SPATIAL DISTRIBUTION AND HABITAT USE OF THE BLISS RAPIDS SNAIL

by

Barry M. Bean

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Barry M. Bean

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The following individuals read and discussed the thesis submitted by student Barry M. Bean, and they evaluated his presentation and response to questions during the final oral examination. They found that the student passed the final oral examination.

Matthew R. Dare, Ph.D.	Co-Chair, Supervisory Committee
Ian C. Robertson, Ph.D.	Co-Chair, Supervisory Committee
Peter Koetsier, Ph.D.	Member, Supervisory Committee

The final reading approval of the thesis was granted by Matthew R. Dare, Ph.D., Co-Chair of the Supervisory Committee. The thesis was approved for the Graduate College by John R. Pelton, Ph.D., Dean of the Graduate College.

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ABSTRACT

I conducted a three-year field study to gather information about the Bliss Rapids snail, *Taylorconcha serpenticola*, a rare aquatic mollusk endemic to the Snake River drainage of southwestern Idaho. The goals of my study were to 1) gain an understanding of the species' distribution and dispersion in the Snake River, 2) characterize habitat relationships, and 3) evaluate methods for monitoring the species' abundance. When monitoring at-risk species, it is important to understand their spatial distribution and habitat requirements in order to design a study that will provide reliable data with good statistical power. My study suggests the species is not limited to a small number of densely-populated colonies within specific habitat types as previously thought. Instead, the Bliss Rapids snail is patchily distributed throughout the study area. Bliss Rapids snails were found at the majority of my sample sites, but only occurred in 5-13% of the cobbles I sampled. The species exhibits contagious dispersion: the variance-to-mean ratio was greater than 1 for all four spatial scales I examined. When sites containing Bliss Rapids snails were paired between years, abundance was significantly correlated at three spatial scales when compared with a Spearman rank order correlation test. Bliss Rapids snail abundance was positively correlated with bed shear stress, and negatively correlated with distance from the nearest upstream rapid and bank slope (angle), but correlations were weak in both cases. The species was more abundant in the deeper (0.5-1.5 m) transects compared to shallow transects (0-0.5 m) as well as north-facing aspects compared to south-facing aspects. I used a bootstrap method to simulate the probability

of not detecting the species at a site (bank-section) when occurrence rates were low. This simulation revealed that increasing the bank-section sample size from 40 to 100 cobbles would reduce the probability of not detecting the species when they were present from 0.39 to 0.08 when the occurrence rate was 0.025. I also performed a Monte Carlo simulation-based power analysis to determine the sample size needed to identify 10, 20, 25, 35, and 50% declines in Bliss Rapids snail abundance over a five-year period using data from over 15,000 cobble counts. The analysis indicated that declines in abundance of 10-50% could be detected with statistical power of at least 0.8 over a five-year period (with α=0.1). I recommend a protocol to detect a 25% decline in abundance over a five-year period, which would require sampling 6,000 cobbles annually.

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INTRODUCTION

Rare or elusive species are the focus of many wildlife management studies because low population sizes and limited distribution generally increase an organism's risk of extinction (Fagan et al. 2002, Hartley and Kunin 2003, Fox 2005). The Endangered Species Act (1973) requires that threatened or endangered species and their habitat be protected in the United States. One of the first steps in developing effective protection for a rare species is to identify sampling techniques that reliably measure population sizes and trends for the species. Achieving this goal can be hampered by the fact that gathering reliable data on rare or elusive species is, by definition, difficult (e.g., Al-Chokhachy et al. 2009). The consequences of failing to detect a declining trend in a biological population may, in the worst case scenario, contribute to the species' extinction through inaction or continued inappropriate anthropogenic activities (Taylor and Gerrodette 1993). Therefore, rare species research and monitoring should aim to improve the quality of data such that resource managers can make informed decisions based on the best possible empirical data (Gibbs et al. 1999).

Although it is generally accepted that rare species have sparse or restricted spatial distribution patterns (Kattan 1992, Cunningham and Lindenmayer 2005, MacKenzie et al. 2005, Hernandez et al. 2006), the definition of rarity is variable in the ecological literature (Gaston 1994). Rabinowitz (1981) described a theoretical framework for seven types of species rarity. Her description of rarity is based on geographic range (large or small), habitat specificity, and local population size. An alternative approach for defining a species as rare or common is based on sampling success (i.e., rate of detection). For example, Green and Young (1993) define a rare species as having a mean density of less than 0.1 individuals per sample unit.

Rare species research often necessitates the development of novel collection methods in order to optimize field collection efforts and improve statistical rigor. Widely accepted methods for sampling more common species may not yield acceptable results when attempting to detect species that are uncommon or elusive (Venette et al. 2002, Strayer and Smith 2003, Elith et al. 2006, Hernandez et al. 2006, Mazerolle et al. 2007). Statistically, organisms are considered rare when they exhibit a low probability of detection (less than 0.1), so improved detection techniques alone can alter a species' perceived rarity (Green and Young 1993, Strayer and Smith 2003, Hernandez et al. 2006, Haddad et al. 2007). Therefore, it may be necessary to adapt survey methods to match the biological characteristics of the target species. I examined the spatial distribution and habitat use of the Bliss Rapids snail (hereafter, BRS), *Taylorconcha serpenticola* Hershler, Frest, Johannes, Bowler, and Thompson, 1994 (Family: Hydrobiidae). BRS are a threatened species, protected by provisions of the U.S. Endangered Species Act and my research was motivated by a need to develop monitoring protocols sufficient to detect population declines.

The BRS is an aquatic snail endemic to the Snake River drainage in Idaho (Figure 1). The species was first recognized as a new taxon by Taylor (1982), but not formally described until after it was listed as threatened under provisions of the U.S. Endangered Species Act (U.S Fish and Wildlife Service 1992, Hershler et al. 1994). This small (2.0-

2.5 mm) Hydrobiid snail is known to occur in three free-flowing reaches of the Snake River. Prior to my project, the species was known to exist at six sites in the Snake River (Hershler et al. 1994, Idaho Power Company unpubl. data). Numerous populations also occur in tributaries and springs that drain into the Snake River (Bates et al. 2009). BRS are not known to inhabit reservoirs except near spring outlets (Frest and Johannes 1992, Hershler et al. 1994, Cazier 1997, Richards et al. 2005).

Potential threats to BRS include spring habitat degradation from substantial aquaculture activity in the region, water quality impairment, invasive species (e.g., New Zealand mudsnail [*Potamopyrgus antipodarum*]), hydroelectric operations, and a downward trend in discharge from the Snake River Plain Aquifer (U.S Fish and Wildlife Service 1992, Richards and Lester 1999, Baldwin et al. 2006). Based on the snail's preference for ventral and lateral sides of cobbles, Bowler (1990) suggested the species is photophobic. The species prefers stable rocky substrates, does not burrow into soft substrates, and is generally found in rocky areas lacking soft sediment (Hershler et al. 1994, Bean, Fore, and Van Winkle 2009).

Previous BRS studies focused on population trends at known locations near spring outlets at a spatial scale of less than 50 m, and were not intended to characterize the species' distribution (Stephenson and Bean 2003, Stephenson et al. 2004). Thus, the first goal of my study was to describe the spatial distribution and dispersion of BRS throughout a 31.6 km section of the Snake River. It is important to understand rare species' distribution patterns in order to assess extinction risk (due to local extinctions or genetic drift) and to develop monitoring protocols for population trend analysis (Gibbs et al. 1999, Ganey et al. 2004, McDonald 2004, McPherson et al. 2004). Therefore, I selected sample sites randomly throughout the study area to determine if BRS occurred in habitats where they were not previously documented.

In addition to distribution patterns, it is important to consider species' dispersion patterns when determining the temporal and spatial extent of a sample frame in order to maximize the efficacy of field collection efforts (Cochran 1977, Gibbs et al. 1999, Ganey et al. 2004, McDonald 2004, McPherson et al. 2004). A species' dispersion patterns can be broadly characterized as random, uniform, or contagious. With random dispersion, the presence of an individual in one location provides no information about the probability of another individual occurring nearby. Regular dispersion is characterized by evenly spaced individuals. Organisms that occur in clusters demonstrate contagious dispersion. Many organisms exhibit contagious dispersion where groups of individuals occur in patches of suitable habitat (Rabinowitz 1981, Brown et al. 1995). Contagious dispersion can result in lower rates of detection if only a fraction of the available habitat is surveyed (Green 1979). Increasing the spatial scale of a sample or collecting numerous small samples across a large area may compensate for patchy distributions (Sawyer 1989, Yamamura 1990, Horne and Schneider 1995, Engen et al. 2008). I used a statistical approach to evaluate field data and determine the dispersion pattern of BRS in the study area.

Species distribution and dispersion patterns are often correlated with habitat variables (Rabinowitz 1981, Brown et al. 1995). As mentioned above, previous BRS research focused on known colonies adjacent to spring outlets while little effort was spent on other potentially suitable habitat locations (Richards et al. 2005). There are many documented cases in the literature of rare or elusive species occurring in habitats that were previously thought to be unsuitable simply because researchers assumed the species were habitat specialists and thus failed to sample elsewhere (e.g., Good and Lavarack 1981, Maggini et al. 2002, Poon and Margules 2004). In the case of snails, three threatened or endangered species that occur in the Snake River have recently been found in reservoirs despite previous claims that reservoirs were unsuitable habitat for these snails (U.S. Fish and Wildlife Service 2007, Idaho Power Company unpubl. data, R. Newman [Bureau of Reclamation] pers. comm.). Thus, the second goal of my research was to characterize habitat use patterns of BRS. I examined correlations between BRS occurrence and abundance and habitat metrics related to light and water velocity based on the assumption that the species is photophobic and requires clean cobbles free of fine sediment (Hershler et al. 1994). My intent was to develop a better understanding of the habitat requirements of the species in order to guide future monitoring of the species' abundance and distribution.

With any sampling protocol, there is a risk that a species will go undetected when it is present. In the field of statistics, false negatives can lead to Type II errors. Failure to detect a species when it is present or failure to detect a population-level decline (an example of a Type II error) can have serious implications for the validity of occurrence and abundance models for a species and management decisions based on those data (Gerrodette 1987, Hatfield et al. 1996, Ganey et al. 2004, Mazorelle et al. 2007). For example, catastrophic losses of commercial fisheries have occurred when monitoring programs were designed to protect against a Type I error (false positives in statistical analysis), but had low statistical power to protect against a Type II error (Peterman 1990, Dayton 1998).

Whereas Type I errors, such as misidentification of a target species are uncommon, the probability of making a Type II error is often unknown for single-visit surveys when the probability of detection is less than one (Thompson 2002). Detection probability is generally correlated with abundance, but can also be influenced by landscape features (Nupp and Swihart 1996, Mancke and Gavin 2000, Odell and Knight 2001), environmental conditions (Pendleton 1995), and observer bias. For example, Hairston and Wiley (1993) reported that observed fluctuations in salamander abundance were due to variation in student motivation to search for the salamanders. Conducting a statistical power analysis, often through the use of statistical simulation, is one way to minimize the chance of making a Type II error while monitoring rare species. According to convention, the desired minimum statistical power (calculated as 1 – the Type II error rate) is 0.8 (Gibbs and Melvin 1997, Al-Chokhachy et al. 2009). I evaluated the probability of not detecting BRS when they were present based on occurrence rates I observed in my data. My intent was to insure future monitoring protocols were robust to the potential for Type II errors.

It is common in studies of rare species to encounter data sets that exhibit low statistical power to detect even a 50% decline over a 5–10 year period (Taylor and Gerrodette 1993, Gibbs and Melvin 1997, Ham and Pearsons 2000, Hatch 2003, Al-Chokhachy et al. 2009). For example, Al-Chokhachy et al. (2009) found that detecting a 50% decline in bull trout, *Salvelinus confluentus*, would require sampling 48% of their study area. Thus, the third goal of my research was to explore methods by which I could monitor BRS abundance with statistical power sufficient to detect declines in abundance ranging from 10-50% while limiting the probability of making a Type II error.

