Effect of Microhabitat and Land Use on Stream Salamander Occupancy and Abundance in the Southwest Virginia Coalfields

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Abstract

Large-scale land uses such as residential wastewater discharge and coal mining practices, particularly surface coal extraction and associated valley fills, are of particular ecological concern in central Appalachia. Identification and quantification of both large-scale land use and microhabitat alterations to ecosystems are a necessary first-step aid in mitigation of negative consequences to biota. In central Appalachian headwater streams absent of fish, salamanders are the dominant, most abundant vertebrate predator providing a significant intermediate trophic role. Stream salamander species are considered to be sensitive to aquatic stressors and environmental alterations with past research having shown linkages among microhabitat parameters, and large-scale land use such as urbanization and logging to salamander abundances. However, there is little information examining these linkages in the coalfields of central Appalachia. In the summer of 2013, I visited 70 sites (sampled three times each) in the southwest Virginia coalfields to collect salamanders and quantify stream and riparian microhabitat parameters. In an *information-theoretic* framework I compared the effects of microhabitat and large-scale land use on salamander occupancy and abundances. My findings indicate that dusky salamander (*Desmognathus* spp.) occupancy and abundances are more correlated to microhabitat parameters such as canopy cover than to subwatershed land uses. Brook salamander (*Eurycea* spp.) occupancy show negative associations to large-scale land uses such as percent recent mining and percent forested. Whereas *Eurycea* spp. abundances are negatively influenced by suspended sediments, stream bank erosion and stream substrate embeddedness. Management implications of these findings include erosion prevention and control as well as protection and management of riparian habitats. However, quantifying physical environmental quality such as stream and riparian habitat often can be quite difficult, particularly when there are time or fiscal limitations. In order to accurately assess stream and riparian habitat in a time- and cost- effective manner, the U.S. Army Corps of Engineers (USACE) developed a functional condition index (FCI) assessment for streams that measures 11 stream and riparian parameters along with watershed land use to calculate three different scores: a hydrology score, biogeochemical score, and habitat score (Noble et al 2010). Using the salamander data from 2013, I then analyzed the FCI scores using collected occupancy and abundance analyses. Both analyses supported the Habitat FCI score as it had strong correlations with both occupancy and abundance of three *Desmognathus* spp., and support the use of the USACE protocol for stream and riparian habitat assessment.

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"If I have seen further, it is from standing on the shoulders of giants."
-Sir Isaac Newton

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Preface

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Chapter 1

Evaluation of Current Measures of Aquatic Biological Integrity in the Central Appalachian Coalfields: Efficacy and Implications

Abstract

Within the central Appalachian coalfields, efforts to minimize and mitigate impacts from surface coal mining have been complicated by incongruent restoration goals and poorly validated post-mine monitoring protocols that are unable to quantify ecosystem change relative to system recovery. At present, assessment techniques range from simple single species examinations to complex multi-taxa/life history decision matrices believed capable of documenting cumulative and additive cross-system impacts. Additionally, some effort has been placed in using abiotic measures or narrow biotic components as system surrogates to comment on stream health of mined streams. These metrics may not accurately indicate biological change particularly when extended out to predict impacts and assess recovery. Benthic macroinvertebrates are valuable indicator species. However, aquatic ecosystem function may not always be able to be inferred from macroinvertebrate assemblages alone, and full assessment of stream health may require use of a suite of taxa such as periphyton, macroinvertebrates, fish, and/or salamanders.

Introduction

Central Appalachia has high levels of terrestrial and aquatic biodiversity and species richness, with many endemic species (Morse et al. 1993). The New River catchment alone is estimated to have 23 endemic plant and animal species (Hocutt et al. 1978). The aquatic systems in this region contain some of the highest levels of biodiversity in North America, including ten percent of global salamander and freshwater mussel diversity (Stein et al. 2000; Bernhardt and Palmer 2011). However, many of these species are listed as threatened or endangered (Bernhardt and Palmer 2011). For example, 46 species of freshwater mussels once inhabited the Powell River in Virginia and Tennessee (Ortmann 1918). Currently 36 species, seven of which are listed as federally endangered, remain (Ahlstedt et al. 2005). In the United States, the biggest threat to biodiversity is loss/degradation of habitat (Wilcove et al. 1998)

Most stream habitats in the eastern half of the United States, including those in central Appalachia, have been significantly altered by past or current anthropogenic disturbances (Karr 1991). From the time of European settlement of central Appalachia in the 1700s through the mid-1900s, central Appalachian streams were commonly used as unchecked drainages for residential wastewater, mining operation byproducts, and industrial discharges. The arrival of the railroads in the late 1800s brought large scale clearing and degradation of the land from exploitative practices such as commercial logging and coal mining that essentially impacted all of central Appalachian forests (Hibbart 1990; Nowacki and Abrams 1997).

In the 1960s and 1970s, researchers, policy makers, and industry became more aware of the degraded nature of not only central Appalachian streams but streams nationwide, which, along with a shift in society's view of environmental degradation, brought widespread, sweeping legislation and policy reform. The Federal legislation most influential to current coal mining practices are the Clean Water Act (CWA) of 1972, and the Surface Mining Control and Reclamation Act (SMCRA) of 1977 (Craynon et al. 2012). The CWA sets Federal regulatory standards for the identification, reporting, monitoring, and mitigation of degraded waters including those impacted by mining operations. However, under SMCRA, states were given primacy to set their own regulatory standards for surface mining as long as they are "no less effective than" Federal standards set by the CWA (Craynon et al. 2012). Reclamation standards of post-mine lands are also enforced under SMCRA (Starnes and Gasper 1995). Since the enactment of these legislative measures, much scientific focus has been placed on monitoring efforts to regulate mining activities, environmental impacts, and reclamation of post-mined lands (Starnes and Gasper 1995). However, data are not currently sufficient to predict the spatial impact of mining on stream conditions, nor is there a good understanding of the scope of subsequent changes in biological structure and function in aquatic ecosystems in watersheds where mining occurs (Simmons et al. 2008). Fully understanding the impacts of coal mining to central Appalachian streams is critical to develop proper reclamation/mitigation techniques.

Coal Mining Impacts

Currently, the extraction of Appalachian coal by mountaintop mining is common as it is cost-effective and allows for mining of shallow coal seams that cannot safely be mined by underground methods (Craynon et al. 2012). Although mountaintop mining accounts for 37% of coal production in Appalachia (National Mining Association 2012), it is the most controversial because of its disproportionately large environmental footprint. Mountaintop coal extraction uses explosives to blast rock layers (overburden) that sit above relatively shallow coal seams (Griffith et al. 2012). This process causes overburden to swell and take up much more volume as haul-able rock pieces than the original compacted geologic material. The amount of overburden produced is too costly to haul off the mine site for disposal, so overburden typically is pushed into nearby valleys. This process exposes vast quantities of unweathered geological materials that when exposed to atmospheric conditions and surface run-off often become acid-forming and/or leach heavy metals along with high levels of salts (Griffith et al. 2012).

Although it is clear that small headwater streams are buried during the mining process altering physical stream habitat, water chemistry, and aquatic biota, the longitudinal spatial scale of indirect disturbance is more challenging to empirically describe. The United States Environmental Protection Agency (EPA) estimates that in central Appalachia, ten percent of the region’s land area (500,000 ha) has been affected by mountaintop mining, and over 2,000 km of headwater stream habitat buried by valley fills (EPA 2011). Coal mining is considered a point source pollutant by regulatory agencies such as the EPA and the Virginia Department of Environmental Quality (VADEQ). However, the larger sizes of many surface mines along with the hydrologic changes that occur often require nonpoint source management strategies (Simmons et al. 2008).

Along with direct physical stream loss from valley fills, typically watershed hydrology, stream chemistry, and biological communities also are altered from pre-mined conditions (Starnes and Gasper 1995; Pond et al. 2008; Griffith et al. 2012). Physical alteration of streams resulting from the mining process such as dredging, burial of headwaters, and dam construction clearly alter stream hydrology. Changes from surface coal mining on upland terrestrial processes such as clearing of vegetation, increased impervious surfaces, and changes in topography, can also alter stream hydrology (Simmons et al. 2008; Merriam et al. 2011).

Impact of Other Land Uses

Many of these hydrological changes are not unique to coal mining land use, and are often seen in streams with urban development, and to a lesser extent agricultural and forestry land uses as well. For example, research in both urban and mining watersheds suggests that as watersheds become increasingly altered, vegetation is replaced by impervious surfaces such as parking lots or coal processing areas, the flow regime (hydrology) of a stream system responds more quickly to rain events becoming “flashy,” and results in more frequent and intense stream flood events (Starnes and Gasper 1995; Price et al. 2006). These high flow events can lead to channelization of the stream along with eroded, unstable stream banks (Booth et al. 2004). Instream habitat is buried by sediment as terrestrial erosion continues. If erosion is not controlled, terrestrial processes can be significantly altered by the loss of soil and nutrients (Baker 1985). These dramatic terrestrial changes alter physical stream habitat, and directly and indirectly influence water chemistry. All of these factors have had both acute and chronic effects on specific aquatic biota in mining-influenced streams (Northingham et al. 2011; Pond 2012).

Stream water changes are in part a function of upland land use alterations caused by practices such as mining, logging, and urbanization within the watershed. In central Appalachia these terrestrial alterations degrade water quality conditions ranging from elevated sediment rates, total dissolved solid (TDS) levels, acid mine drainage (AMD), and heavy metal concentrations in mined watersheds to increases in nutrients and organic matter from improperly discharged wastewater from residential areas (Simmons et al. 2008). The largest (by volume) and most pervasive stream pollutant in the U.S. is stream sediment, which can impact stream biota in two ways: streambed sedimentation and suspended sediment in the water column (Sweeten and McCreedy 2002).

Streambed sedimentation is one of the most prevalent headwater stream pollutants in central Appalachia (Angradi 1999), and occurs when heavier soil particles settle thereby clogging, covering, and embedding in-stream habitat (Harding et al. 1998). Many aquatic species rely on interstitial spaces between rocks and rocky surfaces for breeding, feeding, and predator avoidance. These rock surfaces also provide important attachment substrate for algae, benthic macroinvertebrates, and freshwater mussels. As interstitial spaces fill and streambed substrate covered, many aquatic taxa including benthic macroinvertebrates, freshwater mussels, algae, salamanders, and fishes are negatively affected. Suspended sediment from smaller silt- and clay-sized particles can remain suspended in the water column for extended periods, reducing visibility and light penetration (Matter and Ney 1981; Hartman et al. 2005). Primary producers such as algae and sight-feeding larval fishes such as brook trout (*Salvelinus fontinalis*) found in the headwaters of central Appalachia have been shown to be particularly sensitive to suspended sediments (Marschall and Crowder 1996; Sweeten and McCreedy 2002). In some areas of the central Appalachians, such as northern West Virginia and south-central Pennsylvania, streams have been heavily impacted by AMD from mine runoff and/or discharge that cause stream pH far below natural conditions. Acid mine drainage is estimated to effect over 4,000 km of streams in Appalachia, and currently it is the single largest non-point source pollutant in Pennsylvania (Reinhardt 1999).

Much is still unknown about many of the terrestrial and aquatic ecosystem dynamics and interactions caused by dramatic landscape alterations from coal mining practices (Stout and Wallace 2005; Simmons et al. 2008). Measuring land use impacts on stream health is further complicated by the additive effects caused by multiple land use practices such as agriculture, forest management, and residential areas (Welsh et al. 2005; Morgan et al. 2012). Studies of aquatic macroinvertebrate communities in central Appalachia found that mining stream impacts along with improper wastewater discharge from residential areas had an additive effect on species richness and diversity (Merovich and Petty 2007; Passmore and Pond 2009; Merriam et al. 2011).

Clean Water Act Requirements

In order to regulate and mitigate many of these negative impacts to streams in coal mined watersheds, Section 303(d) of the Clean Water Act requires each state to identify water bodies with degraded water quality, and to set geographically-based total maximum daily load (TMDL) standards. These TMDL standards are designed to best allocate loadings from multiple pollutant sources so as to not exceed the load capacity of the aquatic system (Barbour et al. 1999). The purpose of this process is to assess stream quality, establish pollution reduction controls, and monitor remediation outcomes. Under SMCRA, each state may establish their own regulatory and TMDL standards as long as they are "no less effective than" Federal standards under the CWA (Craynon et al. 2012). If a state determines a waterbody to be degraded, chemically or biologically or both, the stream must be reported to the EPA and placed on the impaired waters list (303(d)).

In Virginia, the VADEQ is responsible for ensuring the compliance of mining operations under the CWA. In order to determine a stream's impairment level, the VADEQ regularly monitors water chemistry parameters, and developed the Virginia Stream Condition Index (VA-SCI), a multi-metric benthic macroinvertebrate assessment protocol to identify biological impairment of streams (Burton and Gerritsen 2003). This protocol uses eight different metrics that evaluate different levels of pollutant sensitivity of macroinvertebrates, such as total taxa numbers, percent pollutant intolerant taxa, and percent pollutant tolerant taxa, to produce an overall stream condition score ranging from 0 (devoid of life) to 1 (optimal stream life). Based on comparison to local reference stream conditions, the VA-SCI places any stream scoring below 60 on the EPA impaired waters list (CWA 303(d)).

Once a stream is listed as impaired, the VADEQ uses a three-step TMDL process to improve water quality conditions. Individual stream characteristics and specific impairments such as sedimentation, conductivity, and low pH, are used to set TMDL standards, and an appropriate improvement implementation plan is developed and implemented (MapTech 2008). However, monitoring using the TMDL approach does not always provide sufficient guidance for the development of best management practices to effectively improve water quality and biotic communities. Guidance is particularly lacking in systems heavily impacted by nonpoint source pollution such as surface coal mines (Barbour et al. 1999). Endpoints for removing a stream from the impaired waters list also are established by the VADEQ, varying on a stream-by-stream basis. Often these endpoints are not clearly outlined in TMDL implementation plans (see Map Tech 2004); therefore achieving an unimpaired status can be complicated and problematic. In order to properly manage water resources using TMDL standards, there must be a full understanding of the relationships among water chemistry, effluent sources, non-chemical stressors, chemical stressors, and control practices.

Biological Integrity

Unfortunately, such TMDL standards often lack clear and consistent linkages to bioassessment results and/or biological integrity (Barbour et al. 1999). For example, in laboratory toxicology testing of water with elevated TDS, hardness, and alkalinity levels from two mining-impacted streams in West Virginia were acutely toxic to freshwater daphnia (*Ceriodaphnia dubia*). However, in field studies, these two streams had relatively high richness of macroinvertebrates in the orders considered to be very sensitive to poor water quality conditions: Ephemeroptera, Plecoptera, and Trichoptera (EPT) (Merricks et al. 2007). Without these linkages, TMDL standards may not be accurate. An overestimation of a stream's loading capacity can result in biological degradation whereas an underestimation may result in time and financial resources wasted on needlessly managing non-impactful chemical parameters. Inherent stream variability also may contribute to the complexity of interactions and linkages that must be understood to develop accurate TMDL standards.

As linear ecosystems, streams exhibit high levels of natural spatial and temporal variability influenced by both stochastic and anthropogenic instream, atmospheric, and watershed changes (Cummins 1975). Appalachian headwater streams are estimated to account for up to 75% of stream mileage in this region (Peterman and Semlitsch 2009). These headwater streams are closely linked to proximal terrestrial processes, have little to no photosynthetic production, and are diverse in benthic macroinvertebrate communities and generally dominated by shredder and collector taxa (Cummins 1975). As stream order increases, the direct influence of adjacent terrestrial habitat lessens and the influence of cumulative discharge from the multiple upstream drainages becomes more significant (Cummins 1975). This river continuum has been well-accepted by the scientific community to explain general changes in biological assemblages and physical habitat along the stream gradient (Vannote et al. 1980). For example, in an evaluation of central Appalachian rivers, Kerans and Karr (1994) recommend that shredder macroinvertebrate abundance should not be used in assessment of higher order streams as natural abundance of this group is not great enough to accurately detect water impairments. However, shredder abundance may be an important metric in headwaters where shredders are more abundant, and shown to be an accurate indicator of riparian alterations (Kerans and Karr 1994).

