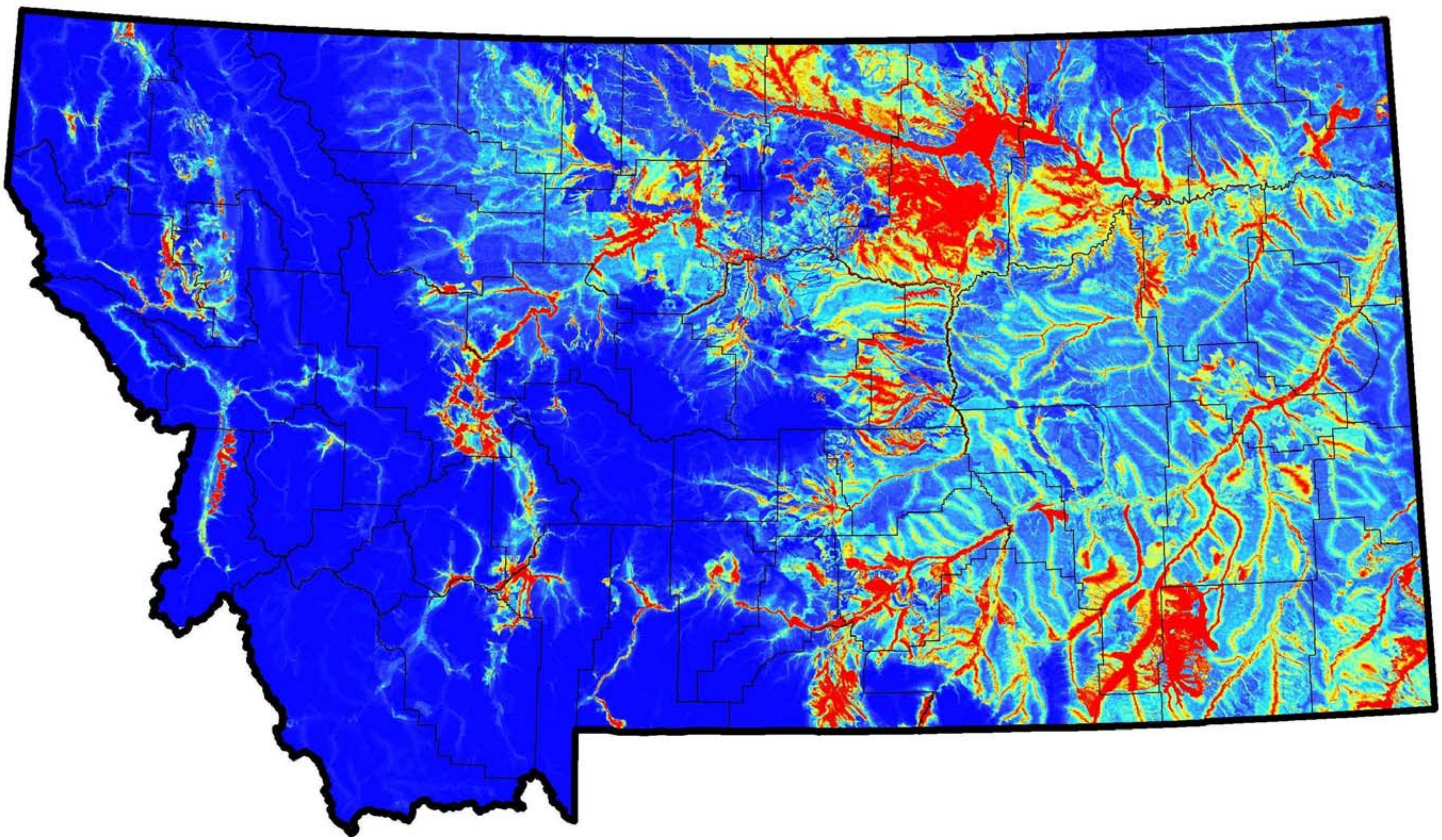


Figure D-27b. Predicted suitable habitat and training (squares) and test (stars) observations for *P. catenifer*.

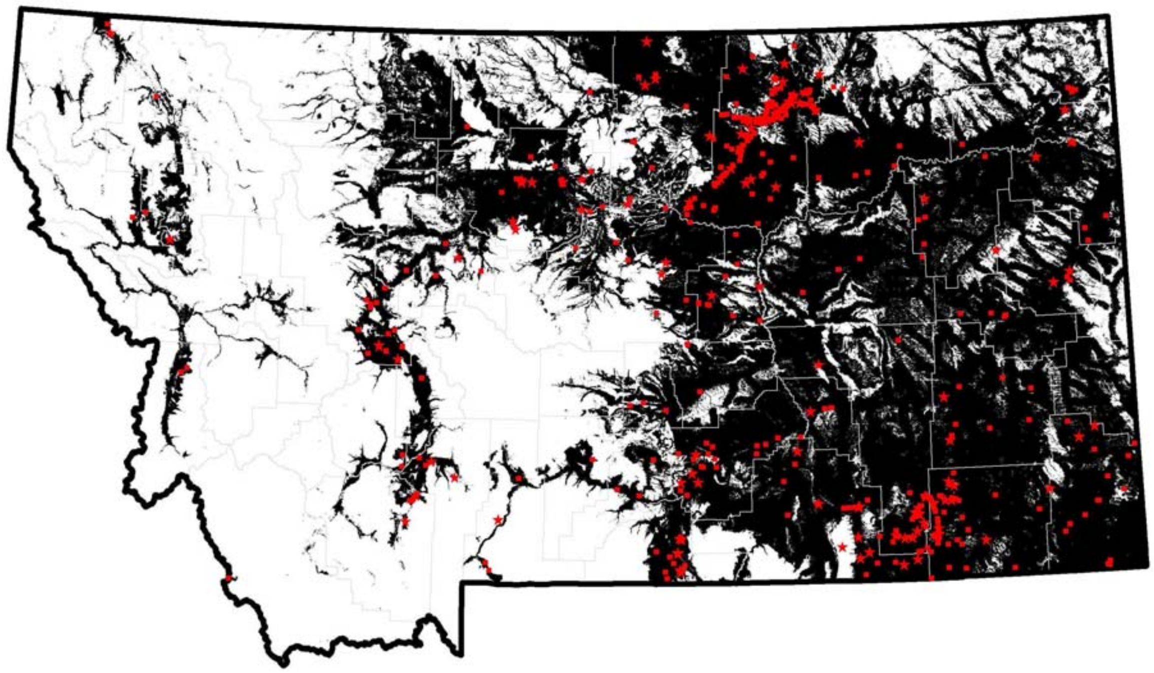
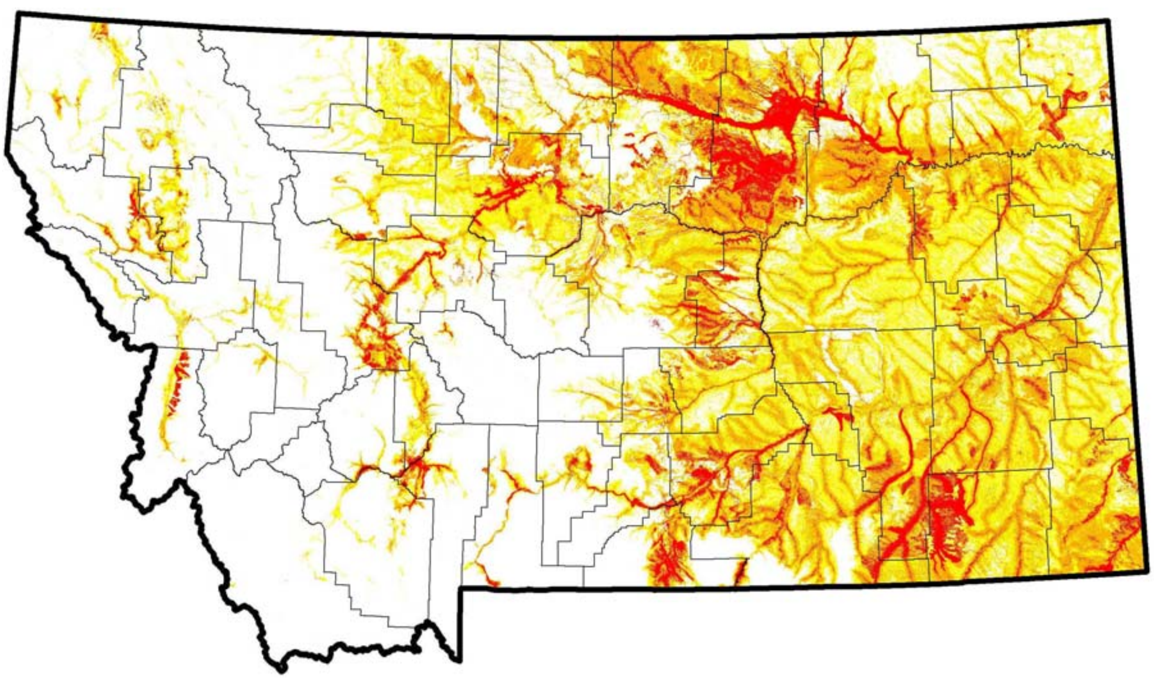