METHODS

Study Area

The study area consisted of two free-flowing reaches of the Snake River in southwestern Idaho, which are separated by Bliss Reservoir (Figure 2). The third river reach known to contain BRS is less than 100 meters in length and supports a relatively small population (pers. obs.) and was not included in my study. The upper reach of the study area begins immediately downstream of Lower Salmon Falls Dam near the town of Hagerman, Idaho, and flows approximately 10.6 km before entering Bliss Reservoir at Shoestring Bridge. The lower reach flows from Bliss Dam to the edge of the species' known range at the confluence with Clover Creek, approximately 21 km downstream from the dam. In this area, the river flows within a basalt canyon, and glides are the most common habitat type, with pools and rapids being present but less common (Welcker et al. 2009a). The majority of the substrate is angular basalt boulders and cobbles that have fallen into the river from the canyon walls. The minimum, median, and maximum discharge during the period 1998-2007 for the upper reach are 81, 210, and 1,348 m^3/s, respectively, while the same values for the lower reach are 127, 218, and 1,458 m^3/s, respectively (Idaho Power Company, unpubl. data). The differences in discharge for the two reaches are due to input from tributaries, mainly the Malad River (Borden and Conner 2009a; Figure 2). The upper reach has a gradient of 1.9 m/km and a width/depth ratio of 11.0, while the lower reach has a gradient of 1.15 m/km and a width/depth ratio of 8.2 (Idaho Power Company unpubl. data).

Currently, the Snake River is regulated by 15 dams from its headwaters near Jackson Lake, in Wyoming, to its confluence with the Columbia River. Much of the water upstream of the study area is diverted for agricultural use. The majority of flows within the study area originate from the springs of the Snake River Plain Aquifer (Baldwin et al. 2006). Such springs are numerous in the study area.

Springs along the Snake River have been extensively developed for aquaculture, with over 70 % of hatchery-raised trout in the U.S. reared in these springs (Shelton et al. 1994). These hatcheries contribute a considerable amount of nutrients to the Snake River. For example, the four largest hatcheries in the valley contribute over 45 metric tons of total suspended solids (TSS) to the Snake River annually (Buhidar 2005). In addition to nutrient loading, hatchery construction activities and water diversions have altered many springs to the point that they are no longer inhabitable by BRS (U.S. Fish and Wildlife Service 1992). The Idaho Department of Environmental Quality (1992) has designated the study area as “water quality limited” due to dissolved oxygen, temperature, and nuisance plant growth that did not meet standards for coldwater biota.

Hydroelectric dams also have impacts on BRS habitat. The species has not been detected in reservoirs except at the mouths of springs (U. S. Fish and Wildlife Service 1992, Hershler et al. 1994). Lower Salmon Falls Dam and Bliss Dam operate as peak-loading facilities, varying discharge downstream of the projects in order meet power demand. Peak-loading operations (for which additional discharge is routed through hydroelectric turbines to meet electrical demands) periodically dewater shoreline habitat, exposing BRS to desiccation and temperature extremes (Richards and Kerans 2008, Richards and Arrington 2008, Bean, Van Winkle, and Clark 2009, Conner et al. 2009). Peak-loading is a common practice for hydroelectric projects, as electrical demand varies over time. Hydropower is better-suited for meeting variable electrical demand compared to coal-fired plants, wind, geothermal, or solar.

The Federal Energy Regulatory Commission (FERC) requires minimum discharge from Lower Salmon Falls and Bliss Dams of 99 and 127 m^3/s, respectively. During 2005 and 2006, both dams were operated as run-of-river projects, meaning the discharge upstream and downstream of each project were equal. In 2007, both projects were operated in peak-loading mode. FERC allows water surface elevation downstream of Lower Salmon Falls Dam to vary by 0.76 m/hr and 1.5 m/day. Bliss Dam can vary water surface elevation by 0.9 m/hr and 1.8 m/day. These water surface elevation changes are measured at the tailrace of each dam, so this change attenuates downstream. Water surface elevation changes more dramatically when rising, as peak-loading events generally last only 1-3 hours, while refilling of the reservoir takes place over a much longer duration. While increasing discharge may elevate shear stress and dislodge BRS, reducing discharge dewaters habitat and is more likely to harm this aquatic species.

Sample Design

Lotic systems can be viewed in terms of a spatial hierarchy (Frissell et al. 1986). My sample design consisted of six nested spatial scales (Figure 3). The study area (scale 1) was made up of two river reaches (scale 2) that were separated by Bliss Reservoir.

Both river reaches were divided into 50-m sections (scale 3) with GIS (ESRI ArcMap 1999-2006) using geo-rectified aerial photographs and bathymetry (Conner et al. 2009). The 50-m sections were measured along the thalweg, so the length of each shoreline varied as the river meandered. I used a random selection procedure to identify 10% of these sections (n=63) to sample. I sampled along the shoreline on each side of the river within each section (bank-section, scale 4). Within each bank-section, I searched for snails and measured habitat along two transects parallel to the river bank (scale 5), one in shallow water (0-0.5 m deep), and one in deeper water (0.5-1.5 m deep). The width of individual transects varied based on channel morphology (i.e., bank-sections with steep shorelines had narrower transects compared to bank-sections with low gradients). I adapted a Wolman pebble count (Wolman 1954), which is normally used to assess substrate size, to sample cobbles for BRS (see Richards et al. 2005). Twenty individual cobbles (scale 6) were collected from each transect (40 cobbles per bank-section, and 80 per section) and examined for both the occurrence and abundance of BRS. For the sake of this study, a cobble was defined as a naturally occurring rock that was small enough to comfortably remove from the river bed and examine for BRS. The cobbles I sampled ranged in size from 6 to 58 cm along the long axis. I did not sample for BRS on fine substrate because I have not collected a BRS in fine sediment in 10 years of sampling with a suction dredge.

Data were collected during May and June of 2005 and 2007, and the September and October of 2006. Sampling took place in the fall of 2006 due to a high spring runoff, which made spring sampling unsafe. The sections randomly selected in 2005 were revisited in 2007 to evaluate variability between years at the section, bank-section, and transect scale.

Spatial Distribution

I used field count data from the section, bank-section, transect, and cobble scales to construct frequency distributions in order to compare and contrast BRS distributions at these spatial scales. I based the sample size of 40 cobbles per bank-section from work by Richards et al. (2005) on a congener, *T. insperata*, in Hells Canyon of the Snake River. I selected a section length of 50 m to ensure there were enough cobbles available to collect in all habitat types, including sections dominated by bedrock or fine substrates where cobbles were relatively uncommon.

In order to examine the dispersion patterns of BRS at different spatial scales, I calculated the variance-to-mean ratio (VMR = variance/mean) for section, bank-section, transect, and cobble scales (Zar 1999). VMR values between 0 to 1.0 suggest random dispersion, VMR values equal to 1.0 suggest uniform dispersion, and VMR values greater than 1.0 suggest contagious dispersion. Contagious dispersion is the most common pattern observed in ecology, as organisms are often concentrated in patches of suitable habitat (Brown et al. 1995, Krohne 1998).

I tested the correlation of BRS abundance between years for paired sites at three spatial scales using Spearman rank order correlation (Zar 1999) to assess changes in BRS abundance between years for these spatial scales. Sites visited in 2005 and 2007 were paired, but a different set of sites were visited in 2006. Therefore, I tested the correlation of BRS abundance between years for section, bank-section, and transect scales between 2005 and 2007.

I did a post-hoc analysis to determine the probability that I failed to detect BRS in bank-sections with a sample size of 40 cobbles per bank-section. I used bootstrap simulations (Manly 2007) for bank-sections to approximate the probability of obtaining a false negative for bank-section sample sizes of 20, 40, 80, and 100 cobbles. I bootstrapped using probabilities of 0.025, 0.05, and 0.075 representing one, two, or three occurrences per 40 cobbles, respectively. I defined an occurrence as a cobble with at least one BRS detected, regardless of the total number of snails present on the cobble. Each combination of sample size and rate of occurrence was simulated 1,000 times.

Habitat Use

The five variables I selected to examine the relationship between BRS distribution and habitat were depth, aspect, bank slope, distance to rapid, and bed shear stress (Table 1). Because duration and intensity of light varies hourly and seasonally, single measurements could be misleading. Therefore, I selected water depth (shallow or deep transect), aspect, and bank slope as habitat predictors that may influence the amount of direct sunlight exposure (Table 1). Also, because the BRS is known to occur near rapids (U.S. Fish and Wildlife Service 1992, Hershler et al. 1994), I measured the distance from each section to the nearest upstream rapid using GIS. Rapid locations were determined by Welcker et al. (2009a). Bank slope, distance to rapid, and bed shear stress were evaluated for both occurrence and abundance, whereas depth and aspect were analyzed for abundance only.

BRS are associated with cobbles that are free of fine sediment (U.S. Fish and Wildlife Service 1992, Hershler et al. 1994). The majority of sediment transport and deposition in rivers occurs during periodic high runoff events (Leopold et al. 1964, Goodwin 2004), which in the Snake River typically occur once every 1-2 years (Welcker et al. 2009b). High runoff events are characterized by high levels of bed shear stress–a measure of the force applied to the bed of a river by moving water–and influence sediment transport and particle size. Recent studies have demonstrated correlations between freshwater mussel distribution and bed shear stress (Hardison and Layzer 2001, Howard and Cuffey 2003, Morales et al. 2006, Gangloff and Feminella 2007, Newton et al. 2008). Borden and Conner (2009a and 2009b) developed a hydrodynamic model from bathymetry, photogrammetry, surveying of temporary benchmarks, and numerous stage (water surface elevation) recorders. I compared BRS occurrence and abundance to bed shear stress values reported by Borden and Conner at three discharge levels for both reaches in the study area. Discharge levels used in the analysis were 99, 311, and 487 m^3/s, measured in the upper reach. Discharge for the lower reach was 28 m^3/s greater than the upper reach due to discharge from the Malad River (Borden and Conner 2009a, 2009b; Figure 2). The discharge values used for this analysis represent the minimum, median, and maximum discharge values measured between 2003 and 2008, when Idaho Power Company monitored water surface elevation in the study area. The range of discharges recorded during my field study were representative of more than 90% of the range of measured daily average discharge since 1909 (Figure 4).

Each habitat variable was compared to occurrence and/or abundance of snails at one of three spatial scales: transect, bank-section, or section because each variable was scale-specific (e.g., depth differed between adjacent transects while distance from rapid applied to the entire section; Table 1). The sections I sampled in 2005 were revisited in 2007. To avoid double weighting these sections, I excluded the data from 2007 in my analysis of habitat relationships with the exception of the 2007 bed shear stress comparison following a high water year in 2006. Data from 2005 and 2006 were pooled for all habitat analyses except for the bed shear stress comparisons. I employed a Mann-Whitney rank sum test (Zar 1999) to test for differences in abundance in deep versus shallow transects (n=252) and north-facing versus south-facing aspects. For the comparison of BRS abundance and bank slope, distance from rapid and bed shear stress, I used Spearman's rank correlation coefficient (Zar 1999). Because occurrence of BRS is binomially distributed (i.e., snails were either present or absent), I used logistic regression to test whether habitat variables could predict occurrence. I used SigmaPlot 11.0 (SigmaPlot 2008) to perform the statistical analyses described above.

Long-Term Monitoring

My goal in developing a monitoring protocol for BRS was to select a response variable that accurately reflected abundance of the snail, and was sensitive enough to detect reductions in abundance. Because it was not feasible to census the entire population of BRS in the study area, a suitable sampling method was required to estimate abundance of the snails. Estimating abundance of the species is problematic because it involves assumptions about rates of detection, BRS abundance on substrate other than cobbles, estimates of available habitat, and habitat suitability based on depth. I chose to use the total count of BRS in the study area, which consisted of a single annual value, as an index of BRS abundance. This approach, which is commonly used in animal population surveys (Thomas 1996, Gibbs and Melvin 1997, Fewster et al. 2000), allowed for simple linear regression analysis of the annual abundance indices.