While the stream continuum theory helps to explain and minimize some of the natural variability that occurs in streams, many factors, such as local precipitation events, temporal scales, site-specific conditions, and transient dynamics (or the often unexplained variation in ecosystem behavior from its long-term trends with or without variation in external conditions). These natural processes all create difficulty in separating natural, "background" variability from anthropogenic impacts on aquatic systems (Hastings 2004; Hastings 2010; Johnson and Hering 2010). For example, in a survey of rivers throughout central Appalachia, Kerans and Karr (1994) found some metrics such as proportion of chironomids, shredders, and detritivores showed a closer relationship to the temporal scale than to location, indicating differences in macroinvertebrate assemblages may not always be the result of site quality (Kerans and Karr 1994). In order to best distinguish natural stream variations from anthropogenic impacts, it is important to have long-term stream condition data sets that analyze a variety of biological responses, since stressors may be revealed at various levels of biological organization as well as in physical stream habitat and water quality changes (Adams et. al 2002; Griffith et al. 2012).

Without historical data to provide insight into pre-disturbance conditions (and thereby an end goal), less-disturbed, i.e., unmined, reference watersheds are generally used as a surrogate (Younos et al. 2007). Finding a "truly undisturbed watershed" in central Appalachia is difficult as very few, if any, exist considering the widespread and pervasive nature of documented and undocumented past mining, timbering, and other cultural activities since European settlement (Simmons et al. 2008). Measuring the deviation of abundance and assemblage of taxa between mining impacted and reference streams may be useful in measuring impairment (Merovich and Petty 2007). Comparisons to reference streams also are used often by regulatory agencies (EPA and VADEQ) to identify and monitor stream degradation (Barbour et al. 1999). Ideally reference watersheds should have similar topography, watershed size, geology, and soil types to impaired stream watersheds.

However, scientific standards for the selection of reference watersheds have not been established, and reference site choices often left to “best professional judgment” (Younos et al. 2007). In central Appalachia, availability of optimal reference sites is limited, forcing water resource managers to select fewer reference sites or choose reference sites with different characteristics from the study watershed (Younos et al. 2007). For example, in the development of the statewide VA-SCI metrics that encompasses six level III ecoregions, a total of 62 reference sites were used; however, only five of these sites were in the central Appalachian ecoregion 69 (Burton and Gerritsen 2003). In a comparative study of mining impacts on headwater streams in southwest Virginia, Matter and Ney (1981) had to choose a 400% larger reference watershed that was located almost 50 km away from the study watersheds. However, even sub-optimal reference watersheds still offer the only insight into the establishment of water quality standards (Younos et al. 2007).

In response to national water quality concerns and monitoring approaches, the EPA Office of Water has advanced the Watershed Protection Approach (WPA) to consolidate and reorient state and Federal agencies that manage and regulate water resources by using watershed level management units. Past management focus has often been placed on small, highly degraded stream sections or individual pollutant sources. Often these efforts only provide a limited view of the total extent of degradation on a watershed with multiple mines or land uses (Barbour et al. 1999; Freund and Petty 2007). Ideally, the watershed approach provides an integrated and comprehensive evaluation of ecological conditions (integrity) in coal mining areas in order to best and most effectively protect aquatic resources. (Barbour et al. 1999).

In central Appalachia, mining impacts often are assessed and mitigated on an individual mine basis. Using the watershed approach helps to better manage the longitudinal, additive effects of multiple land uses such as mining and residential wastewater on stream health (Freund and Petty 2007). Common in mine-impacted watersheds, altered stream chemistry from AMD and/or high conductivity can isolate biological communities in upstream reaches reducing genetic diversity and recolonization ability. Linking upstream and downstream biological communities therefore is particularly important for the protection of the many endemic species found in central Appalachia. These linkages can most effectively be created and maintained when managing water resources at the watershed level (Freund and Petty 2007). The watershed approach allows for a cost-effective, realistic management planning where focus can be placed on restoration efforts for larger sections of less impacted streams instead of wasting resources on small, highly degraded streams that may never recover (Freund and Petty 2007).

Although there is an increased effort to remediate stressors within watersheds using the watershed approach, full understanding of how to measure stream conditions in the context of post-mining recovery continues to be a highly debated topic among scientists, regulators, and mining companies (Kerans and Karr 1994; Harding et al. 1998; Allan 2004; Milman and Short 2008; Simmons et al. 2008). The EPA and many states have developed detailed techniques for assessing streams over a large spatial scale, such as a region or state (i.e., EPA Rapid Bioassessment Protocol, VA-SCI and the West Virginia Stream Condition Index or WV-SCI). For example, the VADEQ uses the same protocol and scoring metrics (VA-SCI) on streams impacted by coal mining in the Appalachian Plateau region of southwest Virginia as are used for agricultural or urban-impacted streams in the Piedmont and Coastal Plain.

Discussion

Often to assess recovery and restoration, focus is placed on one stream condition parameter such as water chemistry or benthic macroinvertebrates over a relatively short time period (Kerans and Karr 1994). This approach often provides limited insight into the full extent of recovery and restoration success. Due to the natural temporal and spatial variability of stream ecosystems and the impacts of multiple anthropogenic stressors in central Appalachia on stream ecosystems, parameter choice can be somewhat problematic (Adam et al. 2002; Carlisle et al. 2009). The questions that remain are relatively simple to frame, but more difficult to answer: What is the level of aquatic ecosystem degradation? What spatial remediation is required to reach a valid benchmark of improvement? What factors should these benchmarks include?

Biological sampling "directly measures the condition of the resource at risk, detects problems that other methods may miss or underestimate, and provides a systematic process for measuring progress resulting from the implementation of water quality programs" (EPA 1990). Although not a substitute for chemical and toxicological monitoring, biological assemblages seem to show responses to the wide range of chemical conditions in mining impacted streams, even revealing additive effects of pollutants that individually do not exceed regulatory standards (Karr 1991; Freund and Petty 2007). Aquatic biological communities are sensitive to a range of environmental factors and watershed conditions across both long- and short-term temporal scales (Karr 1981). In central Appalachian coalfields, agency monitoring programs as well as research efforts often have used single taxon assessment protocols that usually focusing on benthic macroinvertebrates such as the VA-SCI and WVSCI (Stout and Wallace 2005; Santos and Stevenson 2011; Hartman et al. 2005; Pond et al. 2008; Passmore and Pond 2009; Merricks et al. 2007). Instead of focusing on a single group of organisms to monitor biological degradation, the integration of multiple taxa into bioassessment protocols has been shown to better measure both aquatic impairment and restoration status (Adams et al. 2002; Freund and Petty 2007).

In east Tennessee, Adams et al. (2002) found that using a suite of taxa such as periphyton, macroinvertebrates, and fish, improved the ability to identify the status of ecological recovery of disturbed aquatic systems. Biotic recovery often occurs on various time and spatial scales. Recolonization rates of biota may be affected by factors such as physiochemical stream conditions, taxa dispersal method, and pollution tolerance levels. Therefore, using a single taxon or trophic level for stream bioassessment may not always reflect a holistic ecosystem recovery. Incorporating bio-indicators from multiple levels of biological organizations and trophic levels helps to ensure a full view of recovery processes (Adams et al. 2002).

Although often considered to be too expensive and time-intensive, biomonitoring protocols have been established by state agencies such as the Ohio EPA and Federal agencies such as the EPA. For example, the Ohio EPA biomonitoring protocols incorporate benthic macroinvertebrates, fish, and salamanders that identify stream permanence as well as biological degradation (Ohio EPA 2012). The EPA’s Rapid Bioassessment Protocol (Barbour et al. 1999) outlines empirical, yet cost-effective protocols for biomonitoring multiple stream taxa including periphyton, benthic macroinvertebrates, and fish. In order to select biomonitoring taxa that will accurately and cost-effectively detect aquatic degradation in a given region, it is important to fully understand each taxa’s strengths and weaknesses for indicating environmental changes.

In central Appalachia, periphyton (algae) has been used infrequently in conjunction with water quality and/or macroinvertebrate surveys of mining impacted streams (Merricks et al. 2007). Primary producers such as periphyton can be directly affected by alterations in physical and chemical parameters, and often show significant responses when compared to other taxa at lower concentrations of stressors, such as herbicides (Stevenson 1998; Stevenson and Pan 1999; Barbour et al. 1999). Due to their short lifespan and quick reproductive rates, periphyton can be excellent indicators for examination of short-term impacts and recovery of degraded water quality. Algae are easy, quick, and cost-effective to sample and require little field training, and sampling is minimally invasive to the stream. However, laboratory analysis can be time consuming and requires specialized knowledge/training, although well-established, standard community assessment protocols do exist (Barbour et al 1999). Periphyton biomass and benthic macroinvertebrate assemblages have been shown to have close linkages in response to sedimentation and nutrient loading in large streams (Molinos and Donohue 2010) as well as to AMD impacts (Meegan and Perry 1996).

In central Appalachia, benthic macroinvertebrates have been relied on heavily for bioassessment for a variety of reasons. First, benthic macroinvertebrates have been shown to serve well as a site-specific, short-term biological indicator. Many trophic- and pollution-tolerance levels are represented in this group, and sampling is relatively easy and inexpensive. Invertebrates are also present in headwater streams that have limited fish communities (Barbour et al. 1999). Although bioassessment using benthic macroinvertebrates has many advantages, large variations in seasonal assemblages, sampling location within a watershed, and analysis of data often causes conflicting and unclear results (Kerans and Karr 1994; Fulk et al. 2003). Seasonal variability of macroinvertebrate assemblages has been debated with little resolution. Fulk et al. (2003) and Timpano et al. (2011) found seasonal variability between spring and fall macroinvertebrate scores when using the WV-SCI and VA-SCI assessment protocols, respectively. However, in the VA-SCI development and field validations, no seasonal differences in invertebrate scores were shown (Burton and Gerritsen 2003). If macroinvertebrate assemblages show significant seasonal variations, the VA-SCI protocol scoring should be adjusted in order to account for seasonality differences (Timpano et al. 2011). Currently the VA-SCI score has no such allowances for temporal variability and scores below 60 are considered impaired regardless of season (Timpano et al. 2011).

Sampling location selection at both the regional and local level has also been shown to produce variability in macroinvertebrate community structures. The VADEQ found that reference streams in the coalfields of southwest Virginia tended to have lower scores using the VA-SCI assessment method as compared to other ecoregions in Virginia. This may be due to natural differences in macroinvertebrate assemblages of central Appalachian streams relative to those of other Virginia ecoregions (Burton and Gerritsen 2003). The Ohio EPA recommends designing quantitative biological criteria based on individual ecoregion characteristics (Yoder 1991). The VA-SCI attempts to parsimoniously fit metrics that will work for all six ecoregions in Virginia. However, this approach may sacrifice some accuracy and resolution on the ecoregion level in order to encompass the whole state with one bioassessment protocol.

Macroinvertebrate assemblages may vary at different stream order and permanence levels within a watershed. In the southern Appalachians, Feminella (1996) found little variation in community structure between intermittent first-order streams and permanent lower order streams. Santos and Stevenson's (2011) results from a Massachusetts study were similar to Feminella's whereas Brown and Brussock (1991) found lower macroinvertebrate diversity in intermittent headwater streams in the Ozarks. Although these studies were conducted in different regions of the United States, they do indicate possible variability in macroinvertebrate assemblages along stream permanence gradients. Moreover, watershed size has been shown to influence macroinvertebrate assemblages and abundances as well. In central Appalachia macroinvertebrate species richness and abundance particularly the number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa increase rapidly along the stream gradient until watershed size of about 100 acres where the rate of increase becomes constant (Stout and Wallace 2005). The percentage of collector macroinvertebrates also increased along this gradient whereas percent shredders decreased (Stout and Wallace 2005).

Without resolution to the scope/scale of these natural temporal and spatial influences on macroinvertebrate assemblages, it is extremely difficult to properly account for natural variation within biomonitoring protocols such as the VA-SCI and WV-SCI. Without a full understanding of these natural variations, there may be inaccuracies in assessment of macroinvertebrate abundance and diversity. Analysis of macroinvertebrate surveys varies greatly among agencies and biologists, ranging from traditional EPT evaluation to multiple-metric analysis of community function and structure (Blocksom and Winters 2006). Taxonomic identification level of macroinvertebrates has not been standardized among regulatory agencies, researchers, and managers. Hilsenhoff (1987) created a simple index of biotic diversity by assigning pollution tolerance values to macroinvertebrates at the genus-level and dividing the sum of the tolerance values by the total number of individuals identified (around 100). In order to address time and fiscal constraints faced by many management agencies, Hilsenhoff (1988) used 120 Wisconsin streams to develop a family-level version of the index that reduced assessment time by 50%. However, the family-level index lost some resolution compared to genus-level evaluation. Although Pond et al. (2008) did not find significant differences in bioassessment results between identification to the genus-level and identification to the family-level, genus-level indices were recommended for measuring stream conditions in mined watersheds. These identification level differences can make direct comparison of bioassessment results to analyze regional and temporal trends in macroinvertebrate assemblages and coal mining impacts difficult.

Due to their high mobility and relative longevity, fish can serve as good long-term indicators of chronic stressors and reflect broad-habitat conditions of central Appalachian streams (Matter and Ney 1981; Angermeier and Karr 1986; Fulk et al. 2003; Freund and Petty 2007). Fish account for almost half of the endangered vertebrate species in the United States (Warren and Burr 1994) and represent a wide range of trophic levels and life histories (Angermeier and Karr 1986). A study of mine-impacted streams in West Virginia that compared indicator characteristics of fish and benthic macroinvertebrates found fish communities to be less responsive than macroinvertebrates to degraded water quality, suggesting that fish may not be a good indicator for bioassessment of local conditions (Freund and Petty 2007).

However, Birge et al. (2000) observed that fish were good biosensors for heavy metal levels in Kentucky streams, and may more accurately reflect watershed conditions than macroinvertebrates. In addition, life history and environmental requirements of most fish species are well understood, allowing for better understanding of community structure and function (Fausch et al. 1990). Most stakeholders, policy makers and land managers are more familiar and knowledgeable about fish (especially game species) than macroinvertebrates, thereby making fish assessment results more relevant and relatable (Karr 1981). Compared to other taxa, sampling of fish communities is cost-effective, and requires relatively little training or laboratory time (Karr 1981; Fausch et al. 1990). However, as with many taxa, sampling can be quite labor intensive and identification requires extensive knowledge. Fish communities often are influenced by stream permanence and by both diel and seasonal variations (Karr 1981). Fish assemblages may also be limited due to physical or chemical (such as AMD) barriers lower in the watershed.

Amphibians, particularly salamanders, are an important component to both terrestrial and aquatic ecosystems in central Appalachia (Davic and Welsh 2004). Long-lived stream salamander species also have been recommended for use as an indicator for aquatic bioassessment protocols due to their natural high abundances in undisturbed areas, sensitivity to stream and watershed degradation, and role as a keystone faunal group (Welsh and Ollivier 1998; Welsh and Droege 2001; Southerland et al. 2004). In headwater streams absent of fish, salamanders are the dominant, most abundant vertebrate predator. Stream salamanders provide a significant intermediate role in trophic cycling, feeding on small prey such as benthic macroinvertebrates and, in turn, are an important prey item for larger vertebrates (Petranka et al. 1993; Davic and Welsh 2004; Ohio EPA 2012). Central Appalachian riparian areas typically have relatively stable salamander populations (compared to macroinvertebrates), with densities of up to $1.4/m^2$ (Kleeberger 1984; Hairston and Wiley 1993; Welsh and Ollivier 1998). Life history and behavior vary greatly among salamander species with some larval forms having long aquatic phases of up to 48 months, whereas other species have very short- or even no aquatic larval phase (Ohio EPA 2012). Due to such variation in life history, the Ohio EPA uses larval salamander surveys as a method for accurately and cost-effectively indicating stream permanence and headwater habitat quality. Evidence of reproduction (eggs and larvae) of salamander species with larval phases exceeding 12 months has been found to indicate perennial stream flow while presence of species with larval phases of less than 12 months indicates intermittent stream flow (Ohio EPA 2012).

Salamander species that move between aquatic and terrestrial environments provide an important link between upland and stream ecosystems (Fisher et al. 1998). Stream salamanders are considered to be sensitive to aquatic stressors and environmental degradation, and low stream salamander abundance have been shown to be closely linked to terrestrial watershed degradations (Petranka et al. 1993; Willson and Dorcas 2003). Stream salamander abundances have been shown to be negatively impacted, even to the point of local extirpation, by decreases in stream conditions such as streambed sedimentation, temperature, pH, and suspended sediment, and from watershed land uses such as mining, urbanization, and timber harvesting (Gore 1983; Willson and Dorcas 2003; Welsh et al. 2005; Moseley et al. 2008). This gradient of response to anthropogenic impacts is critical for an indicator taxa group. If a group of organisms is highly sensitive to environmental degradation they will be quickly extirpated from the area, and may only indicate mild water quality impairments (Welsh and Ollivier 1998).