Figure D-27c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *P. catenifer*.



Terrestrial Gartersnake (*Thamnophis elegans*)

The state-wide continuous model seems to appropriately predict the distribution of the species across its known range, including broader distribution west of the Continental Divide as compared to the patchier and more stream dependent distribution east of the Continental Divide (Figures D-28a-b). The extent of the species' range is probably fairly well understood. Characteristics of environmental variables of importance (Table 3.5) include areas within 500 m of major streams, annual precipitation of less than 180 cm, slopes less than 20 degrees, and average maximum July temperatures less than 27 °C. Aspect and the solar radiation indices did not improve the models as indicated by the jackknife plots of training gain, test gain, and AUC, and the solar radiation indices actually made the model worse in the jackknife plot of test gain. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.897 and 0.96, respectively) indicated good model fit and the AVI was better than the GAP model (0.89), which was based on the majority of cover types, except water, below 3,300 m (Table 3.6, Hart et al. 1998). As a result the low cutoff threshold model predicted a 59 percent (178,895 km^2) smaller area than GAP (Table 3.7, Figure D-28b). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, and low Kappa values for both the low cutoff threshold and GAP models (Table 3.8). The low cutoff threshold model outperformed GAP as evaluated by overall map accuracy and Kappa. High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds appeared to be appropriate at the scale of the species' range as well as the level of a public land survey section in almost all areas examined (Figure D-28c).

Figure D-28a. Continuous state-wide habitat suitability index for *T. elegans*.