I used SAS 9.1 (SAS 2008) to estimate the statistical power to detect five levels of BRS population decline: 50%, 35%, 25%, 20%, and 10% reductions in BRS population size over a 5-year period. This exercise was based on the distribution of BRS counts I recorded during my field observation of 15,000 cobbles. To simulate the decline in BRS abundance, I decreased the distribution by a fixed amount each year so that the decline would be linear and equal to the effect size of interest. For example, the 25% decline was simulated by reducing the abundance distribution by 5% each year for five years (Al-Chokhachy et al. 2009). I employed these distributions in a Monte Carlo simulation (Manly 2007), and analyzed the results using PROC REG (SAS 2008) to determine the proportion of time that the null hypothesis of no trend was rejected (Hatfield et al. 1996). I used a one-tailed test with α of 0.1 instead of 0.05 to protect against a Type II error (Peterman 1990, Taylor and Gerrodette 1993, Di Stefano 2003, Fore and Clark 2005). For each bootstrap, I simulated five years of sampling for 1,000 iterations. The mean proportion of simulations for which the null hypothesis was rejected provided the statistical power. To calculate the minimum power for each Monte Carlo simulation, I subtracted the standard deviation from the mean power.

RESULTS

Spatial Distribution

BRS were uncommon on individual cobbles but the rate of occurrence increased with larger spatial scales. The species occupied 6 to 13% of cobbles I sampled (Table 2). The occupancy rate for bank-sections (n=40 cobbles) ranged from 46 to 69%, while occupancy for sections (n=80 cobbles) was 64 to 86% (Table 2). Of the 378 bank-sections sampled, only three had BRS occurring on more than half of the cobbles sampled. In most sections (n=338), less than 25% of the cobbles were occupied. Average abundance for cobbles ranged from 0.2 to 0.6 BRS per cobble, while the coefficient of variation (CV) for cobbles ranged from 4.0 to 7.2 (see Table 3 and 4 for upper and lower reaches, respectively). The average abundance for sections ranged from 12.2 to 50.9 BRS per section, while the CV for sections ranged from 0.9 to 1.7. Frequency distributions of BRS abundance data were right-skewed for all four spatial scales (Figures 5 and 6). VMR values were greater than 1 at all four spatial scales I examined (Tables 3 and 4), indicating that BRS were dispersed contagiously throughout the study area.

BRS abundance in 2005 and 2007 was significantly correlated at all three spatial scales I tested (Tables 5 and 6). All correlations were positive, indicating that sections containing large numbers of BRS in 2005 tended to have large numbers of BRS in 2007. Correlation coefficients were greater for the upper reach, compared to the lower reach, despite a larger sample size in the lower reach.

BRS occupied three or fewer cobbles in samples at 43% of bank-sections while 21% of occupied bank-sections consisted of a single occupied cobble. The bootstrap analysis revealed that when BRS occurred at a frequency of 0.025 occurrences per cobble (i.e., one of 40 cobbles was occupied), there was a 39% chance of failing to detect BRS in bank sections when 40 cobbles were sampled. This result indicates that I may have failed to detect BRS in some bank-sections when they actually occurred there. For a frequency of occurrence of 0.025 per cobble, increasing sample size from 40 to 100 cobbles per bank-section would reduce the Type II error rate associated with analyzing trends in the BRS population to 0.08 (Figure 7). When BRS occurrence rates were higher (e.g., 0.05 and 0.075 occurrences per cobble), the simulated Type II error rate was also reduced by increasing the sample size to 100 cobbles (Figure 7).

Habitat Use

BRS were more abundant in deep transects compared to shallow transects ($p=0.014$; $U=25{,}911$; $n=252$; Figure 8) and were more abundant in the north-facing aspects compared to south-facing aspects ($p=0.03$; $U=14{,}260$; $n=252$; Figure 9). Abundance showed a weak negative correlation with distance from the nearest upstream rapid ($p = 0.014$; $r_s = -0.219$; $n=126$; Figure 10), whereas no significant correlation was found for snail occurrence. There was a weak, negative correlation between bank slope and BRS abundance ($p=0.03$; $r_s = -0.137$; $n=252$), but bank slope was not correlated with occurrence.

BRS abundance was positively correlated with bed shear stress for all discharge-year combinations except for the 99 m^3/s discharge in 2006 (Table 7, Figures 11-13). The correlation was highest for bed shear stress at 311 and 487 m^3/s discharges in 2007 (Table 7), which followed a high-water event in the spring of 2006. Occurrence of BRS was correlated with bed shear stress for all three discharge levels in 2007, but not for any of the discharge levels in 2005 or 2006 (Table 7).

Long-Term Monitoring

The power analysis for BRS abundance monitoring suggested a 50% decline in abundance could be detected with a statistical power of at least 0.8 with approximately 2,000 cobbles annually over a 5-year period. Detecting a 10% decline with the same statistical power would require sampling 60,000 cobbles annually (Table 8, Figure 14). Effort required to obtain statistical power greater than 0.8 increased in a near-linear manner for modeled population declines of 50% to 25%; however, effort increased nearly exponentially for modeled population declines of 25% to 10% (Table 8).

DISCUSSION

Spatial Distribution

Historically, BRS have been viewed as spring obligates with a few small populations present in the Snake River (U.S. Fish and Wildlife Service 1992, Hershler et al. 1994). However, my study shows that the snails are more abundant and widely distributed in the Snake River than previously known. Moreover, my results suggest that although the snails occur in relatively low densities in the Snake River, they are likely more abundant in the river than in all spring populations combined because of the relatively large amount of habitat in the river (see also Bean and Van Winkle 2009, Bates et al. 2009, Richards and Arrington 2009). Using the data I collected for this thesis, Richards et al. (2009a) estimated BRS abundance in the study area as 6.1-25.7 million (95% confidence interval), while Bean and Van Winkle (2009) estimated abundance as 2-26.8 million. BRS abundance in all of the springs combined was estimated at less than 1 million individuals (Bates et al. 2009, Richards and Arrington 2009). Future conservation efforts for the species should consider Snake River populations of the BRS in addition to spring populations.

While this study demonstrates BRS are more abundant than predicted within their known range, due to the small range occupied by these animals the species is still quite rare (Rabinowitz 1981). This study has changed the categorization of rarity used to describe the species according to Rabinowitz's framework. The species is locally abundant and less of a habitat specialist within the study area than previous data suggested. This study did not assess the third part of Rabinowitz's framework, which is range. Surveys for the species outside of their known range would be necessary to determine the true extent of the species' range.

Results of the bootstrap analysis suggested I may have failed to detect BRS in sections I sampled. Increasing the sample size at the bank-section scale from 40 to 100 cobbles would reduce the probability of committing a Type II error when analyzing trends in BRS. For locations with BRS occurrence rates of less than 0.025, a larger sample size would be necessary to detect the species when they occur. Lower occurrence rates may occur in marginal habitat or in habitats currently thought to be outside the species' range.

I restricted my sampling area to previously described range boundaries for BRS (U S. Fish and Wildlife Service 1992, Hershler et al. 1994), and as a result additional areas occupied by the snail may remain undocumented. In a recent study, Bates et al. (2009) reported a small BRS population at a new location upstream of the previously defined range. I did not sample the reservoir between the two reaches in my study area because BRS have not been observed in reservoirs (U.S. Fish and Wildlife Service 1992, Hershler et al. 1994, Idaho Power Company, unpublished data). However, given the results of my bootstrap analysis, previous sampling efforts in the reservoir were likely insufficient to detect BRS if they occurred in the reservoir. Three other snails that occur in the Snake River and were originally thought to be absent in reservoirs have since been collected in reservoir habitats (U.S. Fish and Wildlife Service 2007, Idaho Power Company unpubl. data, R. Newman [Bureau of Reclamation] pers. comm.). These snails include the Utah valvata (*Valvata utahensis*), the Jackson Lake springsnail (*Pyrgulopsis robusta*; previously *Pyrgulopsis idahoensis*), and the Snake River physa (*Physa natricina*), all of which are relatively tolerant of the fine substrates typical of reservoir habitats (Hershler et al. 1994, U.S. Fish and Wildlife Service 2007, R. Newman [Bureau of Reclamation] pers. comm., Idaho Power Company, unpubl. data). By contrast, the BRS is not associated with fine sediments, so it remains a distinct possibility that they do not occur in reservoirs. Nevertheless, more intensive studies outside of the known range of BRS, particularly in the Snake River, are warranted to determine the full extent of the species' range.

The cobble count method I used is a flexible and cost-effective technique for sampling BRS. Prior field studies of the BRS used a suction dredge collection method to vacuum a 0.25 m^2 area (Stephenson and Bean 2003, Stephenson et al. 2004). However, given the dispersion pattern that BRS exhibit, the cobble count method used in my study is a more effective approach. Smaller samples spread across a larger area are more likely to encounter a species exhibiting contagious dispersion compared to larger, localized samples (see Richards et al. 2005). The cobble count method is also more cost efficient than suction dredging (personal obs.). By reducing the time required to collect each sample, I was able to collect data across a large area, including habitat that was previously thought to be unsuitable (U.S. Fish and Wildlife Service 1992, Hershler et al. 1994). Intensive sampling within an isolated area, accomplished using tools like a suction dredge, may not produce data sufficient to detect changes in BRS population trends.

In addition to monitoring the species' abundance trends, the cobble count method has proven useful for determining the occurrence or abundance of BRS in areas affected by anthropogenic activities such as construction activities (e.g., boat ramps, bank stabilization, bridge construction, etc.). Variations of the cobble count method have been used to evaluate potential impacts to snails of various anthropogenic activities in both river and spring environments (Dillon 2006, Stephenson 2006, Bates 2009, Bean and Stephenson 2009). With better data available with which to gauge impacts of human activities, future impacts to the species can be reduced through careful management.

The new information about the spatial ecology of the BRS provided here suggests the snail may exist in two large populations in the Snake River divided by Bliss Reservoir. This conclusion is supported by Liu and Hershler (2009) who examined 11 microsatellite loci from BRS throughout the species' range. They found that the genetic variation among snails collected within and between these two reaches was not significantly different, whereas many spring and tributary populations of the snail exhibited significant amounts of genetic differentiation, including some populations separated by less than 300 m. Results of the genetic analysis suggest gene flow occurs within and between the two river reaches, or that the dam, which was built in 1948, has not been in place long enough to result in genetic differentiation between the two reaches (Liu and Hershler 2009). The results reported by Liu and Hershler are parallel to microsatellite studies of more mobile species that have fragmented habitats due to dams (Burridge and Gold 2003, Kelly and Rhymer 2005, Reid et al. 2008), yet other studies have shown genetic differentiation due to dams (Laroche et al. 1999, Stamford and Taylor 2005). The ability to differentiate between groups using microsatellite DNA is influenced by heterozygosity and population size (Hedrick 1999). Populations with high levels of heterozygosity and large population size are less likely to show statistically significant differentiation with microsatellite DNA. The BRS population(s) in the Snake River are much larger and more heterozygous compared to populations in the springs and tributaries (Bates et al. 2009, Bean and Van Winkle 2009, Liu and Hershler 2009, Richards and Arrington 2009). Further research analyzing BRS population size and heterozygosity in the Snake River is warranted to determine the genetic structuring of the species between these two river reaches.

Habitat Use

The apparent preference for the north-facing side of the river is counterintuitive for a grazing invertebrate, as many laboratory studies have shown that grazing snail abundance and growth rates are positively correlated to light intensity and associated primary productivity (e.g., DeNicola and McIntire 1991, Hill et al. 1995). David Richards ([Econalysts, Inc.] pers. comm.) conducted periphyton sampling of BRS habitat at Banbury Springs as well as gut analysis of individual BRS. Richards found that BRS selected the diatom *Cocconeis* sp. disproportionally to other periphyton taxa. Many species of *Cocconeis* are known to be heterotrophic; they sequester carbon through pores in their cell walls in lieu of photosynthesis (Mohapatra 2008). It is possible that the BRS benefit from reduced interspecific competition in shaded habitats, or that *Cocconeis* grows more readily on the bottoms of cobbles where BRS can find refuge from high water velocity or predation. Further research addressing the species' diet, interspecific competition, and light measurements would be necessary to provide support for this hypothesis.

BRS abundance was correlated with the deeper transect and north-facing aspects, both of which may be indicative of direct sunlight avoidance by the snails. However, Bean and Van Winkle (2009) conducted deepwater SCUBA surveys and found that BRS abundance was negatively correlated with water depth. The pattern observed during SCUBA surveys may have been related to the shape and embeddedness of cobbles in deeper habitat. Substrate in deeper habitats tended to be more rounded and fine sediments were more common, both of which resulted in less interstitial habitat for BRS. Bank-side habitat in the Snake River having depths less than 0.5 m is subject to daily and seasonal water surface level changes. Higher observed abundance of BRS in deeper transects could be a result of more stable habitat at depth greater than 0.5 m.