Taxa that are extremely tolerant of impaired stream conditions should also not be selected for use in biomonitoring as they will not show a response until the aquatic ecosystem is heavily degraded. Movement of many Appalachian stream salamander species has been shown to be less than 100m (Pauley et al. 2000), and although stream salamanders are able to cross barriers such as dry streambed sections and waterfalls, mobility may be limited by riparian and upper watershed habitat fragmentation (Resetarits 1997; Williams et al. 2002; Willson and Dorcas 2003; OHEPA 2012). Stream salamanders may also be useful to indicate recovery endpoints as their low mobility reduces the likelihood of increasing abundances being a reflection of immigration, and helps to confirm that local, long-term stream and watershed conditions are suitable for a positive rate of salamander survival and reproduction (Welsh and Ollivier 1998; Welsh et al. 2005).

Conclusion

Traditionally, research and monitoring of aquatic ecosystems has focused on the effects of land use practice disturbances. However, the ability to distinguish natural stream variability from the additive effects of anthropogenic influences is currently limited. Furthermore, there is little known about the multiple temporal and spatial dynamics of the biological recovery of a degraded aquatic system (Harding et al. 1998; Adams et al. 2002; Niemi et al. 2004; Hartman et al. 2005; Milman and Short 2008; Simmons et al. 2008; Northingham et al. 2011). Currently, even identifying and quantifying aquatic ecosystem structure and function of undisturbed systems can be very difficult (Simmons et al. 2008). Fiscal and personnel constraints are often the limiting factors to widespread or more comprehensive sampling. It is, however, imperative to establish scientifically-backed bench marks of success for watershed remediation efforts and/or coal mine reclamation. Biological assessment using a suite of taxa may provide the most extensive view of aquatic changes including degradation and recovery. In order to develop optimal bioassessment protocols, it is essential to first gain a better understanding of the response of each taxa to water conditions as well as the relationship dynamics among taxa. Investigation of salamanders as bio-indicators for streams in mined watersheds should be a priority, as little is known about their indicator abilities in central Appalachia.

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Chapter 2

Effects of Microhabitat and Land Use on Stream Salamander Occupancy in the Southwest Virginia Coalfields

Abstract

Large-scale land uses such as residential wastewater discharge and coal mining practices, particularly surface coal extraction and associated valley fills, are of particular ecological concern in central Appalachia. Identification and quantification of both large-scale land use and microhabitat alterations to ecosystems are a necessary first-step aid in mitigation of negative consequences to biota. In central Appalachian headwater streams absent of fish, salamanders are the dominant, most abundant vertebrate predator providing a significant intermediate trophic role. Stream salamander species are considered to be sensitive to aquatic stressors and environmental alterations, and past research has shown linkages among microhabitat parameters, large-scale land use such as urbanization and logging, and salamander abundances. However, there is little information examining these linkages in the coalfields of central Appalachia. In the summer of 2013, I visited 70 sites (sampled three times each) in the southwest Virginia coalfields to collect salamanders and quantify stream and riparian microhabitat parameters. Using an *information-theoretic* framework I compared effects of microhabitat and large-scale land use on stream salamander occupancy. My findings indicate that dusky salamander (*Desmognathus* spp.) occupancy rates are more correlated to microhabitat parameters such as canopy cover than to subwatershed land uses. However, brook salamander (*Eurycea spp.*) occupancy rates had a strong association to large-scale land uses, particularly percent recent mining and percent forested. Management implications of these findings include protection of riparian habitats and erosion prevention and control.

Introduction

Amphibians, particularly salamanders, are an important component to both terrestrial and aquatic ecosystems in central Appalachia (Davic and Welsh 2004). Long-lived stream salamander species are a keystone faunal group, have natural high abundances in undisturbed areas, and their life history traits that make them sensitive to both stream and upland watershed alterations (Welsh and Ollivier 1998; Welsh and Droege 2001; Southerland et al. 2004). In headwater streams absent of fish in temperate forests salamanders are the dominant, most abundant vertebrate predator. Stream salamanders provide a significant intermediate role in trophic cycling, feeding on small prey such as benthic macroinvertebrates and are an important prey item for larger vertebrates (Petranka et al. 1993; Davic and Welsh 2004; OHEPA 2012). Undisturbed riparian areas of central Appalachia have relatively stable, intra- and inter-annual salamander populations (compared to macroinvertebrates), with densities of up to 1.4/m^2 (Kleeberger 1984; Hairston and Wiley 1993; Welsh and Ollivier 1998). Movement of many Appalachian stream salamander species has been shown to be less than 100m (Pauley et al. 2000), and although adult stream salamanders are able to cross barriers such as dry streambed sections and waterfalls, mobility may be limited by riparian and upper watershed habitat fragmentation (Resetarits 1997; Williams et al. 2002; Willson and Dorcas 2003; OHEPA 2012). Stream salamanders may also be useful in indicating recovery endpoints. Their low mobility reduces the likelihood of abundance being a reflection of immigration, which helps confirm that local, long-term stream and watershed conditions are suitable for salamander survival and reproduction (Welsh and Ollivier 1998; Welsh et al. 2005).

Life history and behavior vary greatly among aquatic salamander species with some larval forms having aquatic phases of up to 48 months, whereas other species have very short- or even no aquatic larval phase (OHEPA 2012). This biphasic life history trait shown by some salamander species can provide a gradient of responses to aquatic and upland environmental conditions giving better insight to both terrestrial and stream conditions (Fisher et al. 1998; Petranka 1998).

Stream salamander abundance has been shown to be closely linked to terrestrial watershed condition (Petranka et al. 1993; Willson and Dorcas 2003). Stream salamander abundances are reduced, even to the point of local extirpation, by decreases in physiochemical stream and riparian conditions from watershed land uses such as mining, urbanization, and timber harvesting that cause changes in pH, streambed sedimentation, suspended sediment, and water and soil temperature (Gore 1983; Willson and Dorcas 2003; Welsh et al. 2005; Moseley et al. 2008).

Recent research has tried to better quantify the effects of anthropogenic land use on salamander abundance, density, and most recently, occupancy. The majority of this work has focused on the effects of forestry practices on terrestrial salamanders. In particular, the Pacific Northwest has been a geographic focal point for much of this forestry/salamander research (Ashton et al. 2006; Bury and Corn 1989; Jackson et al. 2007; Kroll et al. 2008; Kroll 2009; Kroll et al. 2010; Olson and Weaver 2007; Russell et al. 2004; Russell et al. 2005; Stoddard and Hayes 2005; Welsh and Ollivier 1998; Wilkins and Peterson 2000). Stream salamander responses to forest management practices at both microhabitat levels and landscape-levels ranging from local extirpation to stable high abundances following timber harvest (Kroll et al. 2008; Kroll et al. 2010; Russell et al. 2005; Welsh et al. 2005). Timber harvesting often causes changes in microhabitat conditions including decreased canopy cover, increased substrate embeddedness and suspended sediment, and increased soil and water temperatures, all of which have been shown to greatly influence stream salamander occupancy (Russell et al. 2004; Russell et al. 2005; Stoddard and Hayes 2005; Wilkins and Peterson 2000). At the landscape-level, stream salamanders have shown positive associations to older, mature forests, although this association may vary by salamander species

(Kroll et al. 2008; Kroll et al. 2010; Stoddard and Hayes 2005). For example, Kroll et al (2008) found that *Dicamptodon* spp. occupancy was positively correlated to stand ages up to 70 years whereas *Rhycacotriton* spp. occupancy increased with stand ages up to about 50 years old and then decreased in areas with a stand age of 50 to 70 years.

Similarly, salamander research on land use in the eastern United States has also focused on forestry practices in relation to salamander abundances. In the central and southern Appalachians, data suggest forestry practices and younger stand age, at least in the short-term, lead to declines in terrestrial salamander abundance (Ash 1988; Ash 1997; Crawford and Semlitsch 2008; Ford et al. 2002; Harper and Guynn 1999; Harpole and Haas 1999; Petranka et al. 1993; Reichenbach and Sattler 2007). Nonetheless, little data exist on stream salamander occupancy and microhabitat or large-scale land use in the Appalachians. However, stream salamander abundance research in the central and southern Appalachians has suggested linkages between stand age/forest practices and stream salamander abundances (Moseley et al. 2008; Peterman and Semlitsch 2009). At smaller spatial scales, logging-related alterations to riparian quality (such as reduced canopy cover, leaf litter depth and decreased soil moisture) along with the alteration of physical stream conditions (such as change in stream substrate class, substrate embeddedness, riffle/run/pool composition) negatively influence stream salamander populations (Crawford and Semlitsch 2008; Maigret et al. 2014; Moseley et al. 2008; Peterman and Semlitsch 2009; Yeiser 2013).

In examining urbanization as a stressor in the Piedmont region of North Carolina, stream salamander abundances are negatively correlated to increasing urbanization (Miller et al. 2007; Orser and Shure 1972; Price et al. 2006; Price et al. 2012; Willson and Dorcas 2003). At the landscape-level, Price et al. (2012) found significantly higher survival rates and abundance of

Desmognathus fuscus in undisturbed watersheds as compared to urban watersheds. Salamander populations are negatively correlated to microhabitat parameters associated with urbanization such as changes in hydrology from increased impervious land cover, amplified/frequent high flow events, increased sedimentation, and higher water temperature (Miller et al. 2007; Orser and Shure 1972; Price et al. 2006; Price et al. 2011). Price et al. (2011) conducted one of the few studies that examined the effects of urbanization on stream salamander occupancy. They found increased urbanization over a five year period both larval and adult *Eurycea bislineata cirrigera* had decreased occupancy rates. After 2 years of urbanization larval *Desmognathus fuscus* had decreased occupancy rates, whereas adult *D. fuscus* occupancy remained stable (Price et al. 2011). This suggests that not all life-stages or stream salamander species respond in the same way to large-scale land uses.

In contrast, efforts to examine effects to aquatic salamanders from coal mining have been fewer in scope and extent over the past three decades relative to forestry and urbanization research. Most of the amphibian research on mined lands has focused on constructed settling ponds and use and recolonization by pond salamanders such as those in the family Ambystomidae (Fowler et al. 1985; Jansen et al. 2004; Kirk 2000; Lacki et al. 1992; Loughman 2005; Turner and Fowler 1981). Much research has centered on acid mine drainage due to its clear direct effects on stream biota including stream salamanders (Freda 1986; Middlekoop et al. 1998; Whiteleather 2001; Schorr et al. 2013). However, acid mine drainage is not generally an issue in southwest Virginia (Herricks and Cairns 1974; Minear and Tschantz 1976). Often, salamander research in the coalfields has been a small component of a larger study that examined multiple taxa (Carrozzino 2009; Loughman 2005). Furthermore, this research tended to be more observational and descriptive in nature.

To date, salamander research relative to coal mining in the central Appalachians has focused on the examination of valley fills as compared to reference conditions. In the southern West Virginia coalfields, Hamilton (2002) examined the effects of valley fills on stream salamander relative abundance using three streams with valley fills and two reference streams. The younger two valley fill streams (15 years and 5 years post-reclamation) were found to have lower stream salamander abundance than the reference streams. However, the oldest valley fill (18 years post-reclamation) had salamander abundances that did not statistically differ from the reference sites. Wood and Williams (2013a) examined stream salamander abundances in three reference streams compared to three streams with valley fills in southern West Virginia. They found lower relative abundances in the valley fill streams (N = 780 captures) than in the reference streams (N = 1563 captures), however species richness was the same in valley fill and reference streams. This study also examined differences in water chemistry and physical stream habitat among sites, but did not quantify terrestrial riparian conditions such as canopy cover or detritus depth. Muncy et al. (2014) compared stream salamander occupancy and species richness between 11 first-order valley fill streams and 12 first-order reference streams at the University of Kentucky's Robinson Forest. Estimated occupancy probabilities and species richness, along with forest cover and the number of cover objects, were higher in reference streams than in valley fill streams. This study also measured stream width and depth as well as water chemistry parameters associated with conductivity (SO_4, Ca, K, and Na) at each site. Most of this research would have been capable of greater inference with more range of study locations, measurement of additional microhabitat and riparian conditions, and large-scale analysis of watershed land. The results found in these three studies tend to be similar to the ecological changes associated with other anthropogenic land uses i.e. forestry, agriculture, and urbanizations when compared to reference stream conditions.

It is expected that many of the aquatic effects from coal mining in southwest Virginia will closely mirror known effects of forestry and urbanization land use such as increased sedimentation, changes in stream flow and thermal regimes, and decreased riparian and water quality. Past research indicates that stream salamander populations respond similarly to most land use practices as changes to streams, watershed hydrology, and riparian habitat are parallel among land uses; however the magnitude of the land use can produce more negative changes to streams. Regardless of study location or terrestrial land use, both adult and larval stream salamander communities have repeatedly shown to be negatively affected by stream sedimentation, that can be a result of agriculture, mining, urbanization, and roads; however, sedimentation from a large-scale land use such as agriculture will likely have a larger input of sediment than a small-scale land use such as a single-lane, paved road (Ashton et al. 2006; Harding et al. 1998; Hawkins et al. 1983; Lowe et al. 2004; Miller et al. 2007; Moseley et al. 2008; Peterman and Semlitsch 2009; Ward et al. 2008). Stream salamanders may be significantly affected by fine spatial scales (microhabitat) more than landscape-level land use; however, microhabitat conditions are often directly related to cumulative historical and current watershed land use (Bury and Corn 1989; Olson and Weaver 2007; Russell et al. 2004; Russell et al. 2005; Surasinghe and Baldwin 2014; Stoddard and Hayes 2005; Wilkins and Peterson 2000). These data illustrate the need for understanding the dynamics among salamander occupancy, land use, and microhabitat.

In order to investigate the possibility of developing a salamander index of biotic integrity, the relationship of salamander species response to a gradient of environmental conditions and land uses must first be established for the central Appalachia region. The main objective of my study was to investigate these critical correlations between salamander communities and habitat parameters at both a microhabitat and a landscape-level along a gradient of conditions in order to determine if more research is warranted for the development of a salamander index of biotic integrity for central Appalachia. I examined these foundational correlations with an *information-theoretic* occupancy/detection modeling approach using microhabitat data collected in the field along with a Geographic Information System (GIS) analysis of landscape-level land use.

Methods

Regional Description

My study sites are located in the coalfields of southwest Virginia in Wise, Russell, and Dickenson counties. This area is part of the Cumberland Plateau (Ecoregion 69d), a sub-region of the Central Appalachian Mountains (Omernik 1987). This region is also commonly known as the Northern Cumberland Plateaus and Mountains Ecoregion (Anderson et al. 2001; Bailey 1995). Topography is characterized by steep mountains with narrow valleys with a mean peak elevation of 760 m (Woodward and Hoffman 1991). Most soils in this region are Utisols, Alfisols, and Inceptisols (McNab and Avers 1995). Mean annual precipitation is about 1150 mm with a mean temperature of 13°C (McNab and Avers 1995; Woodward and Hoffman 1991). Due to the steep topography, this region tends to have a high density of small/medium-sized streams (McNab and Avers 1995). It is estimated that 93 % of the Cumberland Plateau is forested, whereas 4 % of the region is agricultural/open area (VDGIF 2005). Coal mining (both surface and underground), forestry, agriculture, industry, and residential areas (many with inadequate sewage infrastructure) are the dominant land uses (VDGIF 2005). The forested areas are characterized by a diverse mix of hardwood and conifers (Woodward and Hoffman 1991). Common tree species include red oak (*Quercus rubra*), white oak (*Quercus alba*), pignut hickory

(*Carya glabra*), red maple (*Acer rubrum*), yellow poplar (*Liriodendron tulipifera*), American beech (*Fagus grandifolia*), basswood (*Tilia Americana*), and white pine (*Pinus strobus*) (Hamilton 2002; McNab and Avers 1995).