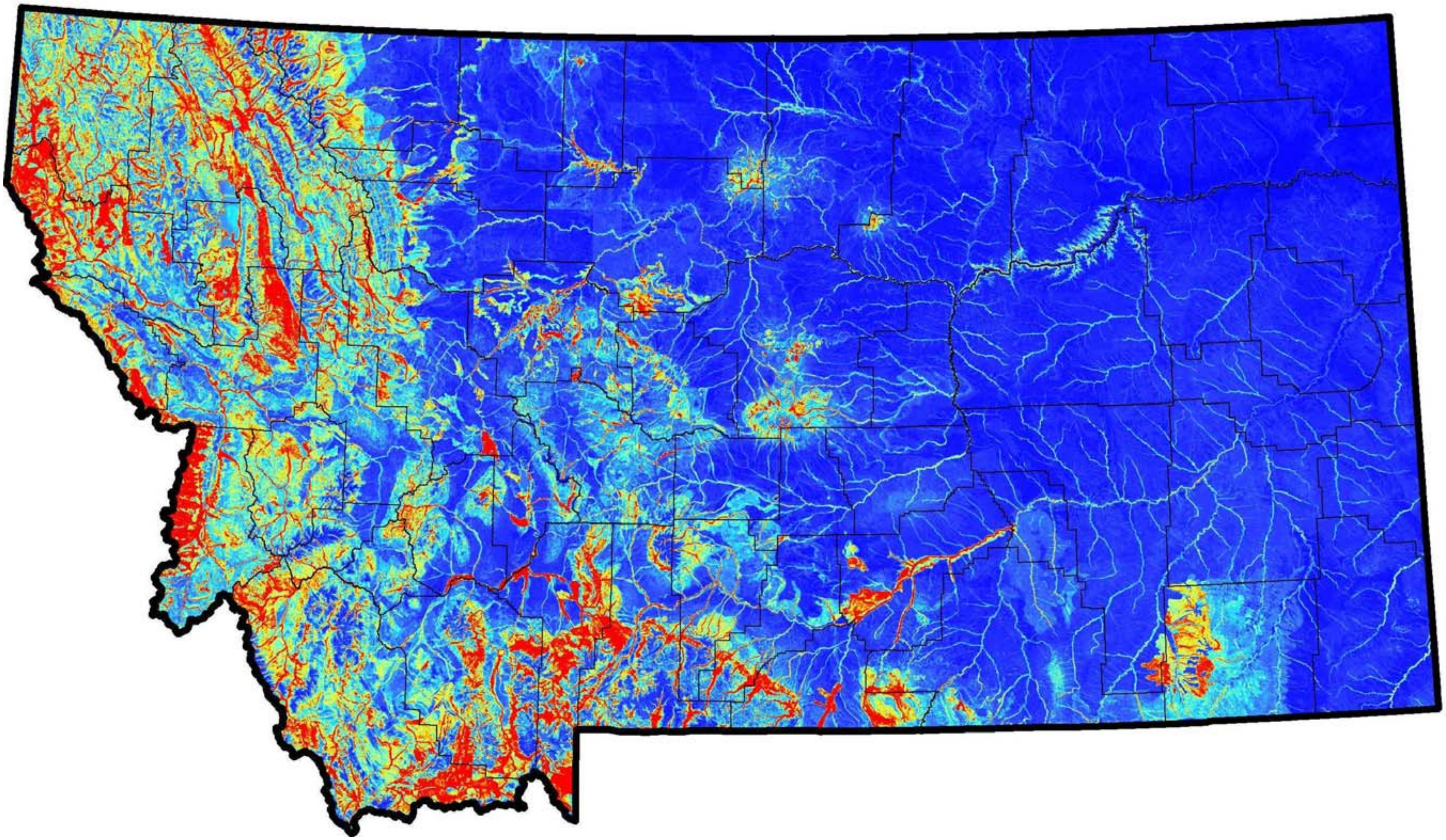


Figure D-28b. Predicted suitable habitat and training (squares) and test (stars) observations for *T. elegans*.

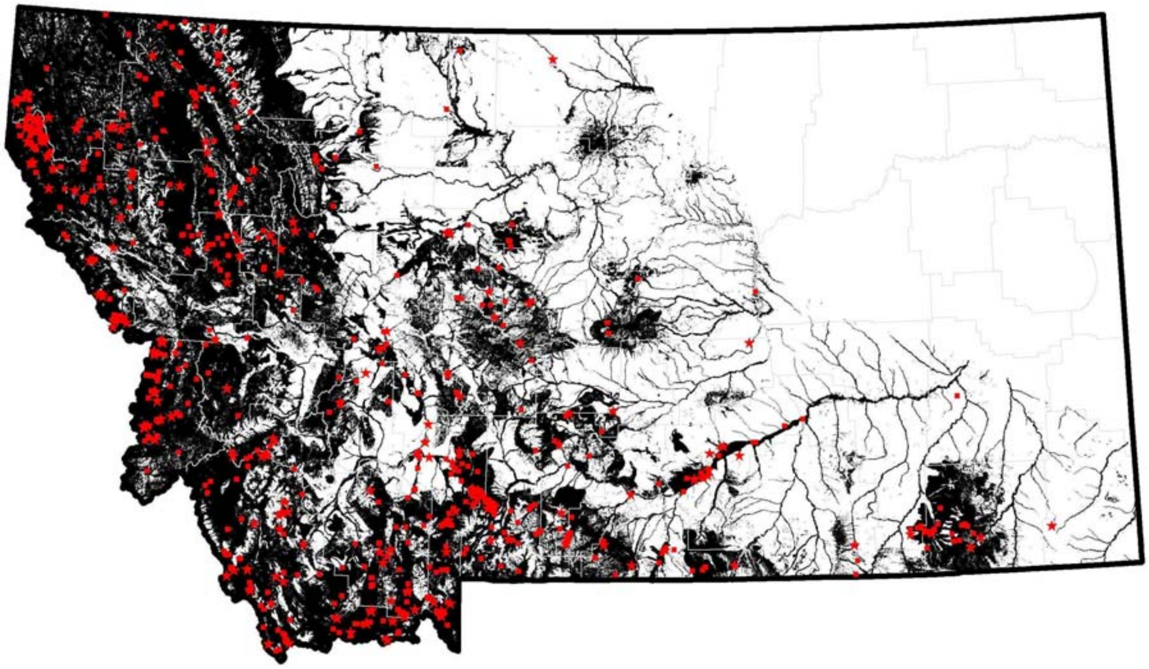
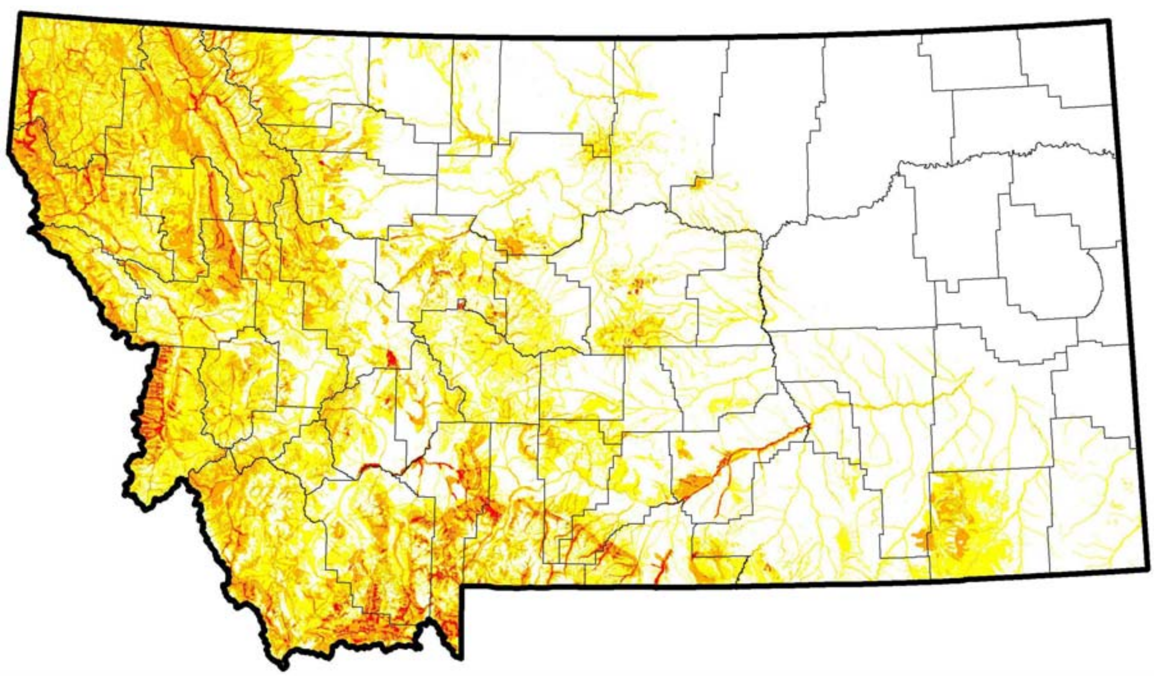


Figure D-28c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *T. elegans*.