Rivers are dynamic in nature. High flow events can transport bed materials and alter habitat, whereas prolonged droughts and flood control can result in fine sediment deposition (Leopold et al. 1964). The correlation between bed shear and BRS abundance and occurrence increased slightly following a high water event in 2006 (Table 6). Because the BRS inhabits the underside of unembedded cobbles, high water events could play a role in removing fine substrate and improving habitat for BRS. The BRS evolved in a river system that likely underwent frequent scour events, which would reduce accumulation of fine sediments. Long-term trend data comparing BRS abundance and annual discharge patterns (e.g., prolonged droughts or high water events) would be necessary to determine if flood events play a role in BRS abundance.

BRS abundance declined as distance from rapid and bank slope increased, albeit the statistical relationship was weak. Prior to this study, most of the known BRS colonies occurred in areas having steep banks near rapids. Based on their observations, previous investigators concluded these habitat characteristics were important for BRS. While indirectly true, such habitat characteristics are surrogates for light and water velocity. My study took a more direct approach at measuring these two variables and identified patterns that contradict conclusions made in previous studies. In the future, direct measures of light intensity and water velocity, using metrics such as insolation or Solar Pathfinder^{tm} data for light and micro-scale water velocity measurements, may identify potential thresholds associated with these habitat characteristics.

My plan to develop a habitat-based model to predict the occurrence of the species in the Snake River was based on the assumption that the BRS were distributed in a small number of isolated patches of suitable habitat, and selected habitat variables that described the known locations where BRS occurred. My intent was to base the predictive model on features of the landscape that could be detected remotely—either with GIS or a hydrodynamic model similar to the methods Strayer et al. (2006) used for lotic mussel populations. Unfortunately, field observations revealed previous conceptions of BRS as existing in isolated patches of habitat were incorrect and the predictive power of the model I developed was low. The model's low predictive power may be a result of the prevalence (i.e., the proportion of samples in which the focal species occurs) of BRS in the study area. Presence-absence models have poor predictive power when prevalence is between 0.4-0.6 (Fielding and Bell 1997, Manel et al. 2001). The occupancy rate for bank-sections in my study ranged from 0.46-0.69. The data I collected from this study suggest the spatial scale I used to evaluate habitat use of the BRS was inappropriate given their widespread distribution patterns and prevalence in the Snake River. Finer-scale metrics such as food preference (diatoms), associated macroinvertebrates, water velocity for individual cobbles, light intensity, and other unknown factors might be better predictors for the species. For example, Ryan Newman ([Bureau of Reclamation] pers. comm.) found that the Snake River physa snail (*Physa natricina*) was associated with the more common snail *Ferrissia rivularis* and the leech *Helobdella stagnalis*. Haynes and Taylor (1984) found that the Hydrobiid snail *Potamopyrgus jenkinsi* was attracted to some types of algae but indifferent to or even repelled by other species of algae. Future research into habitat use of the BRS should focus on micro-scale habitat metrics given the distribution patterns I observed.

Long-Term Monitoring

The power analysis I conducted suggests that BRS abundance can be monitored effectively throughout the study area. Data collection in the study area is relatively costly; the area is fairly remote and is accessible only by boat. A 3-person crew could survey 400-600 cobbles in 10 hours. Therefore, an effect size of a 20% decline in BRS would require approximately 75 person-days. A goal of detecting a 25% decline in BRS abundance would cut effort to about 45 person-days, while a 10% effect size would increase that effort tenfold, and would cover 60% of the shoreline in the study area. Increasing effect size to 35% or 50% may not be conservative enough to protect BRS. However, according to the BRS population estimates provided by Bean and Van Winkle (2009) and Richards et al. (2009a), if the population declined by 50%, over a million BRS would remain in the study area. Based on cost-benefit criteria alone, I recommend setting the target effect size at 25% reduction in abundance and sampling 6,000 cobbles annually.

Wilcove and Terbough (1984) described three patterns of species' decline: 1) declining abundance at the center of the geographic range resulting in little contraction in the size of the range; 2) range contractions with little change in abundance at the center; and 3) a combination of declines in abundance at the center and range contraction. It is therefore necessary to monitor for both declines in abundance and range when studying at-risk species. If BRS were to disappear from sections in the middle of their range, populations could become isolated, resulting in reduced gene flow and increased extinction risk. Therefore, I suggest monitoring at many sites across the species' range as opposed to more intensive effort in a small number of locations. For example, collecting 200 cobbles at each of 30 bank-sections (6,000 cobbles) spread throughout the study area would provide adequate power to detect a 25% decline over a five-year period while representing 10% of the shoreline in the study area. This approach would simultaneously meet the objectives of monitoring BRS abundance and extant range.

An effective abundance monitoring plan must strike a balance between precision, effort (cost), and what is biologically meaningful to the organism of interest. In some cases, the desire to detect small changes in population size may result in exorbitant costs. Additionally, if managers wish to detect very small declines in the abundance of a species, this could result in false alarms should population abundance dip slightly, even though the decline is within normal bounds for the species of concern. Conservation biologists must therefore consider a species' biology and extinction risk when determining an appropriate effect size. When considering the risks associated with population decline, researchers should consider cycles in population abundance typical of the species, minimum viable population size, what genetic impacts could occur due to reduced population size, and what factor(s) contributed to the decline.

Effective population monitoring spans both time and space (Elliott 1990). Field data collection for my study spanned three years. A three-year snapshot is too short to make any assumptions about long-term oscillations in abundance (Elliott 1990, Gibbs et al. 1999, Al-Chokhachy et al. 2009); however, the data presented here span a large spatial scale and are a good starting point to assess temporal trends. Additionally, BRS have a lifespan of approximately one year (Hershler et al. 1994); therefore, my study encompassed three generations in the Snake River. The long-term monitoring methods described here require five years of data collection in order to provide sufficient statistical power to detect a decline of the population index data. If it was known BRS were declining in abundance or distribution, this would not be an appropriate approach. However, the limited data that are available suggest BRS abundance and distribution are relatively stable across their known range (Richards et al. 2006, Richards et al. 2009b). Therefore, the monitoring program I have proposed is sufficient to gather robust, long-term abundance trend data necessary to select an appropriate effect size for this species.

Detection probabilities have not been estimated for BRS. Counts must be collected in a consistent manner over time to reduce variability in detection probabilities between years. Staff training and overlap from previous years are important factors to consider when conducting long-term monitoring programs. Detection probabilities could be estimated to compensate for variable rates of detection (Schmidt 2004); however, some covariates are unknown or difficult to measure. While an abundance index for BRS may be impacted by imperfect detection (Anderson 2001), an index makes far fewer assumptions compared to population estimates, as discussed earlier. Traditional methods used to estimate detections rates, such as mark-recapture and double sampling are not feasible for BRS. A population index is a straight-forward approach that will allow managers to evaluate trends in BRS abundance over time.

CONCLUSION

While the BRS is still considered a rare species, my work demonstrates that the snail is more abundant and widely distributed in the study area than previously known. The species exhibited contagious dispersion and likely exists in one or two populations in the study area (separated by Bliss Reservoir). The hierarchical spatial design of this study was useful in determining the dispersion and distribution patterns of the BRS.

The species was associated with north-facing aspects, high bed shear stress, and were spatially and temporally correlated between sites visited in 2005 and 2007. The snail's association with shaded aspects is counterintuitive for a grazing species and warrants further study. Predictive power was low for the relatively broad-scale habitat variables I used to model the relationship between BRS and stream habitat. My results suggest that future studies should focus on microhabitat variables that can be measured directly, such as light intensity or micro-scale water velocity near cobbles, variables that could affect BRS distribution at the microhabitat scale.

The insight gained from this research provides information valuable for monitoring and assessing future risks for BRS. The cobble count method was an effective method for sampling this species given the snails' dispersion pattern. The bootstrap simulations revealed that I may have failed to detect the species when their occurrence rate was 0.025. The sample size of future studies should be designed with occurrence rates in mind. The power analysis I used to evaluate the ability of long-term sampling to detect population-level declines in BRS demonstrated that the species can be monitored in the study area with sufficient statistical power to detect 10-50% declines in abundance over a five-year period.

When evaluating the status and population trends of rare species, researchers should evaluate the species' habitat use as well as their distribution and dispersion patterns. Once these patterns are understood, monitoring studies can be designed with appropriate spatial and temporal scales. A population index provides a metric specific to the appropriate spatial and temporal scale for each species, and does not rely on assumptions regarding habitat suitability or detection rates. This approach can be adapted for a wide range of organisms, assuming statistical power is sufficient to detect a decline in the population index.

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Table 1 Habitat variables examined for their association with occurrence and abundance of Bliss Rapids snails in the Snake River, Idaho.

Habitat Variable	Description	Measurement Method	Spatial Scale	Variable Type
Depth	Water depth; either shallow (0-0.5 m) or deep (0.5-1.5 m)	Observed	Transect	Categorical
Aspect	North-facing (NW, N, NE, & E) and south-facing (SE, S, SW, W)	GIS	Bank-section	Categorical
Bank Slope	Mean bank slope within 20 meters of water line (degrees)	GIS	Bank-section	Continuous
Distance to rapid	Distance downstream of nearest rapid (m)	GIS	Section	Continuous
Bed shear stress	Force applied to the bed of the river (N/m^2)	Model	Section	Continuous

Table 2 Bliss Rapids snail occurrence by section, bank-section and individual cobbles for each year and river reach from the cobble count study in the Snake River, Idaho. Values represent number of occupied sites for each scale. Percentages are in parentheses. Data for 2005 and 2007 were collected from the same locations.

Upper Reach	Section	Bank-Section	Transect	Cobbles
2005	17 (81)	25 (60)	41(50)	214 (13)
2006	14 (67)	25 (55)	36(43)	109 (6)
2007	17 (81)	29 (69)	45(54)	195 (12)
Lower Reach	**Section**	**Bank-Section**	**Transect**	**Cobbles**
2005	27 (64)	37 (46)	54(32)	171 (5)
2006	31 (74)	50 (59)	75(45)	221 (7)
2007	36 (86)	56 (67)	85(51)	321 (10)

Table 3 Characteristics of Bliss Rapids snail abundance at four spatial scales for 2005, 2006, and 2007 in the upper reach of the study area in the Snake River, Idaho. SD = standard deviation, CV = coefficient of variation, VMR = variance-to-mean-ratio and n = sample size.

			2005			
	Mean	**SD**	**CV**	**Variance**	**VMR**	**n**
Cobble	0.6	2.7	4.5	7.3	12.2	1680
Transect	12.7	25.7	2.0	657.9	51.8	84
Bank-Section	25.5	47.2	1.9	2227.8	87.4	42
Section	50.9	73.0	1.4	5329.0	104.7	21
			2006			
	Mean	**SD**	**CV**	**Variance**	**VMR**	**n**
Cobble	0.2	0.8	4.0	0.6	3.0	1680
Transect	3.0	6.6	2.2	43.6	14.5	84
Bank-Section	6.1	11.2	1.8	125.4	20.6	42
Section	12.2	17.1	1.4	292.4	24.0	21
			2007			
	Mean	**SD**	**CV**	**Variance**	**VMR**	**n**
Cobble	0.3	1.2	4.0	1.4	4.7	1680
Transect	6	9.2	1.5	84.5	14.1	84
Bank-Section	11.9	15.8	1.3	249.6	21.0	42
Section	23.9	22.5	0.9	506.3	21.2	21

Table 4 Characteristics of Bliss Rapids snail abundance at four spatial scales for 2005, 2006, and 2007 in the lower reach of the study area in the Snake River, Idaho. sd = standard deviation, CV = coefficient of variation, VMR = variance-to-mean-ratio and n = sample size.