Historical Mining

Settlers had known about the coal resources in central Appalachia as early as 1750; however, similar to logging, it was not until after the arrival of railroads in the region in the late 1880s that large-scale mining began. Mechanization processes in the 1920s increased productivity and extraction rates, and as a result mining operations became larger and altered more land in the Appalachians. In southwest Virginia, surface coal mining began in the early 1940s (Hibbart 1990). Throughout most of this time, environmental impacts and reclamation of post-mined lands were largely ignored. Until 1939 when West Virginia enacted a law requiring mine land reclamation, there had been no state or federal legislation to regulate post-mine land recovery in the region (Starnes and Gasper 1995). Since the enactment of the Clean Water Act (CWA) in 1972 and the Surface Mining Control and Reclamation Act (SMCRA) in 1976, most scientific focus has been placed on monitoring efforts to regulate mining activities, environmental alterations, and reclamation of post-mined lands (Starnes and Gasper 1995). Under SMCRA, each state may establish their own regulatory and total maximum daily load (TMDL) standards as long as they are "no less effective than" Federal standards under the CWA (Craynon et al. 2012). In Virginia, the Virginia Department of Environmental Quality (VADEQ) is responsible for ensuring the compliance of mining operations under the CWA. In order to determine stream impairment, the VADEQ monitors water chemistry parameters, and developed the Virginia Stream Condition Index (VA-SCI), a multi-metric benthic macroinvertebrate assessment protocol to identify biological impairment of streams (Burton and Gerristsen 2003).

Site Selection

For my study area, I selected five 12-digit Hydrologic Unit Codes (HUC-12) watersheds in southwest Virginia (Figure 2.1; Table 2.1). These five watersheds are similar in area, located within the Cumberland Plateau and Mountains Region, and have active coal mining along with other land uses. I divided streams within these watersheds into segments by their stream order, and gave each stream segment in each watershed a unique identification number. Based on these identification numbers, I randomly selected first- or second-order stream segments for salamander sampling sites; however, in some cases I used best professional judgment to select sites that had landowner access and that I considered safe to sample. Throughout the five HUC-12 watersheds, I sampled a total of 70 sites (N = 70). Due to a lack of accessible sites on Pigeon Creek and Dumps Creek, these watersheds had 10 salamander sampling locations. Roaring Fork, and Rocky Fork had 15 salamander sampling locations, and there were 20 sites on Callahan Creek.

To accurately estimate detection rates, I visited 67 of the 70 three times in 2013 (Bailey et al. 2004; MacKenzie and Royle 2005). Due to access issues, I was only able to sample three of the sites twice in 2013. At each sampling site location a 25 m long by 5 m wide quadrat was placed parallel to the stream with the stream center as the right or left edge of the quadrat (Figure 2.2; Hairston 1986; Kucken et al. 1994; Jung et al. 2000). Right or left quadrat placement was determined randomly using a coin flip. I hand-captured adult salamanders by overturning all rocks, detritus, and logs within the 25 m x 5 m quadrat at each sampling site. I identified all adult salamanders to species in the field and immediately released to within 2 m of capture location. A D-frame dip net was used to sample in-stream habitat (Davic 1983; Gore 1983). All larval salamanders were removed by hand from the dip net, placed in a bucket of fresh stream water for identification, and then released within 2 m of the capture location. Due to the difficulty of identifying larval salamanders to the species level, I identified larval salamanders to genus.

Field Parameters

Along with adult and larval salamander sampling, I collected physical stream attributes, water chemistry, and ambient conditions were collected from a quadrat 25 m x 10 m that combined the left and right sides of the salamander quadrat (Figure 2.2). Overall, the parameters in this study are a compilation of parameters past research has suggested to influence salamander abundance and/or occupancy (Bury and Corn 1989; Kroll et al. 2010; Miller et al. 2007; Moseley et al. 2008). I sampled parameters thought to have high temporal variability such as stream flow condition, soil air and water temperatures, and water chemistry, using the appropriate gear at each of the three site visits (See Table 2.3 for details). I measured parameters that were likely to remain constant over the summer such as percent canopy cover, tree species richness, and large woody debris counts, once during the field season. Descriptions of all parameters collected in the field for this study are provided in Table 2.3.

GIS Analysis

I assembled multiple GIS layers including the 2011 National Land Cover Dataset (NLCD), the 2011 National Agriculture Imagery Program (NAIP images, historical topographical maps, land use cover data from Maxwell et al (2014), mining GIS data from the Virginia Department of Mines Mineral and Energy (DMME) for analysis in ArcMap (ESRI, Redlands, California) to determine land use at both the subwatershed (the watershed from the above the sampling point) and HUC-12 scale (Table 2.4). I measured watershed area, stream order of sampled streams, length of stream segment, and stream order of receiving streams. Land use was classified as percent of each watershed with recent mining (active mining permits and areas still barren), past mining, and forested. Additionally, I measured area of valley fills and kilometers of roads. I also counted the number of structures and the number of constructed ponds in each subwatershed.

Statistical Analysis

To assess salamander occupancy and detection probabilities in an *information-theoretic* framework, I used the Program PRESENCE software (available for download at www.mbr-pwrc.usgs.gov/software/presence). Program PRESENCE was developed to examine the relationship between species presence and measured parameters through statistical modeling. Program PRESENCE fits models of these relationships using Akaike's Information Criterion (AIC) values and maximum likelihood to rank all detection and occupancy models, and will determine the best-fit models for presence/absence data (Bailey et al. 2007; Kroll et al. 2010).

Occupancy modeling uses species presence/absence data along with habitat data to examine the best-fit relationships between species presence and a suite of measured parameters MacKenzie et al. (2002; 2006). This technique best analyzes observational research by helping to avoid the over-fitting of models that is oftentimes associated when correlating a large number of habitat parameters to species abundance. The best-fit models and occupancy probabilities produced in the data analysis provide foundational and currently unknown probabilities of finding a given species at a site with certain microhabitat or landscape-level conditions. These occupancy modeling techniques have been widely used on a number of taxa ranging from salamanders in the eastern United States to tigers, *Panthera tigris*, in India (Nichols and Karanth 2002; Bailey et al. 2004; MacKenzie and Royle 2005).

Additionally, I estimated detection probabilities for each species to determine which environmental conditions most influence detection. Detection is important in occupancy analyses in order to get the best estimate of psi (probability of occupancy; Ψ). Without considering detection in data analysis, the true presence of a species may be misclassified as absent when the species was present but not detected. Often this misclassification will underestimate the occupancy probability (Dorazio et al. 2006; MacKenzie 2006). Calculating detection probabilities also helps determine the optimal sampling design to maximize efficiency while minimizing the probability of a false absence or not finding a species that is present at a given site. This detection rate can provide natural resources managers a tool for measuring aquatic biotic health as well an estimate of future sampling effort required (MacKenzie and Royle 2005).

Species Selection

In order to determine which species had a large enough sample size for analysis, I used the following equation from Wintle et al. (2012):

$$n = \frac{\log\left(\frac{\propto}{1-\propto}\right) - log\left(\frac{\varphi}{1-\varphi}\right)}{\log(1-\rho)}$$

where n = the number of times a site needs to be sampled, α = the given probability of a Type I error (0.1 for this study), ψ = the probability a site is occupied by the species, and *p* = the probability the species was detected.

Model Selection

I used a two-step method to determine which detection covariates to include for each species in the occupancy analysis (Burnham and Anderson 2002). I ran *a priori* detection covariates for each species against the null (intercept) model. All detection covariates that had an AIC smaller than the null were then run for all combinations to determine the best detection covariate for each species. I then used this detection covariate for the occupancy analysis. Utilizing this two-step method helps to reduce the total number of models in the final analysis. For this study, I analyzed seven detection covariates (Table 2.5). Prior to analysis, I normalized continuous detection covariates in Program PRESENCE.

I developed 18 *a priori* models for microhabitat occupancy analysis (Table 2.6). These models contained the 21 parameters denoted in Table 2.3 plus the elevation parameter in Table 2.4. Incorporating 14 GIS-derived covariates (Table 2.4), I developed 15 landscape-level *a priori* models (Table 2.7). For both sets of models, I grouped covariates based on biologically relevant information or recommendations from available literature (Ford et al. 2002; Moseley et al. 2008; Petranka et al. 1993; Willson and Dorcas 2003). All combinations of the parameters were not considered due to the large number of models this would have created and the subsequent unreliable outputs (Burnham and Anderson 2002; Dillard et al. 2008). For both sets of models, I normalized continuous variables prior to analysis.

AIC Comparison

After analysis of microhabitat and landscape-level models was completed, I did a comparison of microhabitat and landscape-level AIC values and weights in order to examine the relative importance of the two model groups. I calculated delta AIC values by subtracting the AIC of each model from the AIC of the model with the smallest AIC value. AIC weights were calculated using the equation (Burnham and Anderson 2002):

$$AIC_{Weight} = \frac{Exp(-0.5 * \Delta AIC)}{Summed\ Value\ of\ all\ models\ Exp(-0.5 * \Delta AIC)}$$

Model Averaging

In order to examine the strength of individual covariates across models, I calculated model averages for each covariate using the equation (Burnham and Anderson 2002):

$$\hat{\theta} = \sum_{i=1}^{R} w_i \, \hat{\theta_\iota}$$

where w_i is the AIC weight and $\hat{\theta}$ is the beta estimate of the covariate.

Results

Species/Genus Selection for Analysis

Overall, I detected nine species of aquatic salamanders during the 207 surveys (Table 2.9). Due to the large number of *Eurycea* spp. larval salamanders (442 individuals) and the small number of adult *Eurycea* spp. (39 individuals), I combined larval and adult *Eurycea longicauda* and *Eurycea bislineata cirrigera* salamanders to the genus-level for the *Eurycea* spp. group. Based on estimated occupancy and detection probabilities, I calculated the number of survey replicates needed to infer absence for 14 species or genus groups (Wintle et al. 2012). Five of these groups had enough statistical power based on three sampling events to be used in analysis (Table 2.8). However, after further analysis the genus group of *Desmognathus* spp. adults I found that although this group had enough sampling replicates, it consistently had a chi-square statistic of about four indicating an overall lack-of-fit, and I therefore excluded this group from further analysis. Species or genus groups that were detected too infrequently to be included in the occupancy analysis were *Desmognathus welteri*, *Desmognathus* larval, *Eurycea* adults, *Eurycea bislineata cirrigera*, *Eurycea longicauda*, *Gyrinophilus porphyriticus*, *Pseudotriton montanus*, *Pseudotriton ruber*, and the combined genus group of *Pseudotriton* and *Gyrinophilus* (Table 2.9).

Of the salamander groups used in analysis, observed occupancy rates (naïve occupancy or the percentage of sites occupied) ranged from 0.3857 for *Desmognathus fuscus* to 0.7 for *Eurycea* spp. (Table 2.8). As expected, estimated model averaged occupancy probabilities (Ψ) were higher than observed occupancy rates and ranged from 0.4507 (SE = 0.0743) for *Desmognathus fuscus* to 0.7262 (SE = 0.0577) for *Eurycea* spp. (Table 2.8). These model-average occupancy probabilities were larger than observed occupancy due to the inclusion of detection (p) covariates. Detection probabilities ranged from 0.4824 (SE = 0.0701) for *Desmognathus fuscus* to 0.6787 (SE = 0.0432) for *Eurycea* spp. (Table 2.8).

AIC Comparison of Landscape-Level and Microhabitat Results

In comparison of microhabitat and landscape-level models, the models with empirical support (within Δ AIC of 4) for *D. fuscus* were all microhabitat models (Table 2.10). The **Canopy Model** showed the most empirical support with an AIC weight of 0.2994 (Table 2.10). The percent canopy cover covariate was in five of the six top models for *D. fuscus* and had beta estimates ranging from 1.12 to 2.03 which shows a strong positive relationship to occupancy probability (Table 2.11).

With the exception of the **Non-Mining Model** (a landscape-level model) with a Δ AIC of 2.78, all models with empirical support (within Δ AIC of 4) for *D. monticola* were microhabitat models (Table 2.10). The **Microhabitat Model** was the top ranked model with an AIC weight of 0.2895 (Table 2.10). Within this model, the covariates percent detritus cover and percent stream bank erosion had the largest betas (1.06 and 1.05 respectively), and showed a positive relationship to *D. monticola* occupancy probability (Table 2.12).

Desmognathus ochrophaeus results only showed empirical support for microhabitat models (Table 2.10). The best-ranked model was the **Macrohabitat Model** with an AIC weight of 0.4202 (Table 2.10), and percent canopy cover covariate had the largest beta (1.20) within this model (Table 2.13).

Landscape-level models had empirical support for the *Eurycea* spp. group (Table 2.10). The **HUC-12 Recent Mining Model** is the top model with an AIC weight of 0.5599 (Table 2.10). Within this model, the percent of recent mining within the HUC-12 watershed had a beta of -1.49 showing a negative correlation to *Eurycea* spp. occupancy probabilities (Table 2.14).

Model Averaged Results

Results from the model averaging showed the strongest support for a positive correlation to the percent canopy cover covariate for the three *Desmognathus* species (Table 2.15). Percent detritus cover (0.74) and the number of LWD (0.53) covariates were also positively correlated to *D. monticola*. *Eurycea* spp. showed a negative relationship to the percent of the HUC-12 recently mined covariate and a positive relationship to the percent of the HUC-12 forested covariate (Table 2.15).

Discussion

I found that *Desmognathus* species had similar results with most top models being microhabitat models. Within the microhabitat models, most top models (within a Δ AIC of 4) contained canopy cover as a covariate. Five occupancy models best explained presence of the three *Desmognathus* species. These models were the **Canopy Model**, the **Trees Model**, the **Shading Model**, the **Microhabitat Model**, and the **Macrohabitat Model**. Multiple parameters comprised these models including canopy cover, DBH, tree species richness, herbaceous cover, sapling/shrub density, detritus cover, LWD, and stream bank erosion. Because data were normalized prior to analysis, it is possible to compare the betas, or effect size, of these different parameters and examine correlation trends among models and species groups.

Canopy cover was a parameter in four of the five models. All three *Desmognathus* species showed a strong positive correlation to increased canopy cover. This was also confirmed in both the model averaging analysis where canopy cover was the covariate with the largest support for the three *Desmognathus* species. Additionally, beta values for canopy cover in the top models were large indicating a strong effect size. Past research has also shown strong associations between canopy cover and salamander abundance (Crawford and Semlitsch 2008; Davic and Welsh 2004; Ward et al. 2008). *Desmognathus* salamanders are lungless and therefore are required to constantly have moist skin in order for oxygen exchange across the skin membrane (Petranka 1998). Lungless salamanders have a high risk of desiccation particularly when foraging away from the stream. Canopy cover not only provides cover from solar exposure lowering soil, stream, and air temperatures, it also increases other microhabitat parameters associated with maintaining a cool, moist environment such as detritus cover. Crawford and Semlitsch (2008) found a positive correlation between detritus depth and *D. monticola* and *E. b. cirrigera*. In the southern Appalachians of North Carolina, Harper and Guynn (1999) found more salamanders including *D. ochrophaeus* and *D. aeneus* in moist microhabitats with increased detritus depths.

A closed canopy cover is also often an indication of mature, less-disturbed or more recovered forest conditions. Although forest stand age was not a covariate in this study, *Desmognathus* were positively correlated characteristics often found in more mature forests such as high canopy cover, native tree species dominance, high detritus cover, more LWD, and increased sapling/shrub densities. In the central Appalachians of West Virginia, Moseley et al. (2008) found a positive relationship between *Desmognathus* spp. abundance and time since forest harvest. Ford et al. (2002) observed that *Desmognathus* spp. abundance was most correlated to the basal area within a forest stand in southern Appalachia. Additionally, in a review of North American literature on amphibian ecology and forest management deMaynadier and Hunter

(1995) suggested more salamanders in older forests functionally are an indirect measure of microhabitat conditions such as LWD, detritus cover, and canopy.

Results from my study show that *Desmognathus* were negatively correlated to parameters associated with open areas/grasslands such as increased herbaceous cover and areas dominated by invasive species. Walz (2002) found decreased abundances of *D. fuscus* and *D. ochrophaeus* in agricultural fields and pastures. Wood and Williams (2013b) found lower abundances of *Desmognathus* in reclaimed grassland and shrubland where there was less detritus, lower stem densities, less LWD, less canopy cover, and an increase in invasive herbaceous species such as *Lespedeza* as compared to forested or partially forested sites. Invasive herbaceous species may just indicate recent disturbance, or it may be that invasive plant species do not produce the necessary microhabitat (i.e. leaf litter, cover, and LWD) to provide the cool, moist habitat needed for salamanders (Lemke et al. 2012).