Plains Gartersnake (*Thamnophis radix*)

The state-wide continuous model seems to appropriately predict the distribution of the species across its known range and indicates that the extent of that range is fairly well understood with the possible exceptions of Judith Basin County, southern Fergus County, and the Blackfeet Indian Reservation (Figures D-29a-b). Characteristics of environmental variables of importance (Table 3.5) include elevations below 1,000 m at the state-wide scale (much less important within the species' range), and average minimum January temperatures below -15 °C. Other environmental variable response curves showed no clear trends or cutoffs, probably as a result of the species being widespread across a variety of habitats within its range. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.908 and 0.98, respectively) indicated good model fit and the AVI was better than the GAP model (0.85), which was based on the majority of cover types, except water, below 2,100 m (Table 3.6, Hart et al. 1998). As a result, the low cutoff model predicted a 42 percent (83,083 km^2) smaller area than GAP (Table 3.7, Figure D-29b). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, low overall map accuracies, and low Kappa values for both the low cutoff threshold and GAP models, but the low cutoff threshold model outperformed GAP on each (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance. Low, moderate, and optimal habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section in most areas (Figure D-29c). However, the region in northeastern Montana with a relatively smaller predicted area may either be appropriate as a result of the different soil characteristics in this region, or it may be under predicting the species' distribution as a result of the lack of observations. Until there are more observations in this region, it may be best to use a combination of a deductive model and this inductively based continuous model.

Figure D-29a. Continuous state-wide habitat suitability index for *T. radix*.

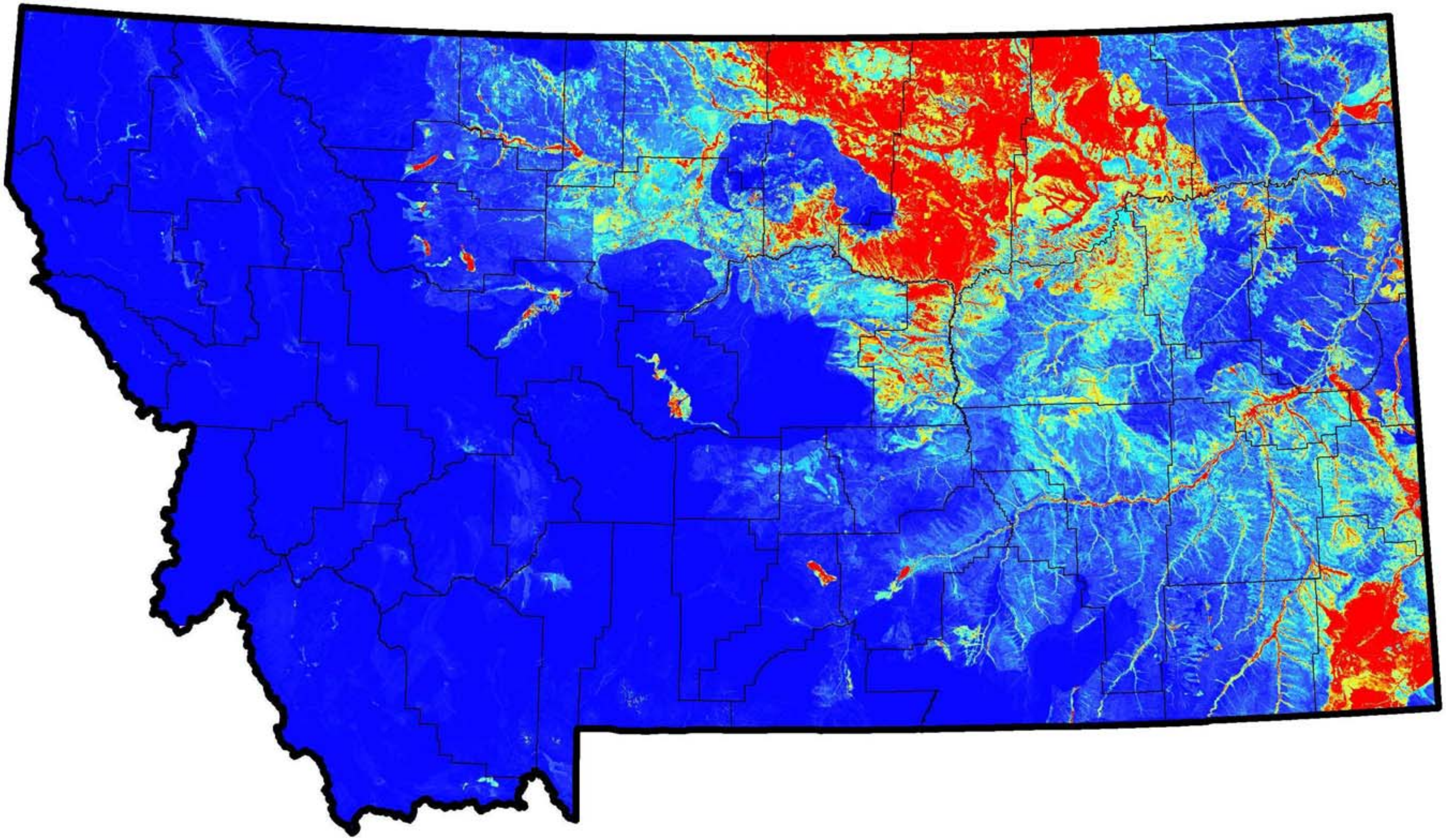


Figure D-29b. Predicted suitable habitat and training (squares) and test (stars) observations for *T. radix*.