			2005			
	Mean	**SD**	**CV**	**Variance**	**VMR**	**n**
Cobble	0.2	1.1	5.5	1.2	6	3340
transect	3.0	7.7	2.6	59.3	19.8	168
Bank-Section	6.2	13	2.1	169.0	27.2	84
Section	12.6	19.1	1.5	364.8	29.0	42
			2006			
	Mean	**SD**	**CV**	**Variance**	**VMR**	**n**
Cobble	0.2	1.0	5.0	1.1	5.5	3280
transect	3.7	7.7	2.2	59.3	16.0	164
Bank-Section	7.4	12.7	1.7	161.3	21.8	84
Section	14.7	20.6	1.4	424.4	28.9	42
			2007			
	Mean	**SD**	**CV**	**Variance**	**VMR**	**n**
Cobble	0.3	1.2	4.8	1.4	4.7	3351
transect	6.3	13.4	2.1	179.6	28.5	168
Bank-Section	10.0	17.9	1.8	320.4	32.0	84
Section	19.9	33.3	1.7	1108.9	55.7	42

Table 5 Correlations of Bliss Rapids snail abundance between 2005 and 2007 at three scales in the upper reach of the Snake River, Idaho. Right and left sides of the river were determined while looking downstream. Note: these tests have not been corrected for multiple statistical comparisons.

Comparison	Spearman Correlation	P-value	Sample Size
2005 vs 2007 section	0.868	0.0000002	21
2005 vs 2007 Right Bank-Section	0.834	0.0000002	21
2005 vs 2007 Left Bank-Section	0.766	0.0000002	21
2005 vs 2007 Left Shallow Transect	0.624	0.003	21
2005 vs 2007 Left Deep Transect	0.670	0.0008	21
2005 vs 2007 Right Shallow Transect	0.794	0.0000002	21
2005 vs 2007 Right Deep Transect	0.694	0.0004	21

Table 6 Correlations of Bliss Rapids snail abundance between 2005 and 2007 at three scales in the lower reach of the Snake River, Idaho. Right and left sides of the river were determined while looking downstream. Note: these tests have not been corrected for multiple statistical comparisons.

Comparison	Spearman Correlation	P-value	Sample Size
2005 vs 2007 section	0.506	0.0007	42
2005 vs 2007 Right Bank-Section	0.599	0.0001	42
2005 vs 2007 Left Bank-Section	0.429	0.005	42
2005 vs 2007 Left Shallow Transect	0.472	0.002	42
2005 vs 2007 Left Deep Transect	0.319	0.04	42
2005 vs 2007 Right Shallow Transect	0.428	0.005	42
2005 vs 2007 Right Deep Transect	0.535	0.0003	42

Table 7 Bliss Rapids snail abundance and occurrence as a function of bead shear stress for three discharge levels in the Snake River, ID. Sample size for all tests was 63. These data represent the section spatial scale.

Discharge (m^3/s)	Spearman Correlation (p value)		
	2005	**2006**	**2007**
99	0.367(<0.01)	0.313(0.133)	0.411(<0.01)
311	0.328(<0.01)	0.363(<0.01)	0.441(<0.01)
487	0.342(<0.01)	0.312(0.013)	0.489(<0.01)
	Logistic Regression p value		
99	0.093	0.204	0.042
311	0.072	0.073	0.011
487	0.057	0.06	0.008

Table 8 Results for linear regression power analysis for the Bliss Rapids snail. Statistical power is reported as the power to detect a decline (one-sided test) for given effect size over a 5-year period. Power was estimated by bootstrapping from 15,000 records with 1000 iterations for each 5-year simulation with α=0.1. See text for details.

Effect Size	Number of Cobbles (samples)	Mean Power	Standard Deviation (SD)	Mean-SD
50% decline	2,000	1.00	0.05	0.95
	1,000	0.93	0.25	0.68
35% Decline	3,000	0.99	0.11	0.88
	2,000	0.89	0.32	0.57
	1,000	0.77	0.42	0.35
25% Decline	6,000	0.97	0.17	0.80
	5,000	0.96	0.21	0.75
	4,000	0.91	0.28	0.63
	2,000	0.75	0.43	0.32
20% Decline	10,000	0.97	0.17	0.80
	8,000	0.94	0.24	0.71
	6,000	0.90	0.30	0.60
	4,000	0.77	0.42	0.35
10% Decline	60,000	0.98	0.15	0.83
	20,000	0.86	0.34	0.52
	10,000	0.62	0.49	0.13

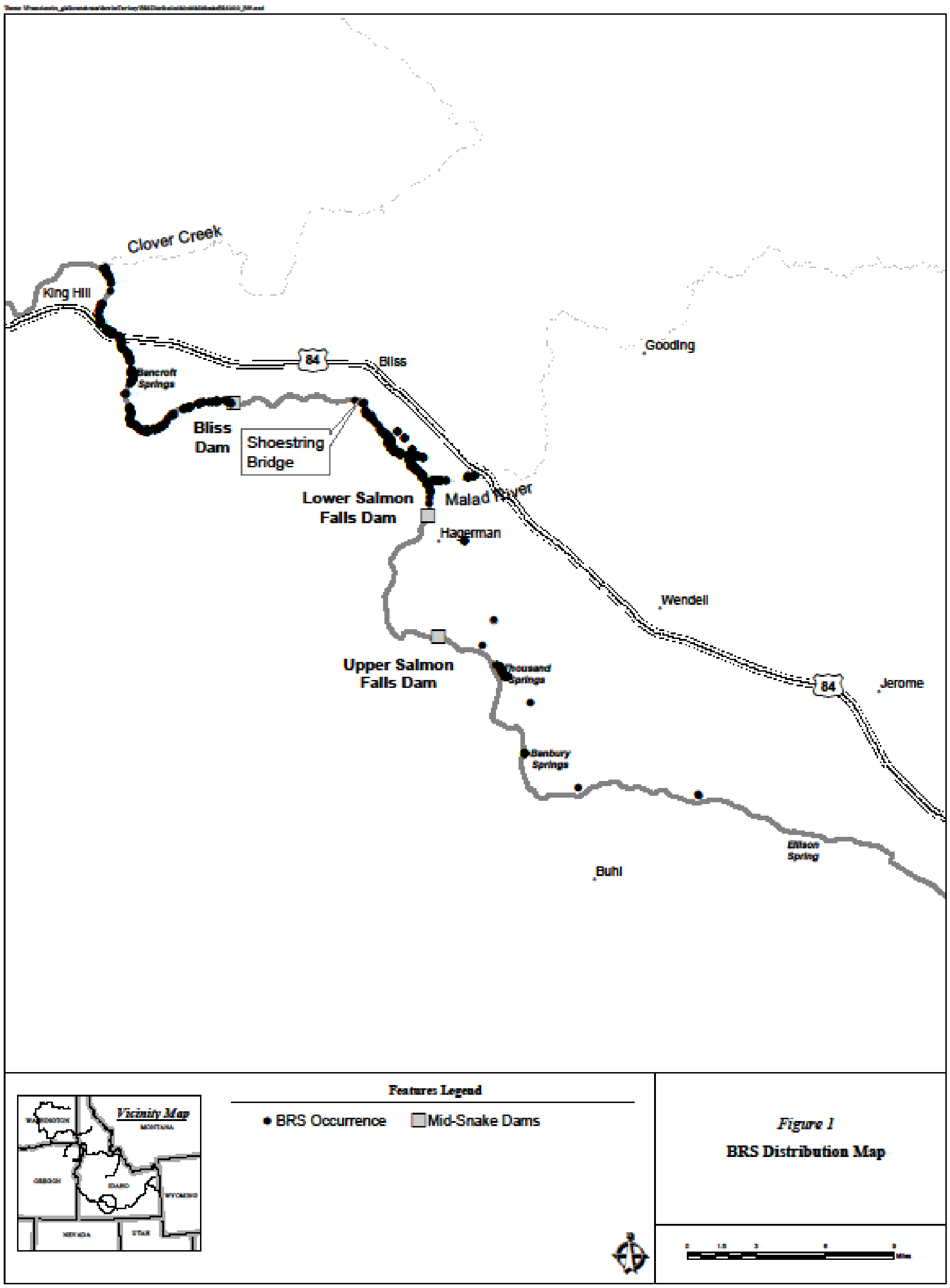

Figure 1 Map of known Bliss Rapids snail distribution in southwestern Idaho.

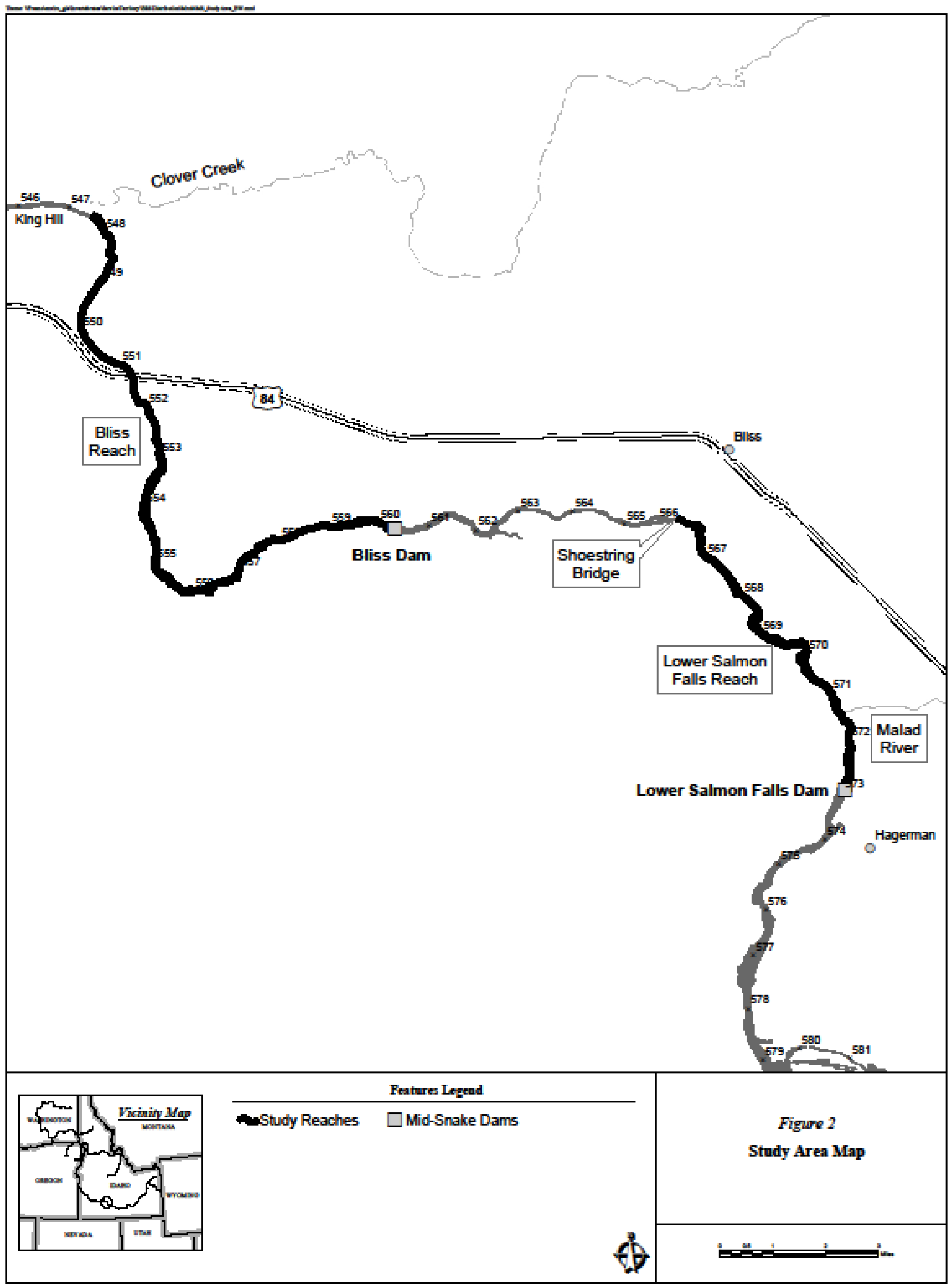

Figure 2 Map of the study area in SW Idaho. The study area is indicated by the two dark black lines.