With the exception of the **Non-Mining Model** for *Desmognathus monticola*, large-scale land use models were not significant for the *Desmognathus*. This indicates that local riparian areas are more important for *Desmognathus* than watershed or subwatershed land uses. In southeastern Kentucky, Maigret et al. (2014) found that *Desmognathus* spp. abundances were lower in areas without a timber harvest stream buffer zone than in undisturbed areas or areas with a 7.6 m stream buffer zone. Pearl et al. 2005 found that in the Pacific Northwest wetland and riparian conditions most influenced native salamander species occupancy and that land use within a 1000 m radius had little effect on salamander occupancy. In central Appalachia, Ward et al. (2008) found that riparian habitat conditions were highly influential on stream salamander richness.

The **Non-Mining Model** for *D. monticola* was the only large-scale land use with empirical support for the *Desmognathus* spp. This model consists of the number of houses and length of roads in the subwatershed. Beta results show that the length of road had a much larger effect size than the number of houses. Ward et al. (2008) found a negative correlation between *Desmognathus* abundance and roads in central Appalachia. Impassible culverts may limit upstream salamander movement, and may prevent migration altering salamander community structure upstream of road-stream crossings (Anderson et al. 2014; Ward et al. 2008). deMaynadier and Hunter (2000) found that forested sites without roads generally had twice the number of salamanders than roadside sites. They suggested that movement of both terrestrial and aquatic salamanders may be limited by roads. Road often tend to cause forest fragmentation, which decreases canopy cover and detritus (Ward et al. 2008).

Roads, particularly paved roads, may also alter water chemistry parameters such as conductivity. Runoff from roads containing de-icing salt has been shown to travel over 170 m from the road into wetlands in the Adirondack Mountains (Karraker et al. 2008). In vernal pools within 50 m of roads, Karraker et al. (2008) reported declines in *Ambystoma maculatum* (spotted salamander) abundance thought to be caused by increased salinity from road salt. Conversely, Jones et al. (in press) found no differences in the toxicity of road salt versus NaCl to *Rana clamitans* or *Eurycea bislineata*. They also found no mortality for *E. bislineata* from conductivity until concentrations reached ~10,000 us/cm (Jones et al. in press). Additionally, Izzo (2013) found that road salt had no lethal effect on *D. fuscus* or *E. bislineata* until chloride concentrations exceeded 5,000 mg/l (~ 9,000 us/cm). The conductivity range for this study was 13-1660 us/cm. My study did not find conductivity to significantly affect salamander occupancy, as this metric (in the **Chemistry Model**) only had weak empirical support for one species, *D. ochrophaeus*, with a Δ AIC of 3.53.

Eurycea spp. occupancy results showed that occupancy was negatively associated with the percent mining in the whole HUC-12 watershed. The percent forested in the HUC-12 watershed was also shown to have some empirical support for *Eurycea* spp. and to have a positive correlation to *Eurycea* spp. occupancy. No microhabitat models had any empirical support for *Eurycea* spp. In research from the Piedmont, *Eurycea* has been shown to be negatively correlated to increased impervious surface from urbanization (Miller et al. 2007; Price et al. 2011). These results are similar to the relationship with percent recent mining as none of the other potential mining-related parameters had empirical support (e.g. the **Chemistry Model**, the **Valley Fill Model**, and the **Surface Mining Model**). Previous research has documented decreases in *Eurycea* abundance in areas with high impervious surface (Miller et al. 2007; Minton 1968; Orser & Shure 1972; Willson and Dorcas 2003). Miller et al. (2007) and Willson and Dorcas (2003) both found that watershed disturbance caused declines in *Eurycea* abundances, but that riparian buffer size and quality was not correlated to relative abundance. *Eurycea* are opportunistic generalists with diets largely consisting of pollutant-tolerant benthic macroinvertebrates such as *Chironomids* (Barrett et al. 2012; Burton 1976; Muenz et al. 2008; Petranka 1984). I postulate that *Eurycea* is perhaps better able to tolerate poor water quality and riparian habitat conditions as prey items (i.e. *Chironomids*) are often readily available.

This difference in response of *Eurycea* and *Desmognathus* to microhabitat and landscape-level land use may be a function of movement and habitat selection. Adult *Eurycea* are more terrestrial than the *Desmognathus* species, and *E. bislineata cirrigera* have been shown to migrate over 100 m away from the stream, whereas *D. fuscus* are largely sedentary having small home ranges of 1.4 m^2 to 3 m^2 (Ashton 1975; Barthalmus and Bellis 1972; MacCulloch and Bider 1975; Pertranka 1998). Larval *Eurycea bislineata* have been shown to move instream. Small first-year larvals tend to drift downstream, whereas larger second-year larvals will move both upstream and downstream equally (Bruce 1986; Petranka 1998). The ability of *Eurycea* *bislineata* to move long-distance may also explain why recent mining in the HUC-12 watershed had more of an influence on *Eurycea* occupancy than recent mining in the subwatershed.

Overall, stream salamanders in my study seemed to be either disturbance avoiders (*Desmognathus*) or disturbance tolerant (*Eurycea*). Disturbance avoiders are generally long-lived, large-bodied salamanders that are dependent on forests and are sensitive to riparian disturbances. Disturbance tolerant species often can be characterized as short-lived, small bodied, microhabitat generalists that can withstand riparian land uses (Surasinghe and Baldwin 2015). Species that comprise these groups may change in different ecoregions. For example Surasinghe and Baldwin (2015) found *D. quadramaculatus* and *G. porphyriticus* to be disturbance avoiders in the Blue Ridge ecoregion; however, in the Piedmont ecoregion these species were considered to be disturbance tolerant. Based on results from my study, *Desmognathus* spp. seem to be disturbance avoiders whereas *Eurycea* spp. were disturbance tolerant. Research throughout Appalachia have found that in undisturbed areas *Desmognathus* spp. is the dominant stream salamander group, whereas in disturbed areas *Eurycea* spp. is the dominant stream salamander (Hamilton, 2002; Hyde and Simons, 2001; Resetarits, 1997). In central Appalachia, Ward et al. (2008) found that abundances of disturbance tolerant salamanders (*E. b. cirrigera*) were higher at roadside sites as compared to forested control sites whereas *Desmognathus* spp. had the inverse response. Riparian disturbances, such as roads, may cause stream salamander communities to shift to disturbance tolerant species without changes in overall abundance (Ward et al. 2008). This indicates the need for research to distinguish stream salamanders to species instead of looking at total salamander abundance. It is important to note, however, that even disturbance tolerant species are not totally disturbance-adapted, and these species could still be vulnerable to localized extirpations above some land use threshold.

Predation and competition may also contribute to differences in results between *Eurycea* and *Desmognathus*. Predation may have both direct mortality and indirect behavioral effects on stream salamanders, particularly on *Eurycea* which is the smallest stream salamander with the most predatory salamander species in this study. For example, Barr and Babbitt (2002) found that densities of *E. b. bislineata* were dramatically lower ($9.4/m^2$ as compared to $54/m^2$) when the predatory *Salvelinus fontinalis* (brook trout) was present. Stomach content analysis of predatory salamander species, *G. porphyriticus*, showed that up to 46% of content was other stream salamanders (Bruce 1972). *Eurycea* may be more susceptible to predation from other stream salamanders than from fish predation. In a laboratory study, Resetarits (1997) found survival rates of *Eurycea b. wilderae* was highest when *Salvelinus fontinalis* was present and lowest when *G. porphyriticus* was present. Predation from other stream salamanders may not only decrease survival rates, but may also influence microhabitat selection and behavior. Without predators *E. b. wilderae* were evenly distributed in pools and riffles, however, when *D. quadramaculatus*, a large predator species that prefers riffles, were present *E. b. wilderae* were more abundant in pools than in riffles (Beachy 1993). Keen (1982) found when *D. monticola*, a predator of *D. fuscus*, was present, activity levels of *D. fuscus* dropped significantly. Others have also hypothesized that decreases in some salamander species abundances when predatory fish and/or salamander species are present may not come from predation but rather from competition (Barr and Babbitt 2002; Bruce 2011; Davic 1983; Hairston 1980). However, much of the research that has examined the influence of competition has been field-based without the ability to control for natural variability.

Overall my research found effects of mining on stream salamanders to be equivocal; however this is not to imply that mining does not affect stream salamanders. While the top model for

Eurycea spp. was the **HUC-12 Recent Mining Model**, none of the species in this study had strong empirical support for many of the other mining-related parameters such as the percent active or past mining in the subwatershed, area of valley fill, conductivity, and the number of settling ponds. This was shown not only in the model ranking, but also in the model-averaged results. Research has also shown negative effects of surface mining and valley fills on stream salamander abundance and relative species richness (Hamilton 2002; Muncy et al. 2014; Wood and Williams 2013a; Wood and Williams 2013b). However, these studies all examined the effects of mining versus reference conditions, and the resulting decreases in salamander assemblages may be a reflection of general riparian disturbance and not factors particularly unique to coal mining. My research helped differentiate effects of coal mining and general riparian disturbance on stream salamanders by using a gradient of land uses and disturbances. This gradient of sites showed poor riparian quality lead to declines in *Desmognathus* abundance regardless of the type of large-scale land use associated with that riparian disturbance.

Although stream salamander recovery from coal mining disturbance has not specifically been examined in my study, Hamilton (2002) found that a valley fill stream 18 years post-reclamation had salamander abundances that did not statistically differ from reference sites. This indicates that stream salamander abundances can recover as riparian habitat quality increases post-mining. Wood and Williams (2013a) found decreases in stream salamander abundance in valley fill streams as compared to reference streams; however, species richness did not differ. Because my research focused on presence/absence, it is possible that abundances have decreased due to large-scale land use, but that the species has not been fully extirpated possibly due to refuges provided by high-quality riparian areas. Nonetheless, results from my research confirm that riparian habitat quality and protection is very important for stream salamanders.

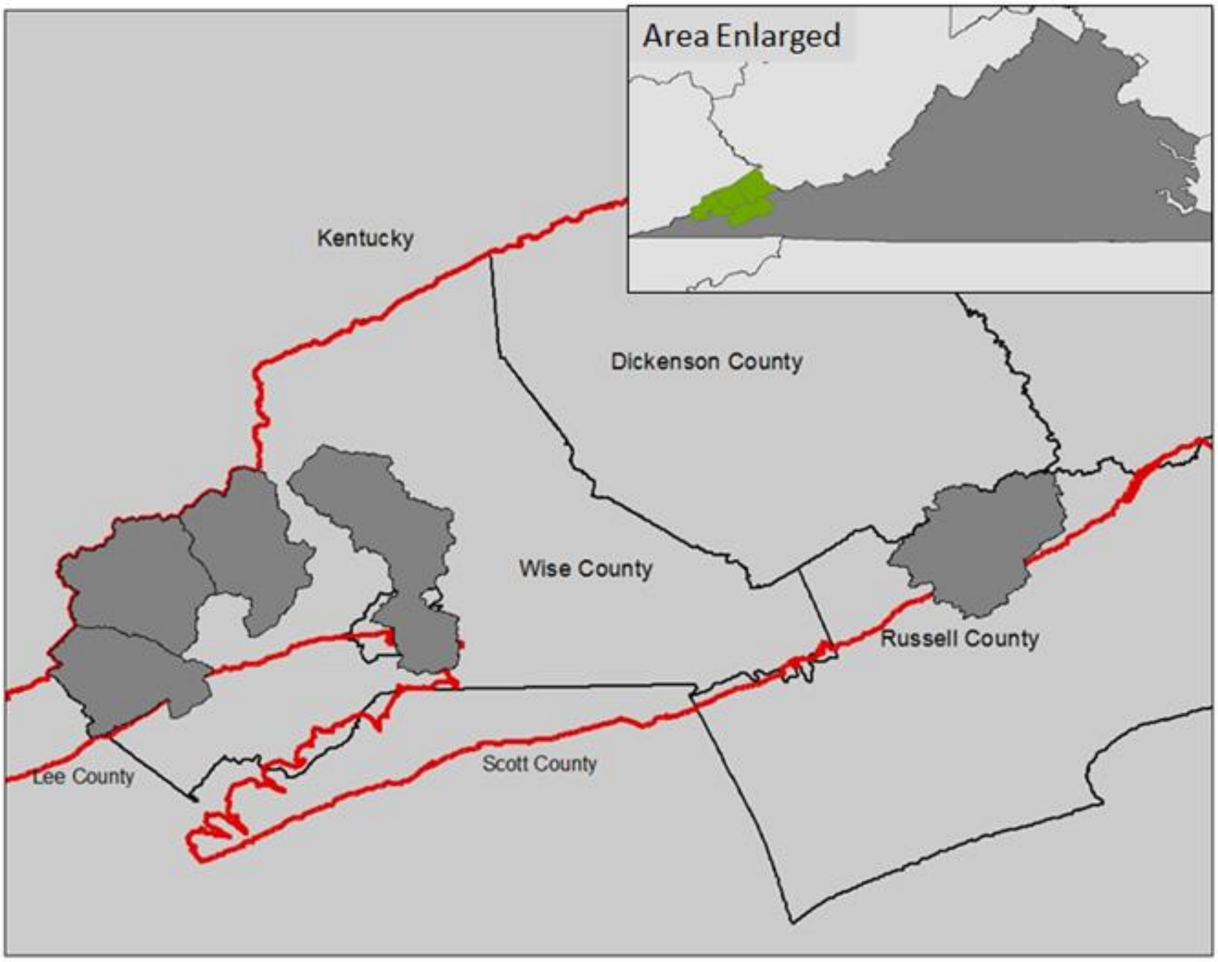

Figure 2.1 Map of the five study watersheds in southwest Virginia (2013), and the relative site locations within the state. The red line shows the southwest coalfield boundary. With the exception of a few square kilometers, all the study watersheds are inside the coalfield boundary of the Cumberland Plateau and Mountains ecoregion.

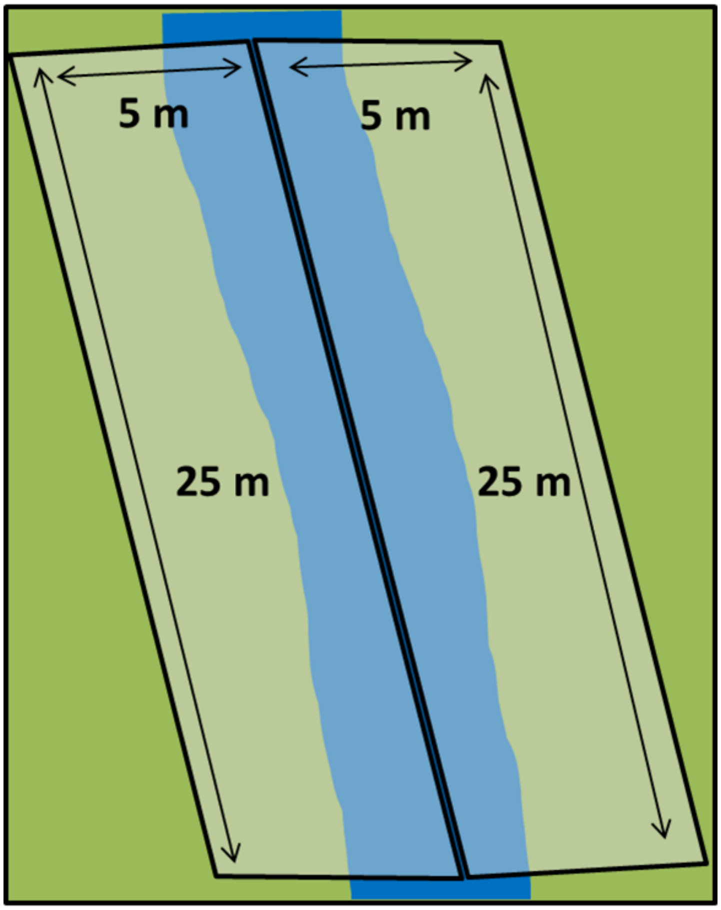

Figure 2.2 Salamander and riparian habitat quadrat placement used in southwest Virginia, 2013. Salamander sampling was done on either the left or right quadrat (determined by a coin flip at each visit). Habitat measurements were taken from the full 25 m x 10 m area.

Table 2.1 Information for the five HUC-12 study watersheds used in 2013 including watershed name, HUC-12 identification number, Virginia County(s) where the watershed is located and the area of the watershed.