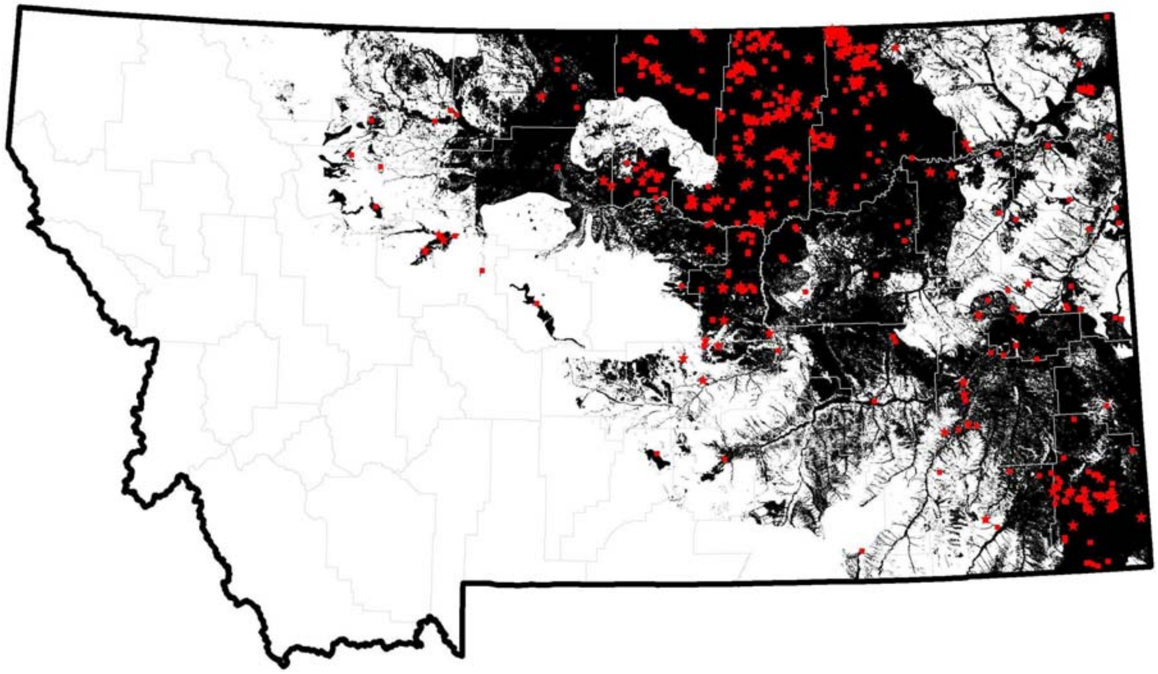
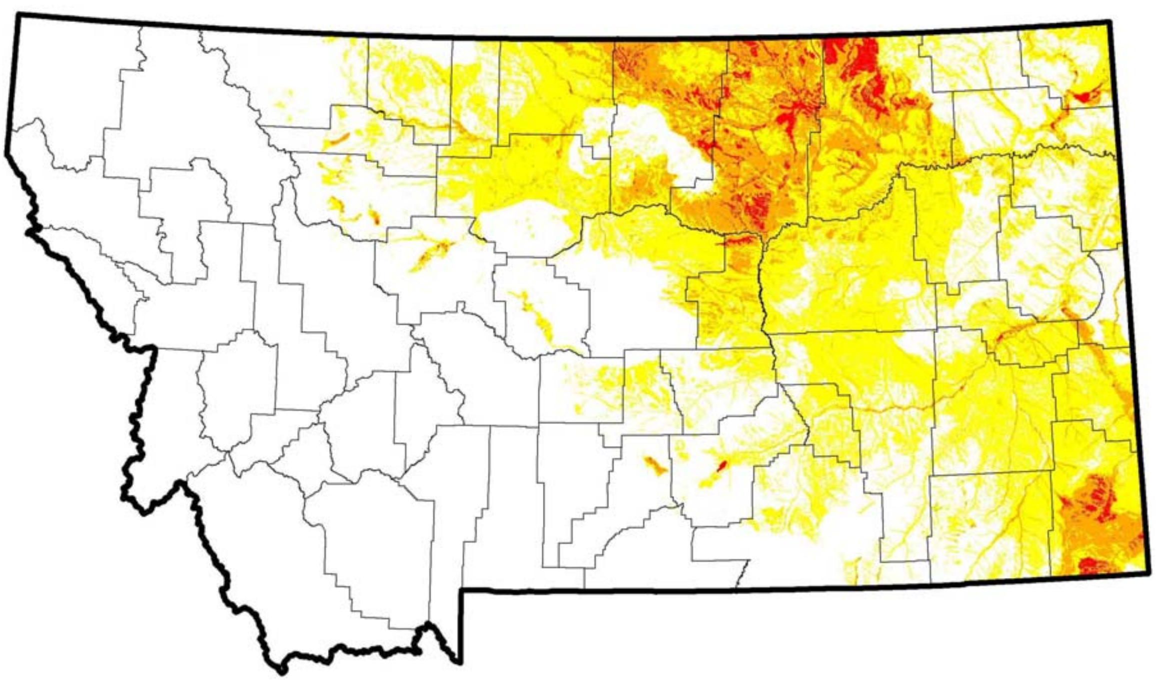


Figure D-29c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *T. radix*.



Common Gartersnake (*Thamnophis sirtalis*)

The state-wide continuous model seems to appropriately predict the distribution of the species across its known range, including broader distribution west of the Continental Divide as compared to the patchier and more stream dependent distribution east of the Continental Divide (Figure D-30a). The extent of the species' range is probably fairly well understood. Characteristics of environmental variables of importance include average minimum January temperatures greater than -12 °C, and annual precipitation of usually greater than 50 cm. Other environmental variable response curves showed no clear trends or cutoffs, probably as a result of the species being widespread across a variety of habitats within its range. Aspect, ruggedness, and the solar radiation indices did not improve the models as indicated by the jackknife plots of training gain, test gain, and AUC. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.935 and 0.91, respectively) indicated good model fit and the AVI was better than the GAP model (0.80), which was based on the majority of cover types within a 500-m stream buffer below 2,250 m (Table 3.6, Hart et al. 1998). As a result the low cutoff model predicted a 59 percent (122,459 km^2) smaller area than GAP (Table 3.7, Figure D-30b). Evaluation with the lentic site survey data indicated low omission error rates, high commission error rates, moderate levels of overall map accuracy, and low Kappa values for both the low cutoff threshold and GAP models (Table 3.8). High commission errors and low Kappa values likely resulted from the inability of models to identify local site variables of importance and the fact that single site visits fail to adequately assess true absence. Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance and low, moderate, and optimal habitat suitability class thresholds appeared to be appropriate at the scale of the species' range as well as the level of a public land survey section in almost all areas examined (Figure D-30c). However, some areas east of the Continental Divide are probably under predicted because the species is rarer in this region and, as a result, there are relatively few observations.

Figure D-30a. Continuous state-wide habitat suitability index for *T. sirtalis*.