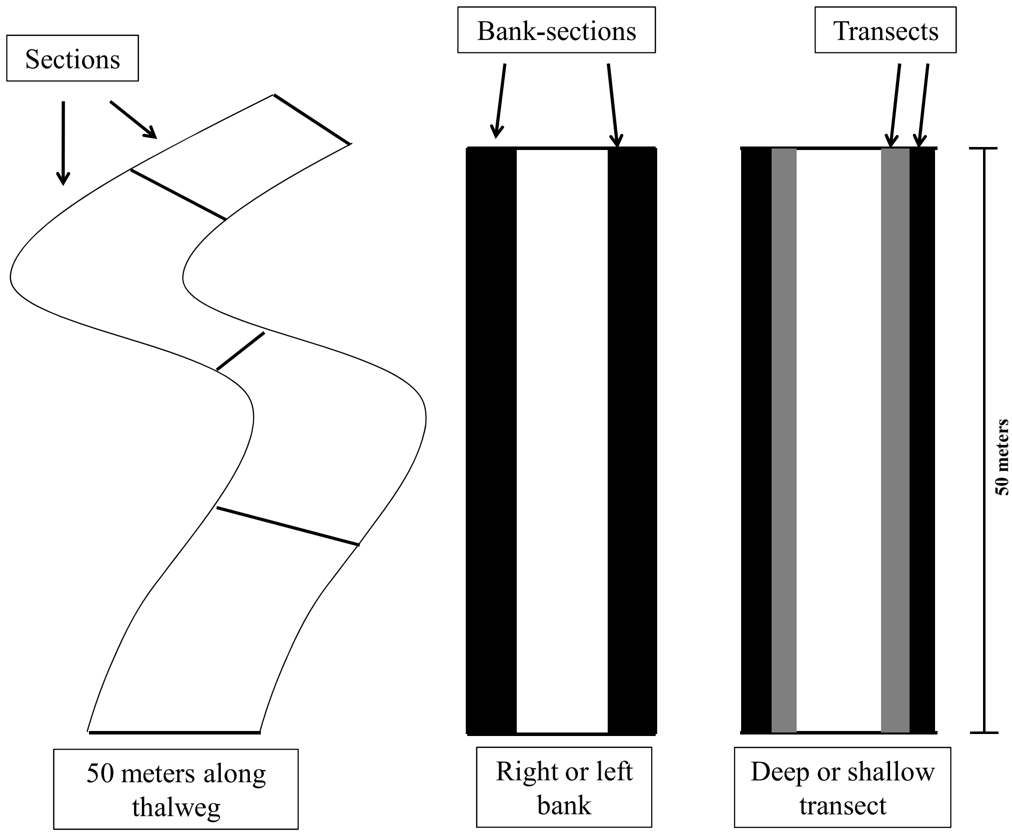

Figure 3 **The nested sample design used in the study. The two river reaches were divided into 50 meter sections measured along the thalweg. Each side of the river (bank-section) was further divided into 0-0.5 and 0.5-1.5 meter depth transects. See text for detailed description.**

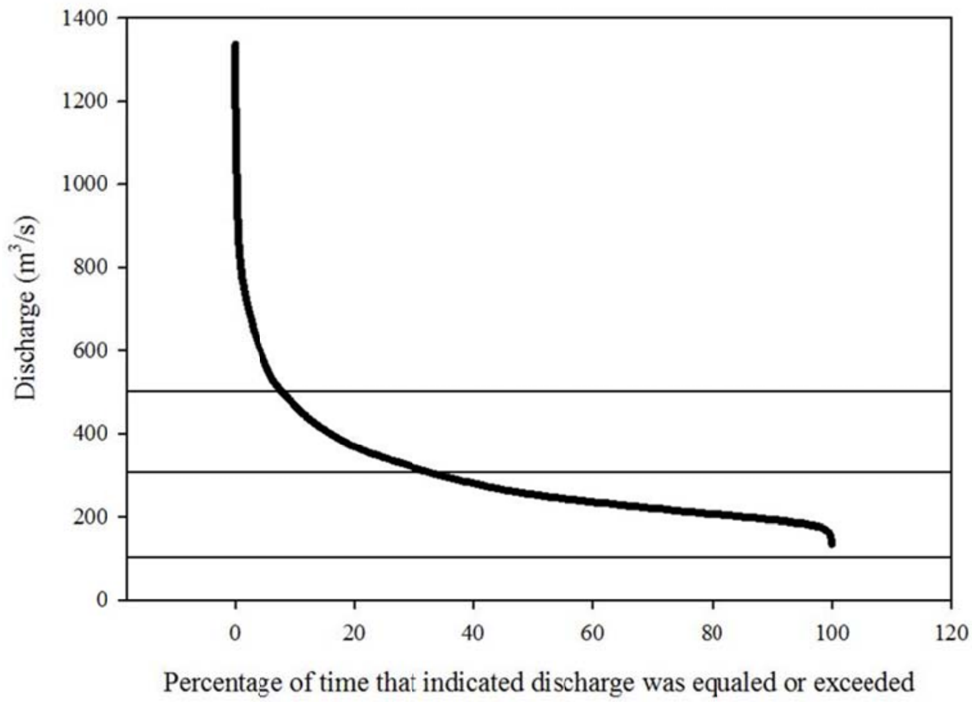

Figure 4 **Flow duration curve from the study area indicating proportion of time a given discharge value has been exceeded. Data are mean daily discharge from 1909-2010 at the King Hill Gage, near the lower end of the study area. Horizontal lines indicate the three discharge levels used in the shear stress analysis.**

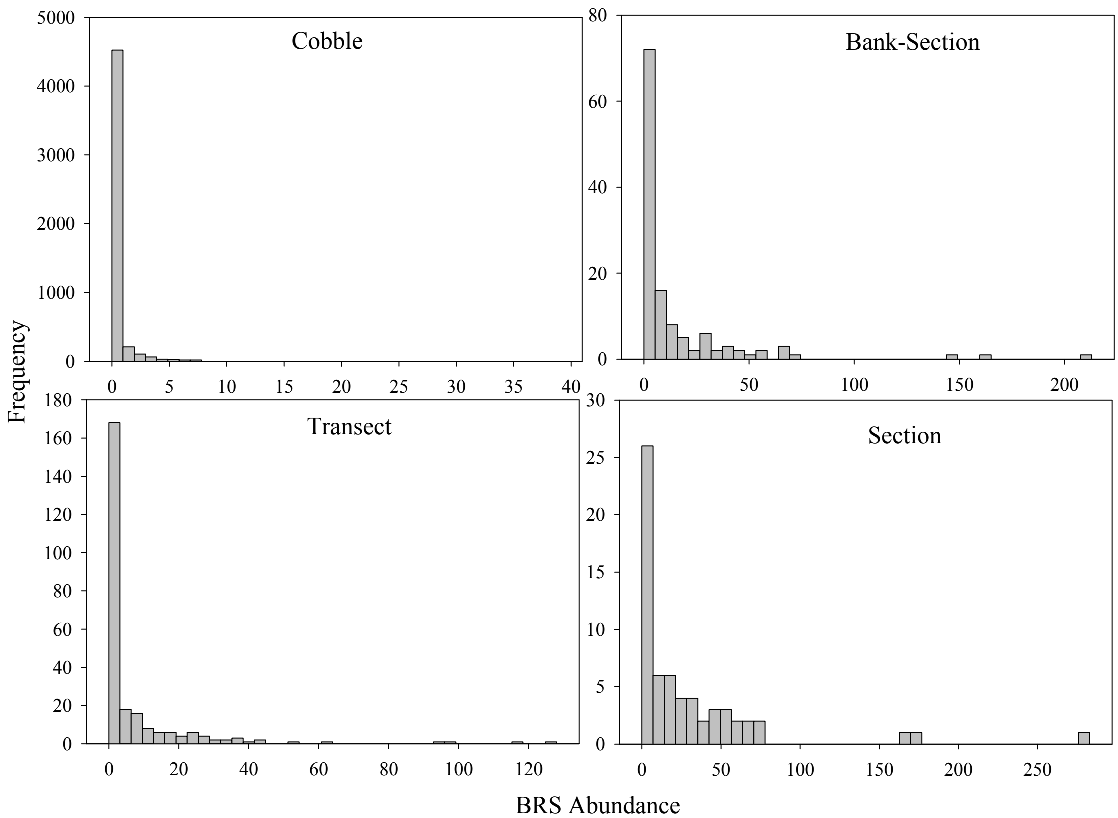

Figure 5 Frequency histograms of Bliss Rapids snail abundance for cobble, transect, bank-section and section spatial scales for the upper reach of the study area in the Snake River, Idaho.

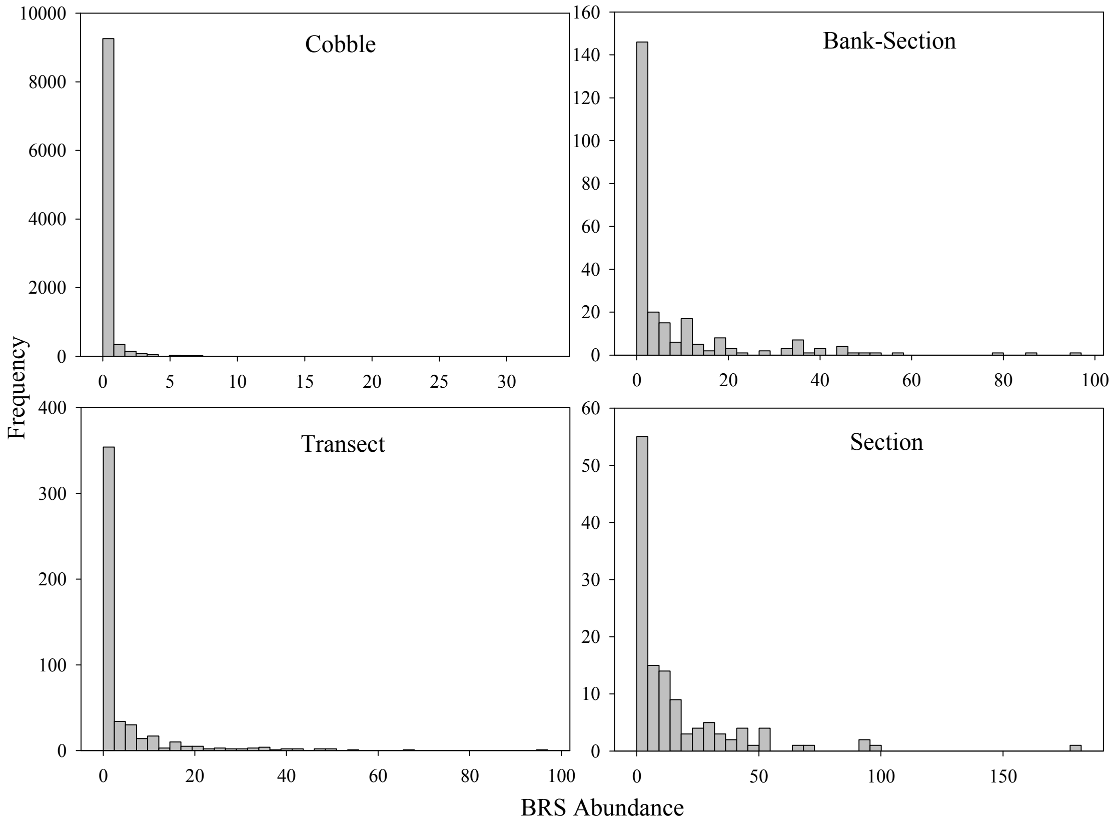

Figure 6 Frequency histograms of Bliss Rapids snail abundance for cobble, transect, bank-section and section spatial scales for the lower reach of the study area in the Snake River, Idaho.