Watershed Name	HUC-12	County	Watershed area (sq. km)
Callahan Creek	60102060103	Wise	54.7
Dumps Creek	60102050402	Dickenson; Russell	82.3
Pigeon Creek	60102060104	Wise	58.9
Roaring Fork	60102060101	Wise	66
Rocky Fork	60102050501	Wise	91

Table 2.2 Common and scientific names for the stream salamander species within geographic ranges of the study watersheds (southwest Virginia, 2013). Terrestrial and uncommon salamander species were not included.

Scientific Name	**Common Name**
Notophthalmus viridescens viridescens	Red-spotted Newt
Desmognathus fuscus fuscus	Northern Dusky Salamander
Desmognathus monticola	Seal Salamander
Desmognathus ochrophaeus	Allegheny Mountain Dusky Salamander
Desmognathus welteri	Black Mountain Dusky Salamander
Eurycea bislineata cirrigera	Southern Two-lined Salamander
Eurycea longicauda longicauda	Long-tailed Salamander
Gyrinophilus porphyriticus porphyriticus	Northern Spring Salamander
Pseudotriton montanus diasticus	Midland Mud Salamander
Pseudotriton ruber ruber	Northern Red Salamander

Table 2.3 Parameters measured in the field during summer 2013, southwest Virginia.

Field Parameters	Units	Sampling Events	Additional Description
1. Stream Flow	Categorical	3	Flow Above; Base Flow; Flow Below
2. Soil Temperature	ºC	3	
3. Air Temperature	ºC	3	
4. Water Temperature[1]	ºC	3	
5. Current Weather	Categorical	3	Clear; Cloudy; Rain
6. Past 24-Hour Weather	Categorical	3	Clear; Cloudy; Rain
7. Fish Presence[1]	Categorical	3	1 = Observed; 0 = Not Observed
8. Conductivity[1]	µs/cm	3	Hach water meter (Hach Company, Loveland, CO)
9. Turbidity[1]	cm	3	Depth of secchi tube visibility
10. Stream Width[1]	m	1	Mean of 3 measurements
11. Maximum Stream Depth[1]	cm	1	At base flow
12. Aspect[1]	Degree	1	Compass reading
13. Gradient[1]	Percent	1	Clinometer reading
14. Canopy Cover[1]	Percent	1	Mean of 3 convex densitometer measurements (Noble et al. 2010)
15. Detritus Cover[1]	Percent	1	Mean of 6 estimations (Noble et al. 2010)
16. Large Woody Debris (LWD)[1]	Count	1	Count of LWD (Noble et al. 2010)
17. Stream Bank Erosion[1]	Percent	1	Mean of left and right banks erosion estimations (Noble et al. 2010)
18. Pool Composition[1]	Percent	1	Estimation
19. Substrate Size[1]	cm	1	Mean of 30 measurements (Noble et al. 2010)
20. Stream Embeddedness[1]	Score 1-5	1	Mean of 30 estimations (Noble et al. 2010)
21. Tree Species Richness[1]	Score 1-5	1	Count of native tree spp. and invasive plant spp. (Noble et al. 2010)
22. Tree Diameter (DBH)[1]	cm	1	Number of trees with DBH > 10 cm (Noble et al. 2010)
23. Herbaceous Cover[1]	Percent	1	Mean of 6 estimates (Noble et al. 2010)
24. Sapling/Shrub[1]	Count	1	# Stems with DBH < 10 cm (Noble et al. 2010)
25. Tree Snags[1]	Count	1	# Standing, dead trees (Noble et al. 2010)
26. Cobble/Boulder[1]	Percent	1	Estimate of stream substrate cobble and boulder composition

[1] Parameters used in microhabitat occupancy models

Table 2.4 Parameters derived from GIS analysis, southwest Virginia, summer 2013.

Landscape Covariate	**Unit**	**GIS Data Used**
1. Forested[1]	Percent	Maxwell et al. 2014
2. Past mining[1]	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
3. Recent mining[1]	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
4. Structures[1]	Count	NAIP 2011
5. Roads[1]	Kilometers	NAIP 2011
6. Valley fill[1]	Hectares	Maxwell et al. 2014; DMME 2013
7. Ponds in segment[1]	Count	NAIP 2011
8. Ponds downstream	Count	NAIP 2011
9. HUC12 forested	Percent	Maxwell et al. 2014
10. HUC12 past mining	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
11. HUC12 recent mining	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
12. Subwatershed area	Hectares	NED
13. Stream order		NHD
14. Receiving stream order		NHD
15. Elevation[2]	Meters	NED

[1] Calculated within subwatershed from the salamander sampling location upstream

[2] Elevation was used for microhabitat models. All other covariates in this table were used in the landscape-level analyses

Table 2.5 Detection covariates and data type of covariate used in occupancy analysis of stream salamanders in southwest Virginia, 2013.

Detection Covariate	**Covariate Name**	**Covariate Type**	**Additional Description**
Stream Flow Above Base	Flow Above	Binomial	1 = Above; 0 = Not Above
Stream Flow Below Base	FlowBelow	Binomial	1 = Below; 0 = Not Below
Soil Temperature	SoilT	Continuous	ºC
Air Temperature	AirT	Continuous	ºC
Water Temperature	WaterT	Continuous	ºC
Current Weather Rainy	Rain	Binomial	1 = Rainy; 0 = Not Rainy
Rain in Past 24 Hours	Rain24	Binomial	1= Rainy; 0 = Not Rainy

Table 2.6 Final *a priori* occupancy models for microhabitat covariates used to examine the influence of fine-spatial scale conditions on stream salamanders in southwest Virginia, 2013.

	Predictive Model	**Microhabitat Environmental Parameters**
1.	Null	Intercept
2.	Microhabitat Global	All microhabitat parameters
3.	Size	Stream width; Maximum stream depth
4.	Stream Location	Stream aspect; Gradient; Elevation
5.	Macrohabitat	Aspect; Gradient; Elevation; Canopy cover
6.	Microhabitat	Detritus cover; LWD Count; Stream bank erosion
7.	Stream Comp.	Percent pool
8.	Predation	Fish presence
9.	Chemistry	Conductivity; Water temperature
10.	Sediment	Turbidity; Stream embeddedness; Stream bank erosion
11.	Substrate	Average substrate size; % Cobble and boulder; Stream embeddedness
12.	Canopy	Canopy cover
13.	Trees	Tree species richness; DBH; Canopy cover
14.	Herbaceous	Herbaceous cover; Saplings/shrubs
15.	Shading	Canopy cover; Herbaceous cover; Sapling/shrubs; DBH
16.	Detritus	Detritus cover
17.	Future Detritus	Tree snag count; Canopy cover
18.	Cover	LWD count; Detritus cover; % Cobble and boulder; Stream bank erosion

Table 2.7 *A priori* occupancy models with large-scale land use covariates used to examine the influence of landscape-level conditions on stream salamanders in southwest Virginia, 2013.

	Predictive Models	**Large-scale Environmental Parameters**
1.	Null	Intercept
2.	Global	All landscape-level parameters
3.	Forested	Subwatershed forested
4.	Residential	Structures
5.	Non-Mining	Structures; Roads
6.	Past Mining	Subwatershed past mining
7.	Recent Mining	Subwatershed recent mining
8.	Surface Mining	Subwatershed recent mining; Valley fill; Ponds in segment; Ponds downstream
9.	Valley Fill	Valley fill
10.	Pond	Ponds in segment; Ponds downstream
11.	Mining Streams	Valley fill; Ponds in segment
12.	HUC-12 Forested	HUC-12 watershed forested
13	HUC-12 Past Mining	HUC-12 watershed past mining
14	HUC-12 Recent Mining	HUC-12 watershed recent mining
15	Physical	Subwatershed area; Stream order; Receiving stream order

Table 2.8 Observed occupancy (naïve occupancy), model-averaged estimations of occupancy (Ψ) and detection (p) along with standard errors, and the number of survey replicate requirements (number of site visits required) for the four specie/genus groups (*Desmognathus fuscus*, *Desmogntahus monticola*, *Desmognathus ochrophaeus*, and *Eurycea* spp.) of stream salamanders used for full analysis of microhabitat and landscape-level models, southwest Virginia,2013.

Species Group	**Observed Ψ**	**Estimate of Ψ**	**Ψ SE**	**Estimate of p**	**p *SE***	**Survey Replicate Requirements**
D. fuscus	0.3857	0.4507	0.0743	0.4824	0.0701	3.0
D. monticola	0.5714	0.6002	0.0633	0.6494	0.0496	2.5
D. ochrophaeus	0.5286	0.5486	0.0627	0.6685	0.0508	2.2
Eurycea spp.	0.7000	0.7262	0.0577	0.6787	0.0432	2.8

Table 2.9 Total number of salamanders caught by species along with the number of sampling sites and sampling events in which the species were found in summer 2013, southwest Virginia.

Scientific name	**Total Captured**	**# of Sampling Sites (n = 70)**	**# Sampling Events (n = 207)**
Desmognathus fuscus	82	27	45
Desmognathus monticola	291	40	81
Desmognathus ochrophaeus	227	37	75
Desmognathus welteri	41	18	21
Desmognathus Larval	6	6	6
Eurycea b. cirrigera	24	15	19
Eurycea longicauda	15	7	8
Eurycea Larval	442	42	85
Gyrinophilus porphyriticus	11	10	11
Pseudotriton montanus	5	3	3
Pseudotriton ruber	7	6	6

Table 2.10 Top models for occupancy estimates including the number of parameters in each model (K), Akaike's Information Criterion (AIC) rankings, Δ AIC, and AIC weight (ω_i) for the four groups of stream salamanders, southwest Virginia, 2013. Landscape-level models are in italics.

Occupancy Models	**K**	**AIC**	**Δ AIC**[1]	**ω_i**
Desmognathus fuscus				
Ψ(Canopy), p(Rain24)	4	182.44	0.00	0.2994
Ψ(Trees), p(Rain24)	6	183.49	1.05	0.1771
Ψ(FutureDetritus), p(Rain24)	5	183.53	1.09	0.1736
Ψ(Shading), p(Rain24)	7	184.68	2.24	0.0977
Ψ(Detritus), p(Rain24)	4	184.94	2.50	0.0858
Ψ(Macrohabitat), p(Rain24)	7	185.58	3.14	0.0623
Ψ(Microhabiat), p(Rain24)	6	186.50	4.06	0.0393
Desmognathus monticola				
Ψ(Microhabitat), p(WaterT)	6	223.77	0.00	0.2895
Ψ(Macrohabitat), p(WaterT)	7	224.07	0.30	0.2492
Ψ(Cover), p(WaterT)	7	225.06	1.29	0.1519
Ψ(Canopy), p(Water)	4	226.44	2.67	0.0762
Ψ(NonMining), p(Rain24)	*5*	*226.55*	*2.78*	*0.0721*
Ψ(Trees), p(WaterT)	6	226.60	2.83	0.0703
Ψ(Shading), p(WaterT)	7	227.75	3.98	0.0396
Ψ(FutureDetritus), p(WaterT)	5	228.44	4.67	0.0280
Desmognathus ochrophaeus				
Ψ(Macrohabitat), p(SoilT*Rain)	8	209.92	0.00	0.4202
Ψ(Shading), p(SoilT*Rain)	8	211.93	2.01	0.1538
Ψ(Trees), p(SoilT*Rain)	7	212.01	2.09	0.1478
Ψ(Canopy), p(SoilT*Rain)	5	212.83	2.91	0.0981
Ψ(Chemistry), p(SoilT*Rain)	6	213.45	3.53	0.0719
Ψ(FutureDetritus), p(SoilT*Rain)	6	214.76	4.84	0.0374
Eurycea spp.				
Ψ(HUC12RecentMining), p(Rain)	*4*	*249.84*	*0.00*	*0.5599*
Ψ(HUC12Forested), p(Rain)	*4*	*251.13*	*1.29*	*0.2938*
Ψ(NonMining), p(Rain)	*5*	*255.24*	*5.40*	*0.0376*

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

Table 2.11 Beta estimates and standard errors for each covariate in the top occupancy models (Δ AIC < 4) for *Desmognathus fuscus,* southwest Virginia, 2013.

Occupancy Models and Parameters for *D. fuscus*	**Beta**	**Standard Error**
Ψ(Canopy), p(Rain24)		
Ψ	-0.7056	0.4560
Ψ.CanopyCover	1.8792	0.7157
p	0.7082	0.5054
p.Rain24	-1.0272	0.5211
Ψ(Trees), p(Rain24)		
Ψ	-0.5200	0.4639
Ψ.TreeSppRichness	0.6642	0.4594
Ψ.TreeDiameter	0.1906	0.3930
Ψ.CanopyCover	1.1249	0.7442
p	0.6709	0.5105
p.Rain24	-1.0185	0.5148
Ψ(FutureDetritus), p(Rain24)		
Ψ	-0.7293	0.4707
Ψ.TreeSnags	-0.2925	0.3213
Ψ.CanopyCover	2.0285	0.7642
p	0.7021	0.5038
p.Rain24	-1.0264	0.5178
Ψ(Shading), p(Rain24)		
Ψ	-0.5810	0.4466
Ψ.CanopyCover	1.5537	0.7052
Ψ.HerbaceousCover	-0.6128	0.4393
Ψ.Sapling/Shrub	0.5268	0.4483
Ψ.TreeDiameter	-0.1179	0.4322
p	0.6907	0.5101
p.Rain24	-1.0281	0.5193
Ψ(Detritus), p(Rain24)		
Ψ	-0.1095	0.4772
Ψ.DetritusCover	1.4998	0.6447
p	0.4973	0.5376
p.Rain24	-0.9281	0.5037
Ψ(Macrohabitat), p(Rain24)		
Ψ	-0.7056	0.4679
Ψ.Aspect	0.0463	0.3427
Ψ.Gradient	-0.3509	0.3389
Ψ.Elevation	0.3397	0.3634
Ψ.CanopyCover	1.9643	0.7214
p	0.7170	0.5052
p.Rain24	-1.0312	0.5223

Table 2.12 Beta estimates and standard errors for each covariate in the top occupancy models (Δ AIC < 4) for *Desmognathus monticola*, southwest Virginia, 2013.

Occupancy Models and Parameters for *D. monticola*	**Beta**	**Standard Error**
Ψ(Microhabitat), p(WaterT)		
Ψ	0.7992	0.4566
Ψ.DetritusCover	1.0607	0.4178
Ψ.LWD	0.6710	0.6664
Ψ.StreamBankErosion	1.0507	0.8032
p	0.6066	0.2139
p.WaterT	0.3000	0.3892
Ψ(Macrohabitat), p(WaterT)		
Ψ	0.8477	0.4770
Ψ.Aspect	0.3168	0.3969
Ψ.Gradient	1.6145	0.7554
Ψ.Elevation	-0.0910	0.4254
Ψ.CanopyCover	1.9307	0.5948
p	0.5605	0.2085
p.WaterT	0.5503	0.3114
Ψ(Cover), p(WaterT)		
Ψ	1.4702	0.6431
Ψ.Cobble/Boulder	-0.2582	0.4073
Ψ.LWD	2.2038	1.0469
Ψ.DetritusCover	1.0890	0.4445
Ψ.StreamEmbeddedness	1.0401	0.4963
p	0.5380	0.1983
p.WaterT	0.4790	0.2785
Ψ(Canopy), p(WaterT)		
Ψ	0.4587	0.3536
Ψ.CanopyCover	1.5255	0.4270
p1	0.5676	0.2153
p1.WaterT	0.4507	0.3050
Ψ(NonMining), p(Rain24)		
Ψ	0.0558	0.3723
Ψ.Structures	-0.5203	0.8207
Ψ.Roads	-2.3194	0.7905
p	0.2825	0.3301
p.Rain24	0.5176	0.4029
Ψ(Trees), p(WaterT)		
Ψ	0.6731	0.4303
Ψ.TreeSppRichness	0.6826	0.4914
Ψ.TreeDiameter	0.5116	0.5467
Ψ.CanopyCover	0.7838	0.5269
p	0.5685	0.2136
p.WaterT	0.4595	0.3480

Table 2.12 (Continued)

Occupancy Models and Parameters for *D. monticola*	**Beta**	**Standard Error**
Ψ(Shading), p(WaterT)		
Ψ	0.6076	0.4177
Ψ.CanopyCover	1.2163	0.5029
Ψ.HerbaceousCover	-0.5912	0.3910
Ψ.Sapling/Shrub	-0.3170	0.4881
Ψ.TreeDiameter	0.4571	0.6029
p	0.5663	0.2185
p.WaterT	0.3638	0.3957

Table 2.13 Beta estimates and standard errors for each covariate in the top occupancy models (Δ AIC < 4) for *Desmognathus ochrophaeus*, southwest Virginia, 2013.