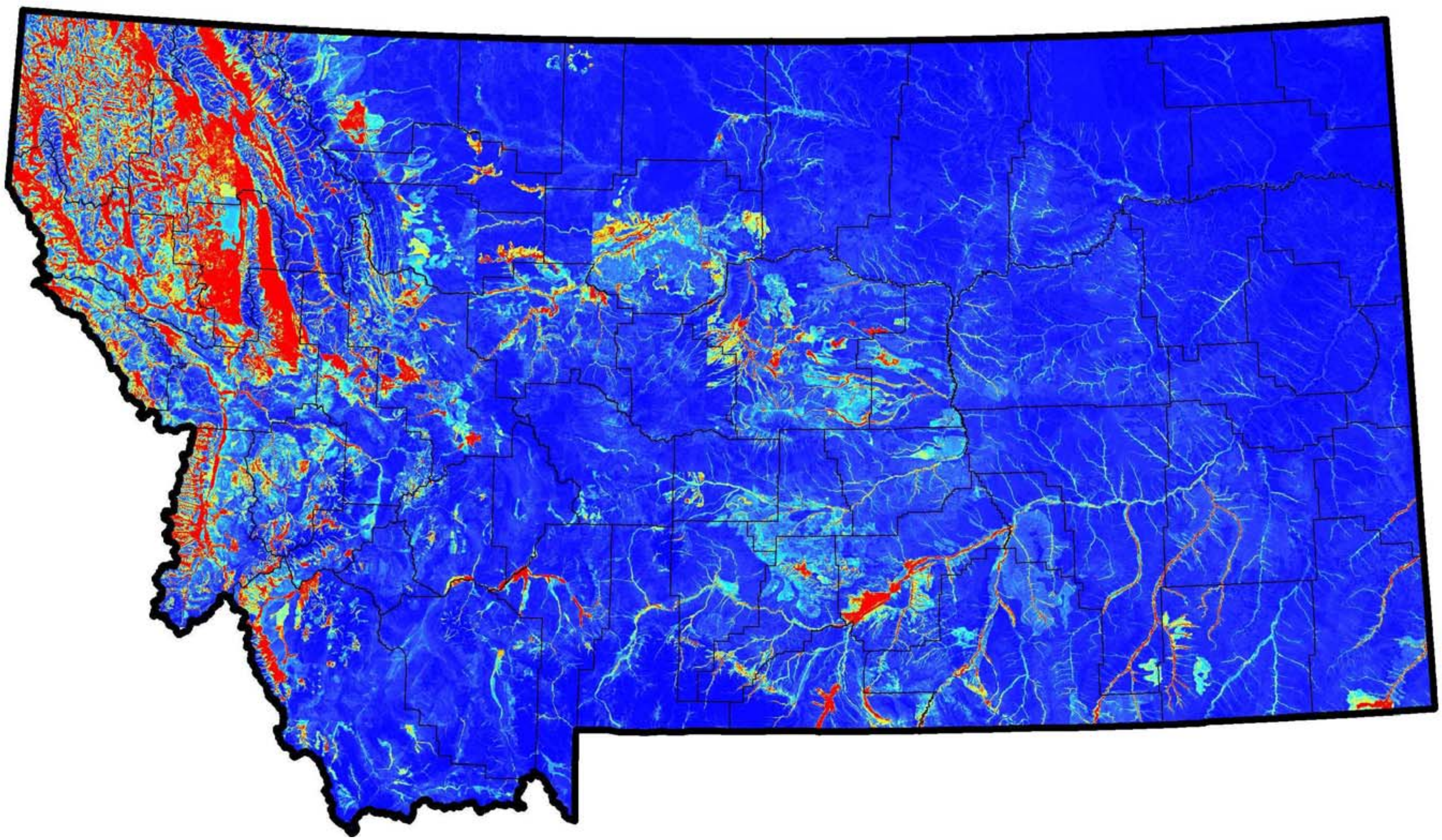


Figure D-30b. Predicted suitable habitat and training (squares) and test (stars) observations for *T. sirtalis*.

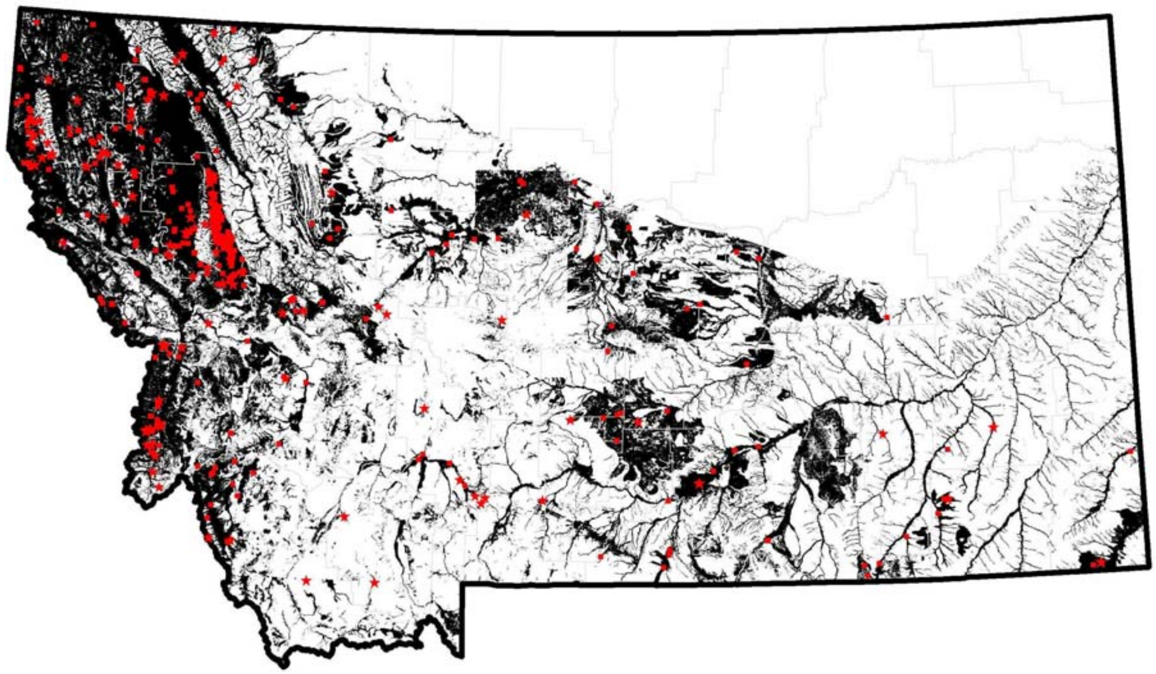
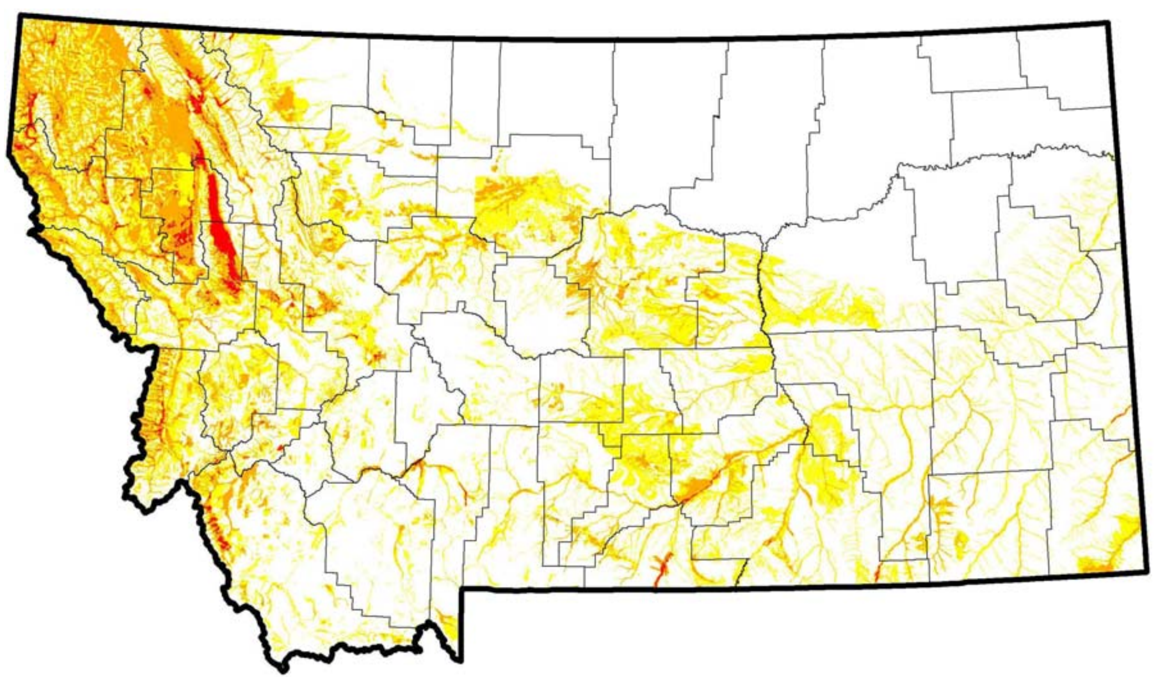