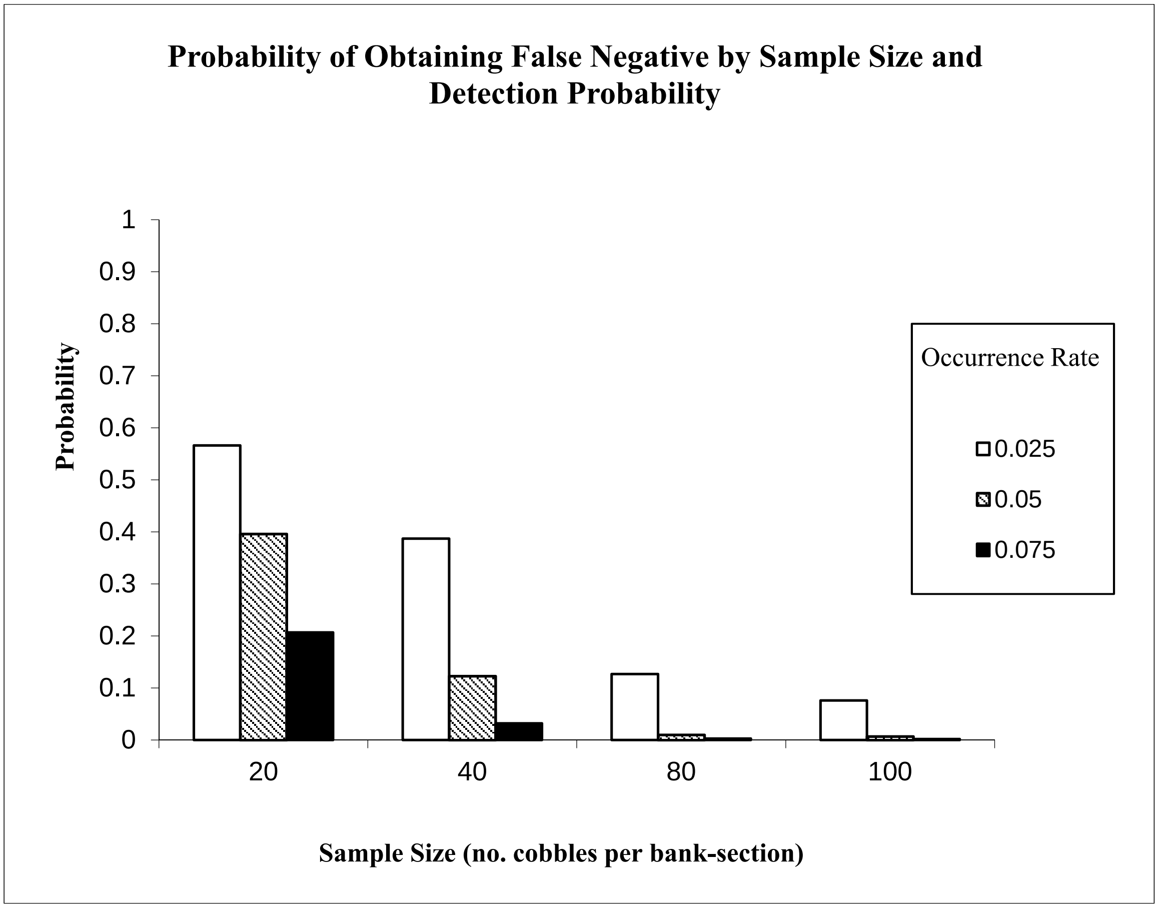

Figure 7 Type II error rates for Bliss Rapids snail occurrence rates of 0.025, 0.05, and 0.075, and four bank-section sample sizes.

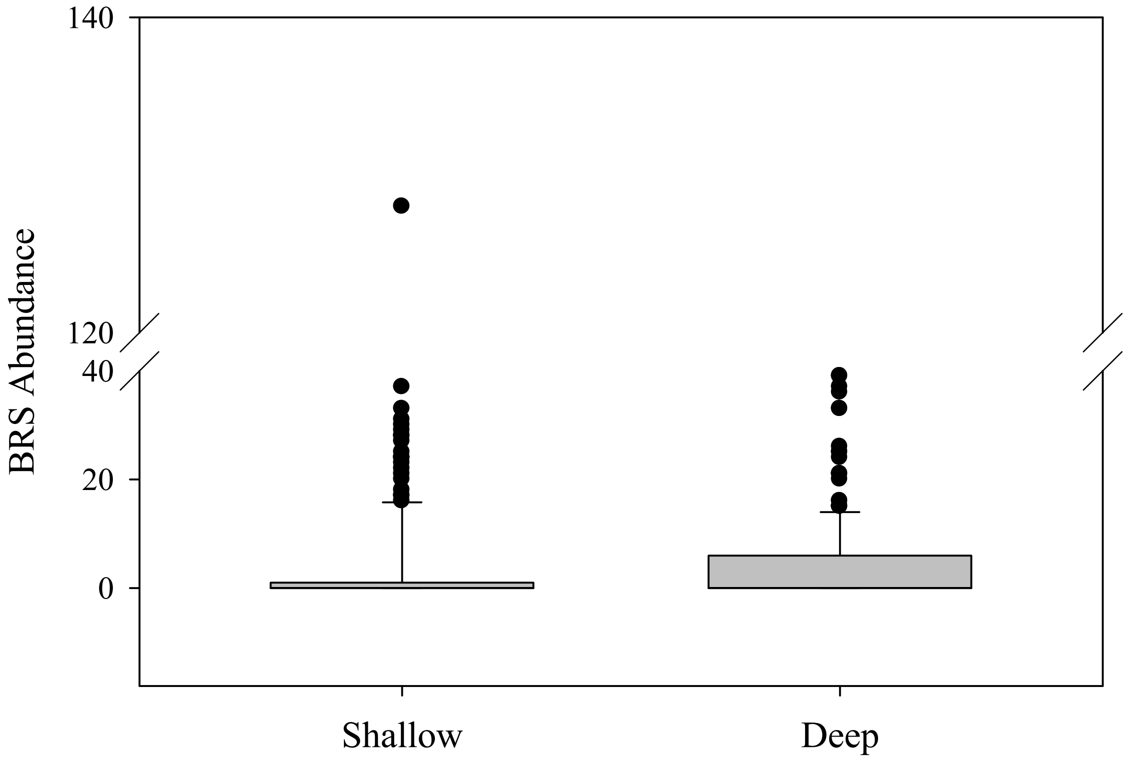

Figure 8 Box plot comparison of Bliss Rapids snail abundance in shallow and deep transects. The shallow transects were 0-0.5 m deep while deep transects were 0.5-1.5 m. The BRS were more abundant in the deep transects (p=0.014; U=25,911; n=252). The center horizontal line within each box represents the median. The 75th and 25th percentile are indicated by the upper and lower limits of each box. The 10th and 90th percentiles are indicated by the whiskers, while the closed circles represent data points outside the 10th and 90th percentiles.**

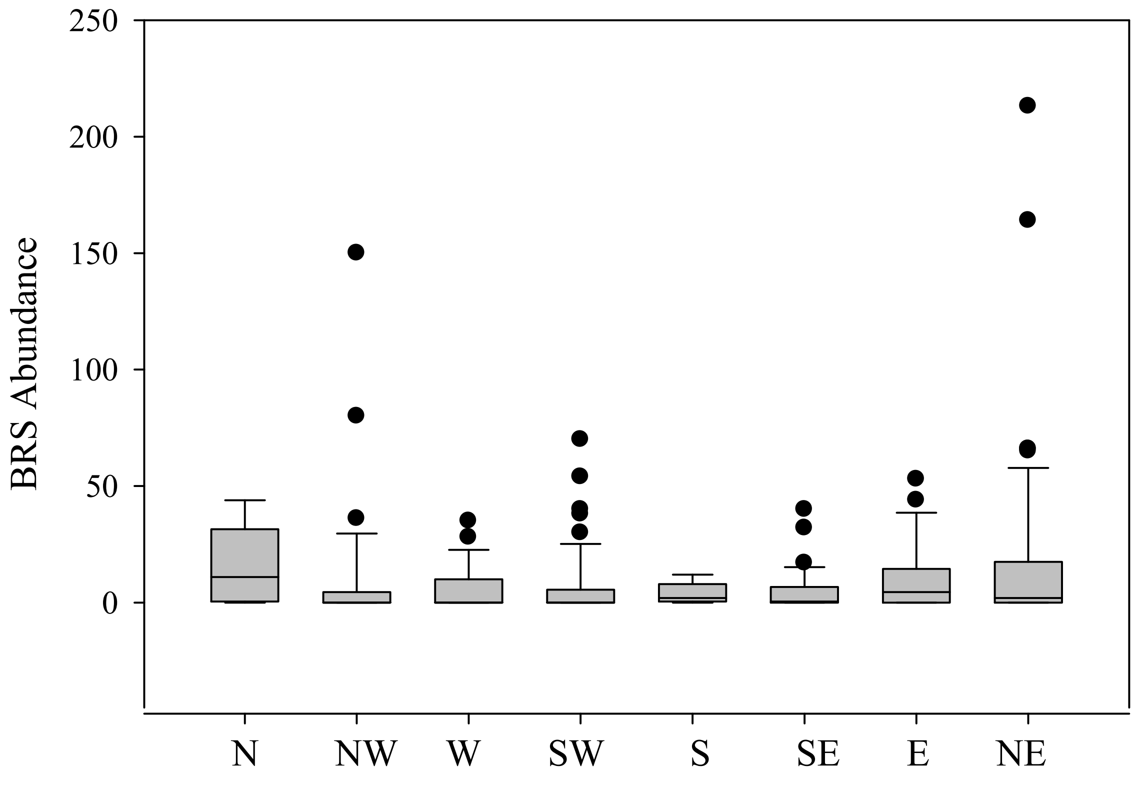

Figure 9 Comparison of Bliss Rapids snail abundance by aspect. Bliss Rapids snails were more abundant in north-facing aspects (N, NW, NE, and E) compared to south-facing aspects (n=252, U=14,260, p=0.03). The center horizontal line within each box represents the median. The 75th and 25th percentile are indicated by the upper and lower limits of each box. The 10th and 90th percentiles are indicated by the whiskers, while the closed circles represent data points outside the 10th and 90th percentiles.

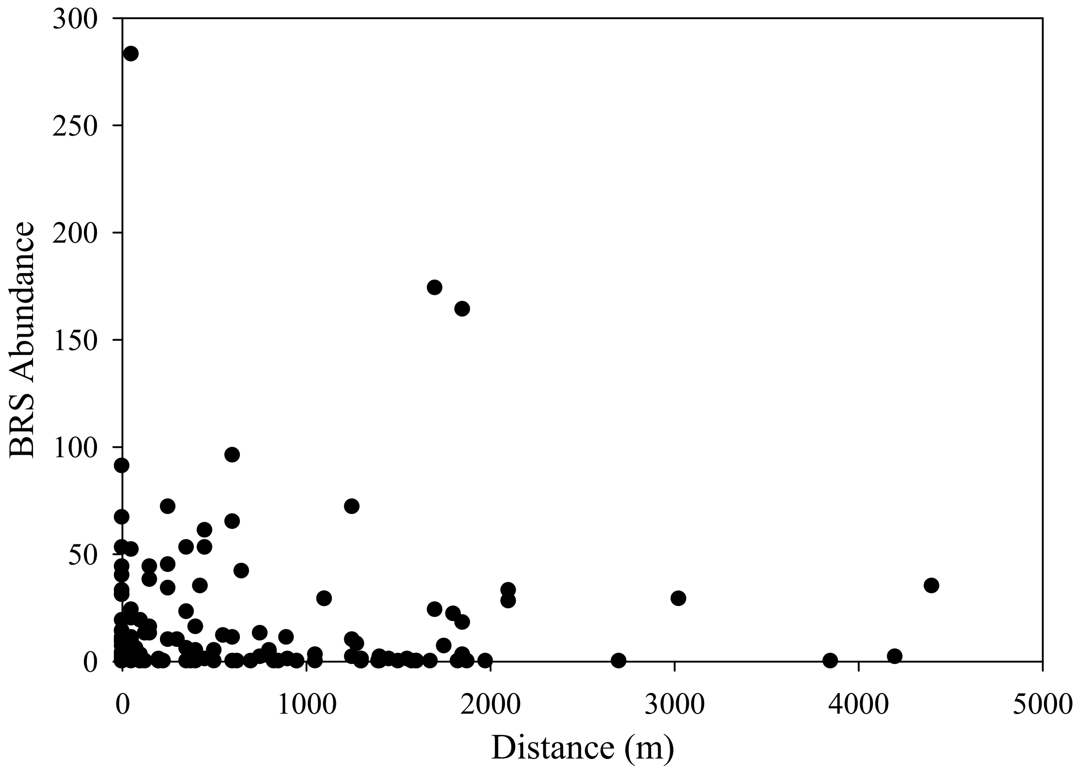

Figure 10 Comparison of Bliss Rapids snail abundance and distance from the nearest upstream rapid. Bliss Rapids snail abundance showed a weak negative correlation with distance of each section from the nearest upstream rapid ($p = 0.014$; $r_s = -0.219$; $n=126$).