Occupancy Models and Parameters for *D. ochrophaeus*	**Beta**	**Standard Error**
Ψ(Macrohabitat), p(SoilT*Rain)		
Ψ	0.3030	0.3220
Ψ.Aspect	0.6174	0.3328
Ψ.Gradient	-0.0700	0.3077
Ψ.Elevation	0.7126	0.3504
Ψ.CanopyCover	1.2020	0.3729
p	0.7670	0.2596
p.SoilT	-0.5662	0.2537
p.Rain	-1.9292	0.7413
Ψ(Shading), p(SoilT*Rain)		
Ψ	0.2868	0.3198
Ψ.CanopyCover	0.5177	0.4049
Ψ.HerbaceousCover	-0.2964	0.3346
Ψ.Sapling/Shrub	-0.5277	0.4192
Ψ.TreeDiameter	0.7922	0.5194
p	0.7529	0.2604
p.SoilT	-0.6009	0.2584
p.Rain	-1.9862	0.7372
Ψ(Trees), p(SoilT*Rain)		
Ψ	0.3083	0.3159
Ψ.TreeSppRichness	0.4186	0.4192
Ψ.TreeDiameter	0.6956	0.4559
Ψ.CanopyCover	0.2222	0.4376
p	0.7589	0.2606
p.SoilT	-0.5858	0.2557
p.Rain	-1.9471	0.7416
Ψ(Canopy), p(SoilT*Rain)		
Ψ	0.2182	0.2906
Ψ.CanopyCover	0.8861	0.3180
p	0.7515	0.2596
p.SoilT	-0.6066	0.2585
p.Rain	-1.9813	0.7375
Ψ(Chemistry), p(SoilT*Rain)		
Ψ	0.5582	0.3361
Ψ.Conductivity	-0.6229	0.3259
Ψ.WaterTemp	-1.5409	0.5724
p	0.6175	0.2474
p.SoilT	-0.6761	0.2478
p.Rain	-1.9810	0.7248

Table 2.14 Beta estimates and standard errors of the covariates in the top occupancy models (Δ AIC < 4) for *Eurycea* spp., southwest Virginia, 2013.

Occupancy Models and Parameters for *Eurycea* spp.	**Beta**	**Standard Error**
Ψ(HUC12RecentMining), p(Rain)		
Ψ	1.5130	0.5883
Ψ.HUC12RecentMining	-1.4876	0.6450
p	0.6052	0.2124
p.Rain	1.0503	0.6620
Ψ(HUC12Forested), p(Rain)		
Ψ	2.3537	1.4586
Ψ.HUC12Forested	2.6829	1.9141
p	0.5556	0.2002
p.Rain	1.0961	0.6578

Table 2.15 Model averaged values (> 0.1) and direction of relationship for individual covariates for stream salamander species occupancy, southwest Virginia, summer 2013. Landscape-level covariates are in italics.

Occupancy Covariate	***D. fuscus***	***D. monticola***	***D. ochrophaeus***	***Eurycea* spp.**
Stream Width	-	-	-	-
Stream Depth	-	-	-	-
Aspect	-	-	0.260(+)	-
Gradient	-	0.402(+)	-	-
Elevation	-	-	0.300(+)	-
Canopy Cover	1.453(+)	0.744(+)	0.738(+)	-
Detritus Cover	0.240(+)	0.499(+)	-	-
LWD	0.107(+)	0.529(+)	-	-
Bank Erosion	-	0.307(+)	-	-
Pool Composition	-	-	-	-
Fish Presence	-	-	-	-
Conductivity	-	-	-	-
Water Temperature	-	-	0.111(-)	-
Turbidity	-	-	-	-
Substrate Size	-	-	-	-
Embeddedness	-	0.158(+)	-	-
Tree Spp. Richness	0.131(+)	-	-	-
DBH	-	-	0.225(+)	-
Herbaceous Cover	-	-	-	-
Sapling/Shrub	-	-	-	-
Tree Snags	-	-	-	-
Cobble/Boulder	-	-	-	-
Forested	-	-	-	-
Past Mining	-	-	-	-
Recent Mining	-	-	-	-
Structures	-	-	-	*0.181(+)*
Roads	-	*0.195(-)*	-	-
Valley Fill	-	-	-	-
PondsIN	-	-	-	-
PondsDS	-	-	-	-
HUC12-Forested	-	-	-	*0.788(+)*
HUC12-Past Mining	-	-	-	-
HUC12-Recent Mining	-	-	-	*0.833(-)*
Subwatershed Area	-	-	-	*0.145(+)*
Stream Order	-	-	-	-
Receiving Stream Order	-	-	-	-

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Appendix A

Table A.1 Habitat variables thought to vary temporally that were measured at each salamander sampling event (n = 3) in southwest Virginia, 2013 along with justification/rationale and supporting literature for each parameter.

Parameter	Justification	Supporting Literature
Current Weather	Covariate thought to influence detection	Bailey et al. 2004; Kroll et al. 2010; Williams 2003
Weather: Past 24 hrs.	Covariate thought to influence detection	Bailey et al. 2004; Kroll et al. 2010; Williams 2003
Fish Presence	Fish presence linked to decreases in salamander abundance	Lowe et al. 2004; Petranka 1998; Sih et al. 1992
Stream Flow Condition	Covariate thought to influence detection	
Turbidity	Most work has looked at the effect of sedimentation on salamanders, but limited data on suspended sediment.	Barbour et al. 1999
Air Temperature	Covariate that may influence detection and occupancy	Kroll et al. 2010; Miller et al. 2007; Pawlik 2008; Whiteleather 2001; Williams 2003
Water Temperature	Covariate that may influence detection and occupancy	Bury and Corn 1989; Kroll et al. 2010; Miller et al. 2007; Pawlik 2008; Whiteleather 2001
Soil Temperature	Covariate that may influence detection and occupancy	Crawford 2007; Pawlik 2008; Williams 2003
Conductivity	Conductivity affects are largely unknown for salamanders	Gore 1983; Miller et al. 2007; Whiteleather 2001

Table A.2 Habitat parameters thought to be relatively stabled and sampled once during the 2013 field season (southwest Virginia) along with justification/rationale and supporting literature for each parameter.

Parameter	Justification	Supporting Literature
Wetted Stream Width	Stream size is thought to affect salamander occupancy	Bury and Corn 1989; Miller et al. 2007; Moseley et al. 2008; Stoddard 2001; Williams 2003
Maximum Stream Depth	Stream size and pools are thought to affect salamander occupancy	Bury and Corn 1989: Miller et al. 2007
Riffle/Run/Pool Composition	Positive correlation between salamanders and pool habitat	Moseley et al. 2008; Murphy and Hall 1981; Sih et al. 1992; Stoddard 2001; Welsh and Ollivier 1998
Stream Aspect	Influences heat load/UV levels.	Hairston 1949; Stoddard 2001; Whiteleather 2001
Stream Gradient	Research has shown gradient influences sedimentation and salamander abundance	Bury and Corn 1989; Crawford 2007; Hall et al. 1978; Moseley et al. 2008; Stoddard 2001; Whiteleather 2001
Stream Substrate Size	Part of USACE protocol.	Noble et al. 2010
Stream Substrate Composition	Salamander abundances have been correlated to rock/cobble substrate	Bury and Corn 1989; Davic and Orr 1987; Miller et al. 2007; Mitchell 1999; Moseley et al. 2008; Pawlik 2008; Petranka 1998; Redmond 1980; Welsh and Ollivier 1998
Stream Bank Erosion	Indication of unstable stream banks and sedimentation	Miller et al. 2007; Orser and Shure 1972; Noble et al. 2010
Large Woody Debris	Important cover object for salamanders.	Miller et al. 2007; Moseley et al. 2008; Murphy and Hall 1981; Stoddard 2001; Noble et al. 2010
Tree Snags	Part of USACE protocol. Successional stage	deMaynadier and Hunter 1998; Noble et al. 2010
DBH of Riparian Trees	Part of USACE protocol. Successional stage	Moseley et al. 2008; Noble et al. 2010
Riparian Tree Species Richness	Part of USACE protocol. Help to determine riparian vegetation quality	Noble et al. 2010

Table A.2 (Continued)

Parameter	Justification	Supporting Literature
Saplings, Shrubs	Part of USACE protocol.	Stoddard 2001; Noble et al. 2010
Riparian Herbaceous Cover	Part of USACE protocol.	Stoddard 2001; Noble et al. 2010
Detritus Cover	Part of USACE protocol. Important for salamander microhabitat	Miller et al. 2007; Noble et al. 2010
Substrate Embeddedness	Research has shown sedimentation to cause declines in salamander abundance.	Bury and Corn 1989; Hawkins et al. 1983; Lowe and Bolger 2002; Lowe et al. 2004; Moseley et al. 2008;Noble et al. 2010;Welsh and Ollivier 1998
Canopy Cover	Research has shown relationship between salamander abundance and canopy cover	Bury and Corn 1989; Crawford 2007; Miller et al. 2007; Stoddard 2001; Noble et al. 2010; Whiteleather 2001

Table A.3 AIC comparison of all landscape-level and microhabitat models for *Desmognathus fuscus*, southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Occupancy Models for *D. fuscus*	**K**	**AIC**	**Δ AIC[1]**	**ω_i**
Ψ(Canopy), p(Rain24)	4	182.44	0.00	0.2994
Ψ(Trees), p(Rain24)	6	183.49	1.05	0.1771
Ψ(FutureDetritus), p(Rain24)	5	183.53	1.09	0.1736
Ψ(Shading), p(Rain24)	7	184.68	2.24	0.0977
Ψ(Detritus), p(Rain24)	4	184.94	2.50	0.0858
Ψ(Macrohabitat), p(Rain24)	7	185.58	3.14	0.0623
Ψ(Microhabiat), p(Rain24)	6	186.50	4.06	0.0393
Ψ(MicrohabitatGlobal), p(Rain24)	13	186.55	4.11	0.0383
Ψ(Cover), p(Rain24)	7	188.96	6.52	0.0115
Ψ(Herbaceous), p(Rain24)	5	191.43	8.99	0.0033
Ψ(Chemistry), p(Rain24)	5	191.49	9.05	0.0032
Ψ(Forested), p(FlowAbove)	*4*	*192.01*	*9.57*	*0.0025*
Ψ(RecentMining), p(FlowAbove)	*4*	*193.11*	*10.67*	*0.0014*
Ψ(SurfaceMining), p(FlowAbove)	*7*	*193.35*	*10.91*	*0.0013*
Ψ(MiningStreams), p(FlowAbove)	*5*	*194.56*	*12.12*	*0.0007*
Ψ(LandscapeGlobal), p(FlowAbove)	*17*	*194.62*	*12.18*	*0.0007*
Ψ(Ponds), p(FlowAbove)	*5*	*195.66*	*13.22*	*0.0004*
Ψ (ValleyFill), p(FlowAbove)	*4*	*195.67*	*13.23*	*0.0004*
Ψ (NonMining), p(FlowAbove)	*5*	*196.74*	*14.30*	*0.0002*
Ψ (StreamComp), p(Rain24)	4	197.15	14.71	0.0002
Ψ (MicrohabitatNull), p(Rain24)	3	198.22	15.78	0.0001
Ψ (Predation), p(Rain24)	4	198.57	16.13	0.0001
Ψ (HUC12Forested), p(FlowAbove)	*4*	*198.59*	*16.15*	*0.0001*
Ψ(Size), p(Rain24)	5	199.26	16.82	0.0001
Ψ(HUC12RecentMining), p(FlowAbove)	*4*	*199.48*	*17.04*	*0.0001*
Ψ(Physical), p(FlowAbove)	*6*	*200.59*	*18.15*	*0.0000*
Ψ(Residential), p(FlowAbove)	*4*	*200.86*	*18.42*	*0.0000*
Ψ(PastMining), p(FlowAbove)	*4*	*201.47*	*19.03*	*0.0000*
Ψ(Macrohabitat), p(FlowAbove)	3	201.51	19.07	0.0000
Ψ(Substrate), p(Rain24)	6	201.98	19.54	0.0000
Ψ(Sediment), p(Rain24)	6	202.86	20.42	0.0000
Ψ(HUC12PastMining), p(FlowAbove)	*4*	*202.99*	*20.55*	*0.0000*
Ψ(Location), p(Rain24)	6	203.01	20.57	0.0000

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

Table A.4 AIC comparison of all landscape-level and microhabitat models for *Desmognathus monticola*, southwest Virginia, summer 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Occupancy Models for *D. monticola*	**K**	**AIC**	**Δ AIC**[1]	ω_i
Ψ(Microhabitat), p(WaterT)	6	223.77	0.00	0.2895
Ψ(Macrohabitat), p(WaterT)	7	224.07	0.30	0.2492
Ψ(Cover), p(WaterT)	7	225.06	1.29	0.1519
Ψ(Canopy), p(Water)	4	226.44	2.67	0.0762
Ψ(NonMining), p(Rain24)	*5*	*226.55*	*2.78*	*0.0721*
Ψ(Trees), p(WaterT)	6	226.60	2.83	0.0703
Ψ(Shading), p(WaterT)	7	227.75	3.98	0.0396
Ψ(FutureDetritus), p(WaterT)	5	228.44	4.67	0.0280
Ψ(Detritus), p(WaterT)	4	229.28	5.51	0.0184
Ψ(LandscapeGlobal), p(Rain24)	*17*	*232.89*	*9.12*	*0.0030*
Ψ(Sediment), p(WaterT)	6	235.26	11.49	0.0009
Ψ(Chemistry), p(WaterT)	5	237.44	13.67	0.0003
Ψ(Physical), p(Rain24)	*6*	*240.63*	*16.86*	*0.0001*
Ψ(Herbaceous), p(WaterT)	5	240.73	16.96	0.0001
Ψ(ValleyFill), p(Rain24)	*4*	*241.65*	*17.88*	*0.0000*
Ψ(MiningStreams), p(Rain24)	*5*	*241.72*	*17.95*	*0.0000*
Ψ(Residential), p(Rain24)	*4*	*242.10*	*18.33*	*0.0000*
Ψ(Forested), p(Rain24)	*4*	*242.15*	*18.38*	*0.0000*
Ψ(RecentMining), p(Rain24)	*4*	*242.31*	*18.54*	*0.0000*
Ψ(Size), *p*(WaterT)	5	242.56	18.79	0.0000
Ψ(Ponds), p(Rain24)	*5*	*242.76*	*18.99*	*0.0000*
Ψ(SurfaceMining), p(Rain24)	*7*	*243.24*	*19.47*	*0.0000*
Ψ(Predation), p(WaterT)	4	244.82	21.05	0.0000
Ψ(LandscapeNull), p(Rain24)	*3*	*245.69*	*21.92*	*0.0000*
Ψ(MicrohabitatNull), p(WaterT)	3	246.91	23.14	0.0000
Ψ(HUC12Forested), p(Rain24)	*4*	*247.07*	*23.30*	*0.0000*
Ψ(HUC12PastMining), p(Rain24)	*4*	*247.45*	*23.68*	*0.0000*
Ψ(Location), p(WaterT)	6	247.52	23.75	0.0000
Ψ(PastMining), p(Rain24)	*4*	*247.54*	*23.77*	*0.0000*
Ψ(HUC12RecentMining), p(Rain24)	*4*	*247.58*	*23.81*	*0.0000*
Ψ(StreamComp), p(WaterT)	4	248.53	24.76	0.0000
Ψ(Substrate), p(WaterT)	6	250.33	26.56	0.0000

[1] Models with a $\Delta \text{AIC} < 2$ are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