Figure D-30c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *T. sirtalis*.



Prairie Rattlesnake (*Crotalus viridis*)

The state-wide continuous model predicts suitable habitat for the species in several regions that currently lack observations (Figure D-31a). These include: (1) a large portion of the area between Great Falls and the Canadian border within the Blackfeet Indian Reservation; (2) scattered areas between the Yellowstone and Missouri Rivers; (3) extreme northeast Montana; (4) portions of the Blackfoot, upper Clark Fork and Little Blackfoot Rivers; (5) lower portions of the Bitterroot and Clark Fork Rivers; and (6) low elevations adjacent to major drainages in northwest Montana. Some of the areas in western Montana may represent historic habitat the species has been extirpated from. Similar to *C. constrictor*, and *P. catenifer*, the continuous models predict separation between eastern and western populations of the species based on availability of suitable habitat (Figures D-31a-c). However, as compared to *C. constrictor* and *P. catenifer*, there are larger areas of predicted suitable habitat that extend up the Clark Fork River to Butte and over Deer Lodge Pass to the Big Hole River that may provide limited connectivity. Characteristics of environmental variables of importance (Table 3.5) include areas within 1,500 m of major streams, average maximum July temperatures greater than 26 °C, annual precipitation usually less than 60 cm, woody wetland, grassland, and transitions between these two cover types, and elevations less than 1,700 m. Solar radiation indices actually hurt the models as indicated by the test jackknife plot. The AUC and AVI for test data evaluating the range-wide continuous and low cutoff threshold models (0.875 and 0.90, respectively) indicated good model fit and the AVI was approximately the same as the GAP model (0.92), which predicted a 46 percent (124,260 km^2) larger area (Tables 3.6 and 3.7). Visual inspection of test occurrence deviances at the range-wide and local-landscape scales revealed no large-scale patterns in the magnitude of deviance. In areas were there are a number of observations, the habitat suitability class thresholds appear to be appropriate at the scale of the species' range as well as the level of a public land survey section when there are occurrences in a region. However, the low cutoff threshold probably under predicts suitable habitat in large portions of eastern Montana where few observations have been reported (Figures D-31b-c). In these areas, it would probably be best to use deductive and continuous model outputs in tandem when considering this species in management plans.

Figure D-31a. Continuous state-wide habitat suitability index for *C. viridis*.