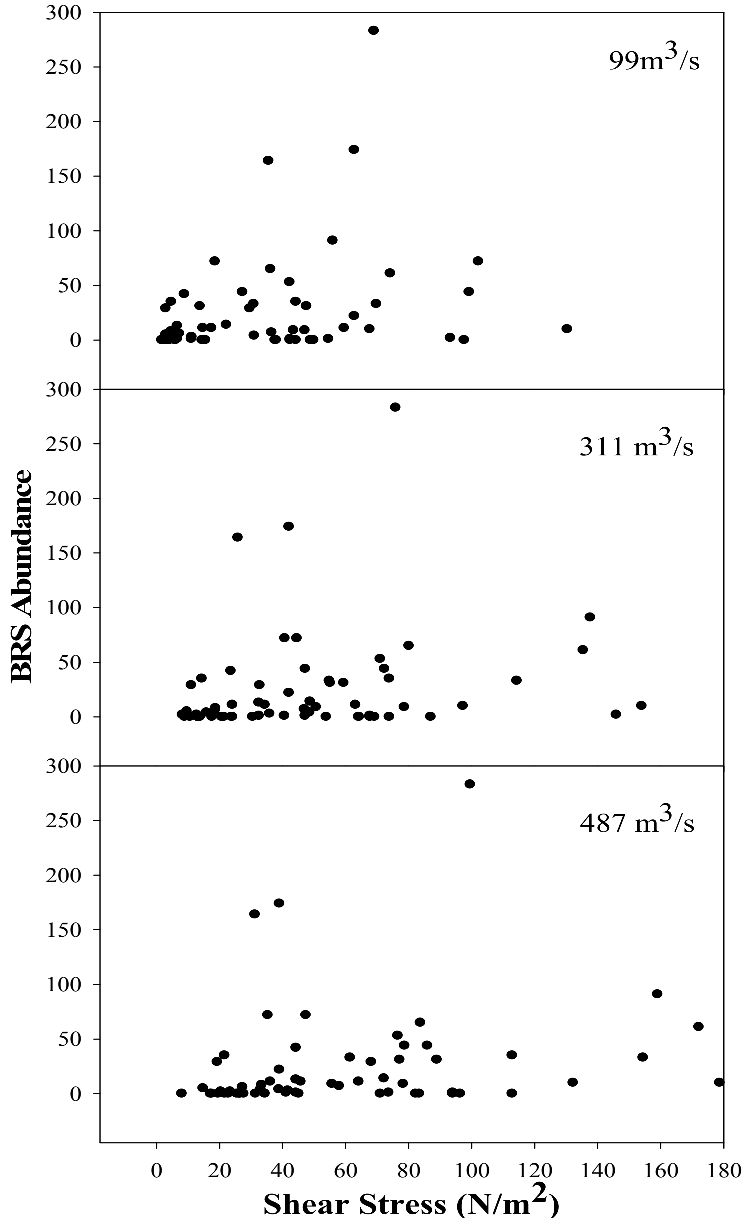

Figure 11 2005 Bliss Rapids snail abundance vs. shear stress at 99, 311 and 487 m^3/s Snake River discharge. Bliss Rapids snail abundance was significantly correlated with shear stress (see Table 7). These data represent the section spatial scale.

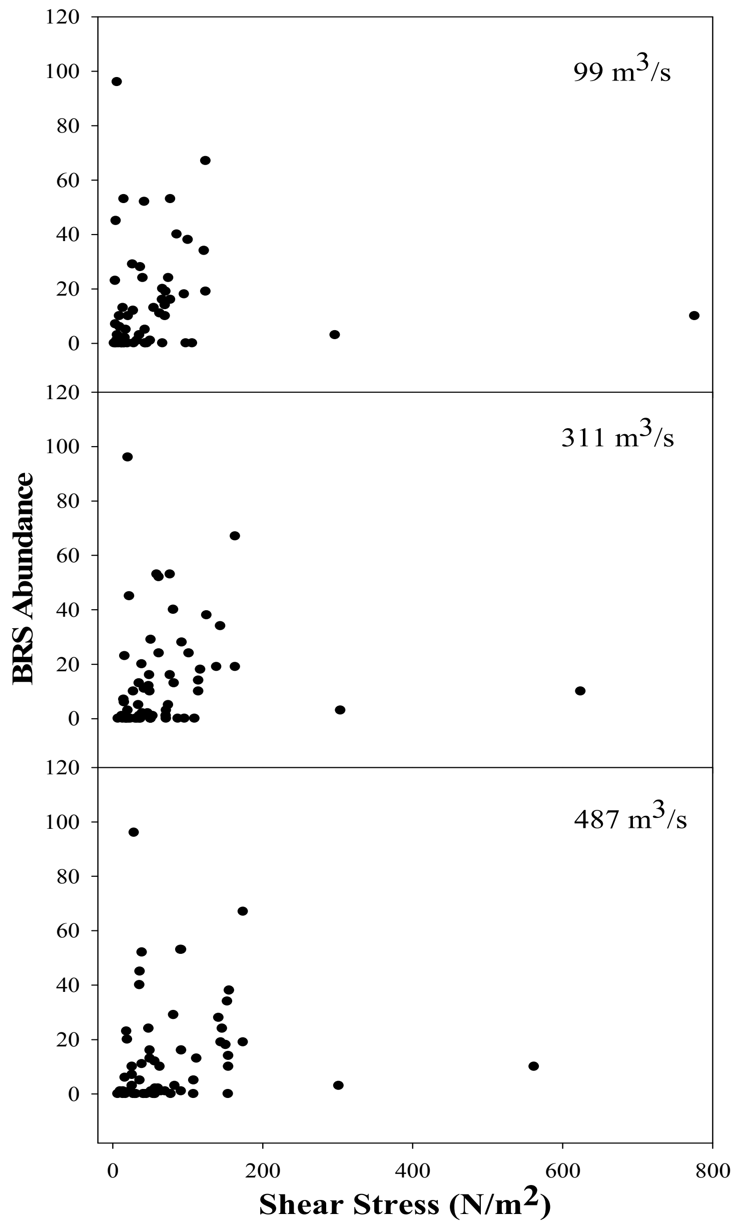

Figure 12 2006 Bliss Rapids snail abundance vs. shear stress for 99, 311 and 487 m^3/s river discharge. Bliss Rapids snail abundance was significantly correlated with shear stress for 311 and 487 m^3/s, but not for 99 m^3/s (see Table 7). These data represent the section spatial scale.

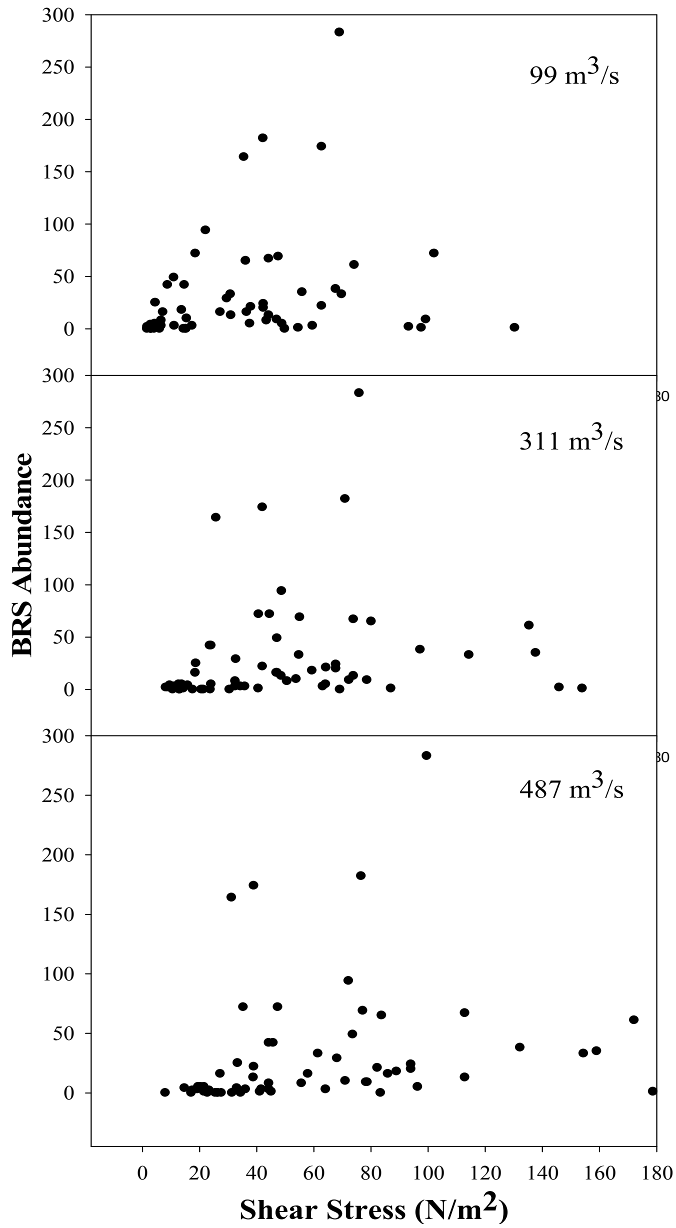

Figure 13 2007 Bliss Rapids snail abundance vs. shear stress for 99, 311 and 487 m^3/s river discharge. BRS abundance was significantly correlated with shear stress (see Table 7). These data represent the section spatial scale.

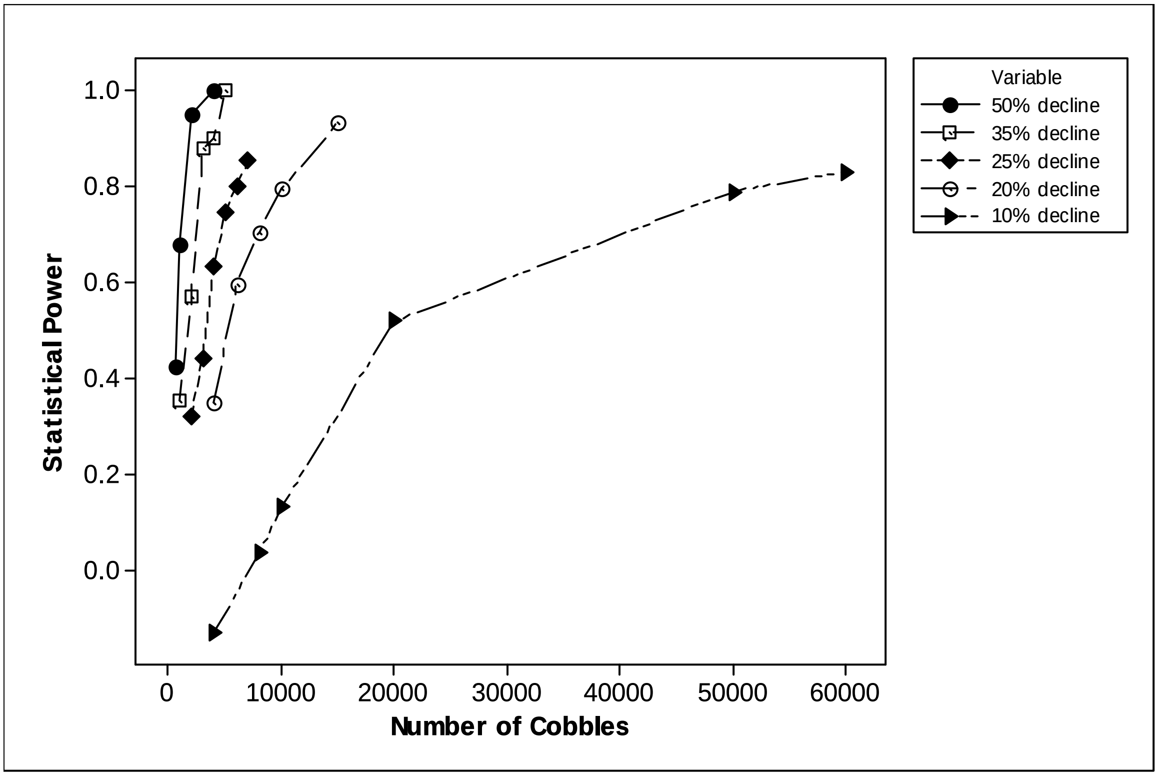

Figure 14 Comparison of the number of sample cobbles vs. statistical power for five simulated declines in BRS populations. Values presented here are minimum statistical power, calculated as the mean power minus the standard deviation (see Table 8).