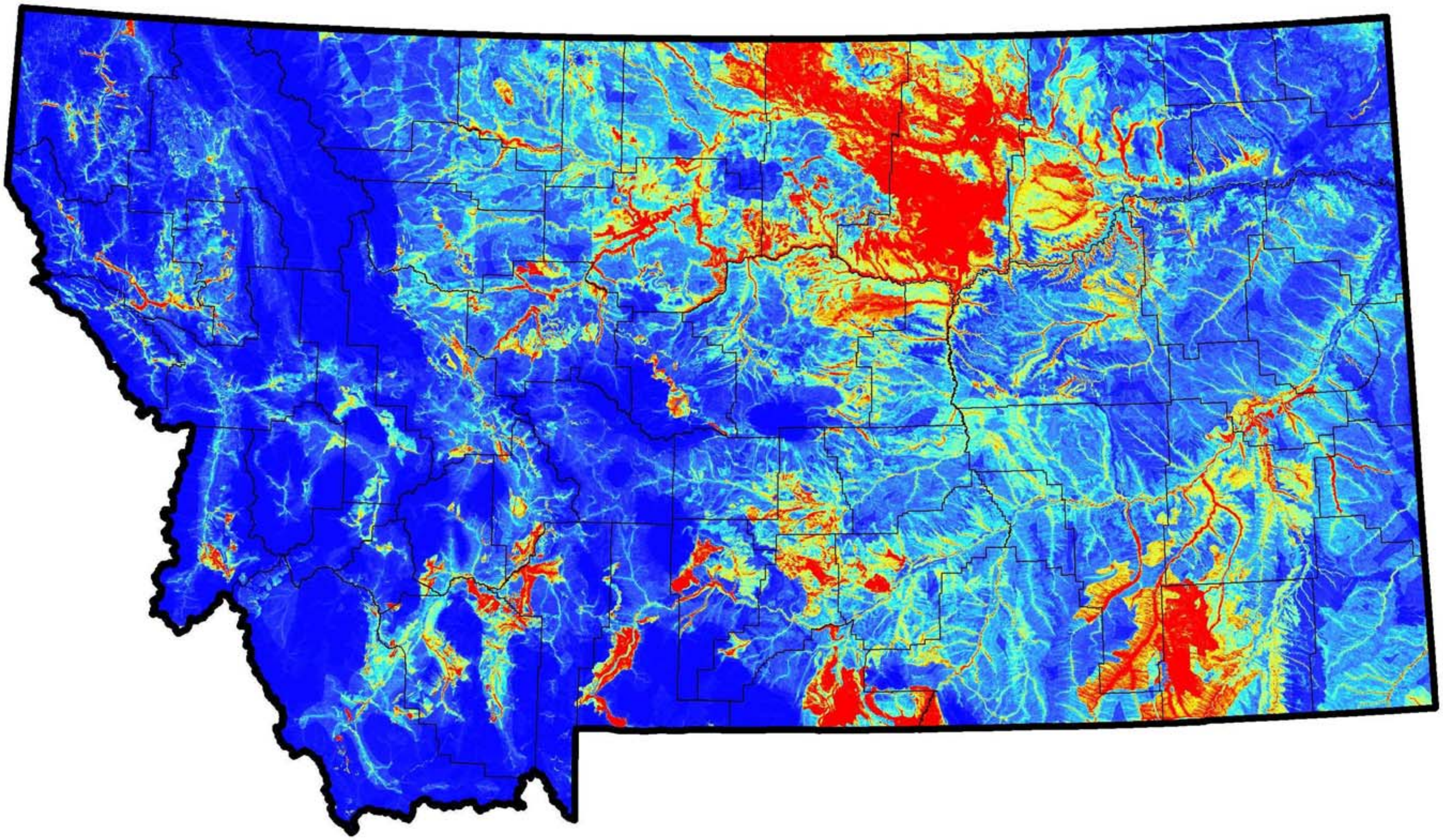


Figure D-31b. Predicted suitable habitat and training (squares) and test (stars) observations for *C. viridis*.

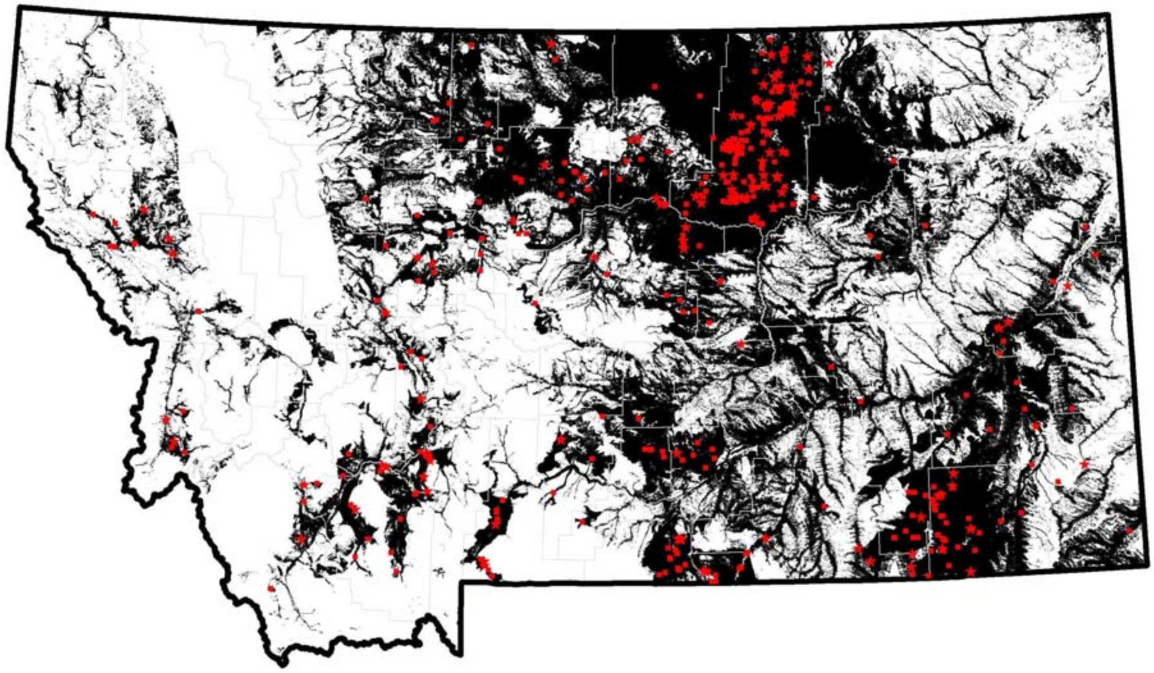
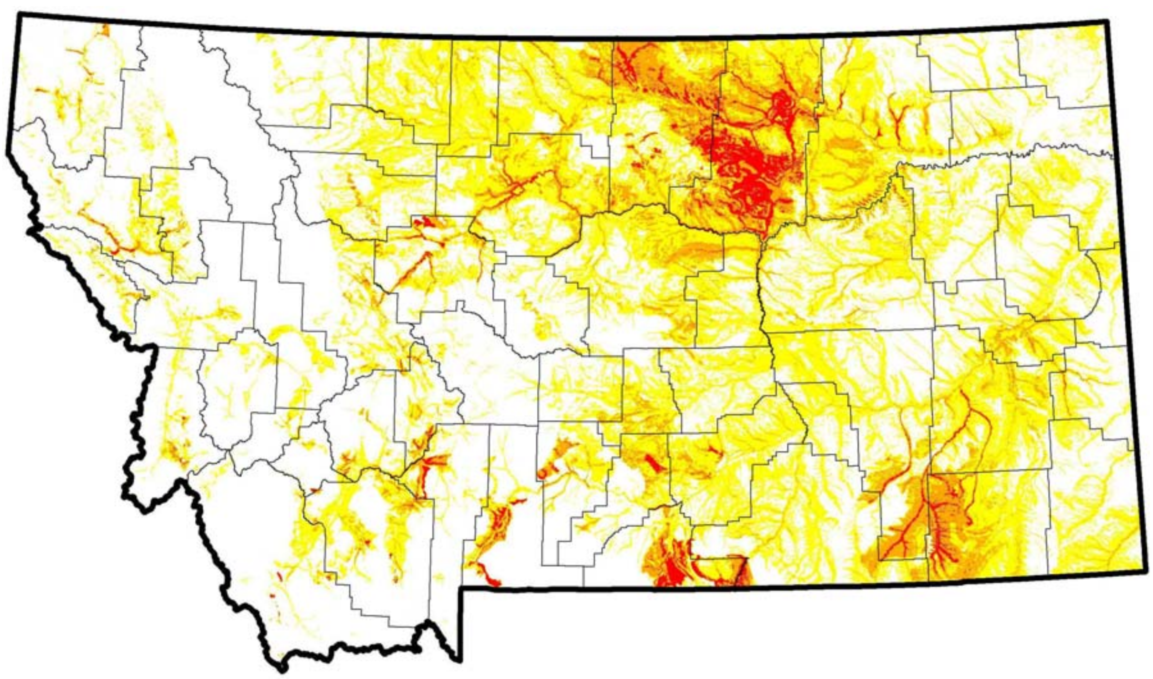


Figure D-31c. Predicted low (yellow), moderate (orange) and high (red) habitat suitability classes for *C. viridis*.



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