Table A.5 AIC comparison of all landscape-level and microhabitat models for *Desmognathus ochrophaeus*, southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Occupancy Models for *D. ochrophaeus*	**K**	**AIC**	**Δ AIC[1]**	**ω_i**
Ψ(Macrohabitat), p(SoilT*Rain)	8	209.92	0.00	0.4202
Ψ(Shading), p(SoilT*Rain)	8	211.93	2.01	0.1538
Ψ(Trees), p(SoilT*Rain)	7	212.01	2.09	0.1478
Ψ(Canopy), p(SoilT*Rain)	5	212.83	2.91	0.0981
Ψ(Chemistry), p(SoilT*Rain)	6	213.45	3.53	0.0719
Ψ(FutureDetritus), p(SoilT*Rain)	6	214.76	4.84	0.0374
Ψ(Microhabitat), p(SoilT*Rain)	7	214.94	5.02	0.0341
Ψ(Detritus), p(SoilT*Rain)	5	217.50	7.58	0.0095
Ψ(Herbaceous), p(SoilT*Rain)	6	217.73	7.81	0.0085
Ψ(Predation), p(SoilT*Rain)	5	219.19	9.27	0.0041
Ψ(Cover), p(SoilT*Rain)	8	219.33	9.41	0.0038
Ψ(MicrohabitatNull), p(SoilT*Rain)	4	220.09	10.17	0.0026
Ψ(Size), p(SoilT*Rain)	6	220.19	10.27	0.0025
Ψ(Sediment), p(SoilT*Rain)	7	221.39	11.47	0.0014
Ψ(StreamComp), p(SoilT*Rain)	5	221.57	11.65	0.0012
Ψ(Location), p(SoilT*Rain)	7	221.68	11.76	0.0012
Ψ(MicrohabitatGlobal), p(SoilT*Rain)	14	222.57	12.65	0.0008
Ψ(Substrate), p(SoilT*Rain)	7	225.50	15.58	0.0002
*Ψ(MiningStreams), p(SoilT*Rain)*	*6*	*225.51*	*15.59*	*0.0002*
*Ψ(Ponds), p(SoilT*Rain)*	*6*	*225.70*	*15.78*	*0.0002*
*Ψ(HUC12PastMining), p(SoilT*Rain)*	*5*	*225.75*	*15.83*	*0.0002*
*Ψ(Residential), p(SoilT*Rain)*	*5*	*226.10*	*16.18*	*0.0001*
*Ψ(Physical), p(SoilT*Rain)*	*7*	*226.60*	*16.68*	*0.0001*
*Ψ(RecentMining), p(SoilT*Rain)*	*5*	*226.87*	*16.95*	*0.0001*
*Ψ(NonMining), p(SoilT*Rain)*	*6*	*227.60*	*17.68*	*0.0001*
*Ψ(LandscapeNull), p(SoilT*Rain)*	*4*	*227.64*	*17.72*	*0.0001*
*Ψ(ValleyFill), p(SoilT*Rain)*	*5*	*228.41*	*18.49*	*0.0000*
*Ψ(HUC12Forested), p(SoilT*Rain)*	*5*	*228.57*	*18.65*	*0.0000*
*Ψ(SurfaceMining), p(SoilT*Rain)*	*8*	*228.77*	*18.85*	*0.0000*
*Ψ(HUC12RecentMining), p(SoilT*Rain)*	*5*	*228.79*	*18.87*	*0.0000*
*Ψ(Forested), p(SoilT*Rain)*	*5*	*228.91*	*18.99*	*0.0000*
*Ψ(PastMining), p(SoilT*Rain)*	*5*	*229.12*	*19.20*	*0.0000*

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

Table A.6 AIC comparison of landscape-level and microhabitat models for *Eurycea* spp., southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Occupancy Models for *Eurycea* spp.	**K**	**AIC**	**Δ AIC[1]**	**ω_i**
Ψ(HUC12RecentMining), p(Rain)	*4*	*249.84*	*0.00*	*0.5599*
Ψ(HUC12Forested), p(Rain)	*4*	*251.13*	*1.29*	*0.2938*
Ψ(NonMining), p(Rain)	*5*	*255.24*	*5.40*	*0.0376*
Ψ(Residential), p(Rain)	*4*	*255.29*	*5.45*	*0.0367*
Ψ(Physical), p(Rain)	*6*	*255.73*	*5.89*	*0.0295*
Ψ(Predation), p(SoilT*Rain)	5	258.50	8.66	0.0074
Ψ(Sediment), p(SoilT*Rain)	7	259.04	9.20	0.0056
Ψ(RecentMining), p(Rain)	*4*	*259.71*	*9.87*	*0.0040*
Ψ(HUC12PastMining), p(Rain)	*4*	*259.73*	*9.89*	*0.0040*
Ψ(Forested), p(Rain)	*4*	*260.26*	*10.42*	*0.0031*
Ψ(MicrohabitatNull), p(SoilT*Rain)	4	260.97	11.13	0.0021
Ψ(StreamComp), p(SoilT*Rain)	5	261.11	11.27	0.0020
Ψ(Location), p(SoilT*Rain)	7	261.48	11.64	0.0017
Ψ(PastMining), p(Rain)	*4*	*261.61*	*11.77*	*0.0016*
Ψ(LandscapeNull), p(Rain)	*3*	*261.76*	*11.92*	*0.0014*
Ψ(Detritus), p(SoilT*Rain)	5	261.79	11.95	0.0014
Ψ(Herbaceous), p(SoilT*Rain)	6	262.67	12.83	0.0009
Ψ(Size), p(SoilT*Rain)	6	262.73	12.89	0.0009
Ψ(Canopy), p(SoilT*Rain)	5	262.74	12.90	0.0009
Ψ(Chemistry), p(SoilT*Rain)	6	262.91	13.07	0.0008
Ψ(Trees), p(SoilT*Rain)	7	263.35	13.51	0.0007
Ψ(Macrohabitat), p(SoilT*Rain)	8	263.44	13.60	0.0006
Ψ(Shading), p(SoilT*Rain)	8	263.45	13.61	0.0006
Ψ(SurfaceMining), p(Rain)	*7*	*263.48*	*13.64*	*0.0006*
Ψ(ValleyFill), p(Rain)	*4*	*263.68*	*13.84*	*0.0006*
Ψ(Ponds), p(Rain)	*5*	*264.13*	*14.29*	*0.0004*
Ψ(FutureDetritus), p(SoilT*Rain)	6	264.54	14.70	0.0004
Ψ(Substrate), p(SoilT*Rain)	7	265.05	15.21	0.0003
Ψ(Microhabitat), p(SoilT*Rain)	7	265.28	15.44	0.0002
Ψ(MiningStreams), p(Rain)	*5*	*265.65*	*15.81*	*0.0002*
Ψ(Cover), p(SoilT*Rain)	8	266.57	16.73	0.0001
Ψ(MicrohabitatGlobal), p(SoilT*Rain)	14	272.85	23.01	0.0000

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

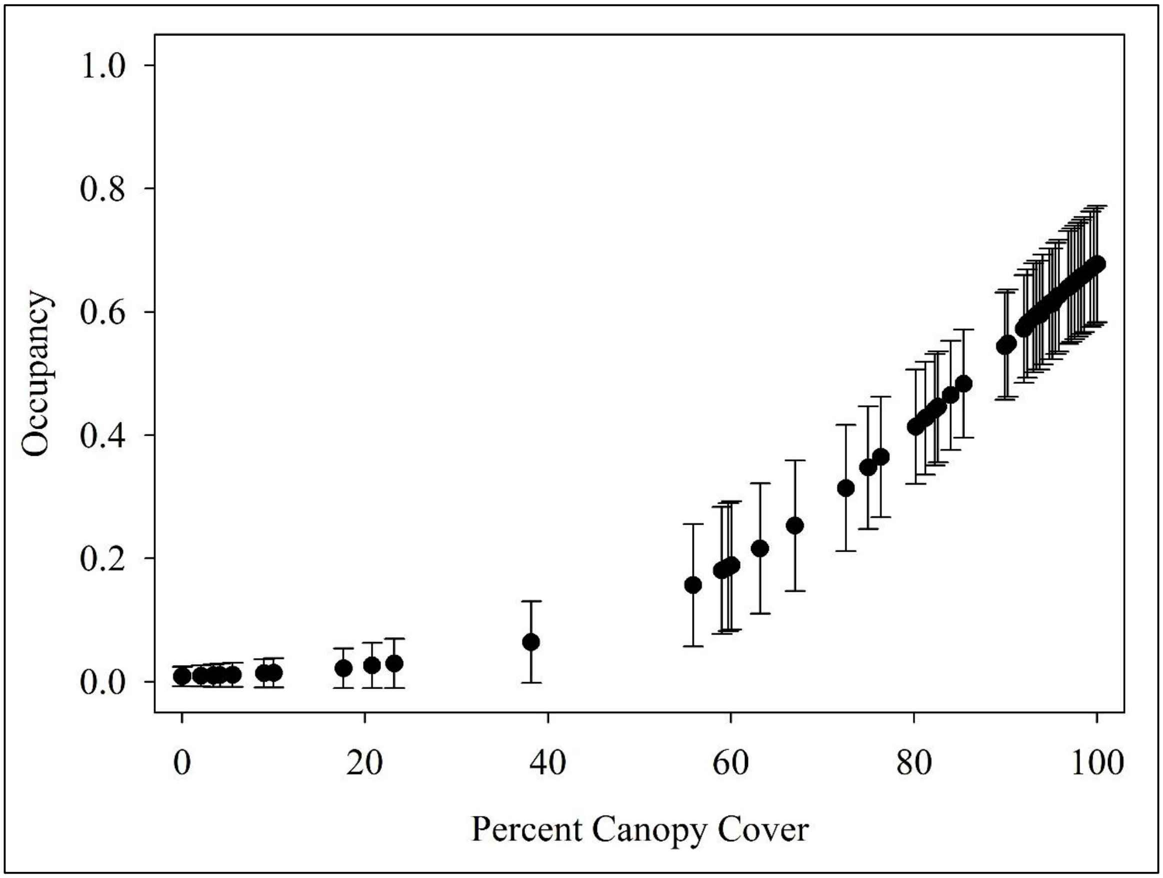

Figure A.1 Individual site estimates and standard errors of occupancy (Ψ) and percent canopy cover for *Desmognathus fuscus*, southwest Virginia, summer 2013.

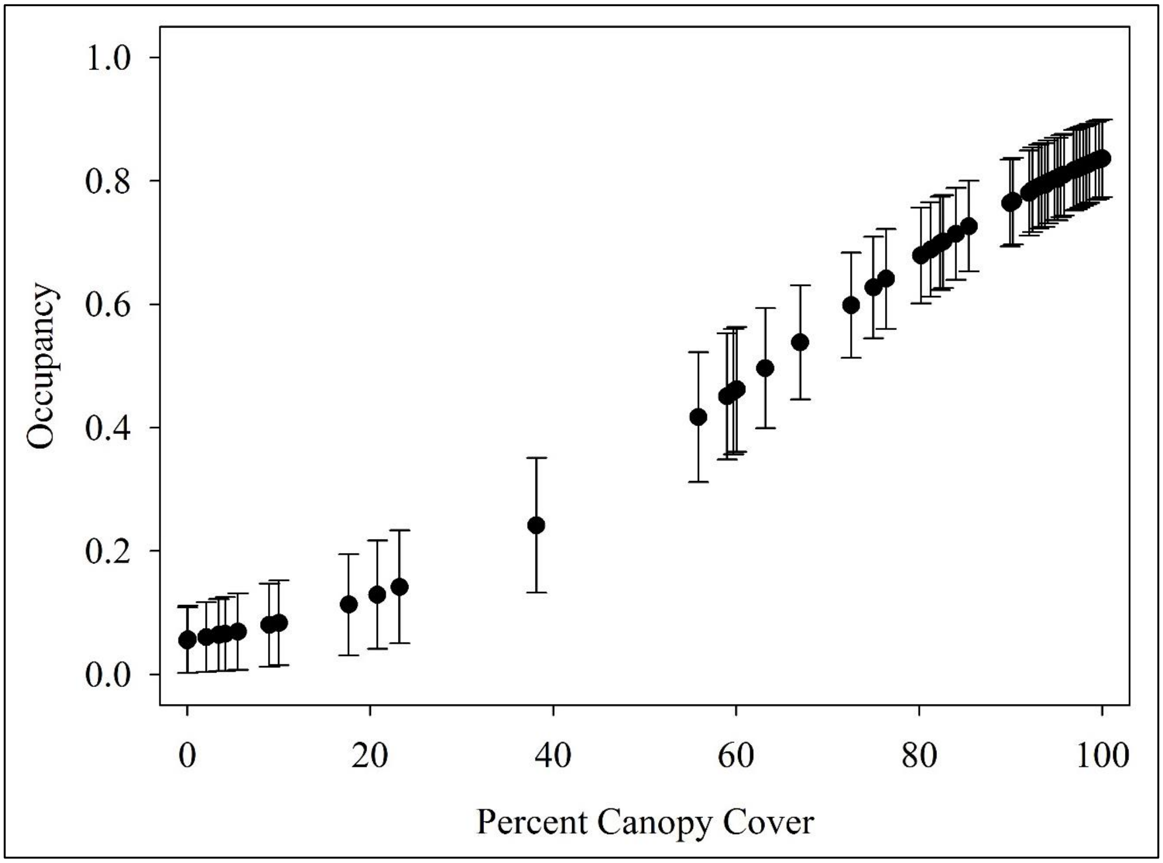

Figure A.2 Individual site estimates and standard errors of occupancy (Ψ) and percent canopy cover for *Desmognathus monticola*, southwest Virginia, summer 2013.

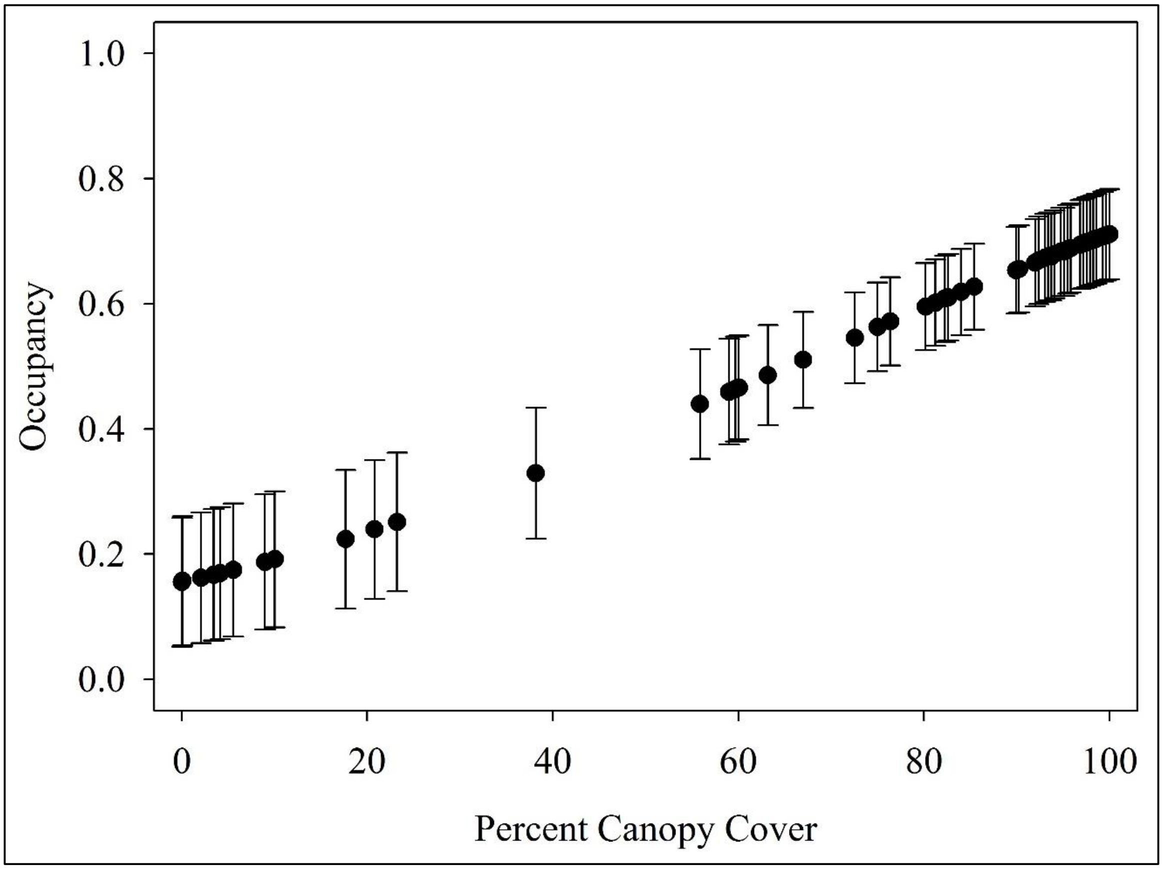

Figure A.3 Individual site estimates and standard errors of occupancy (Ψ) and percent canopy cover for *Desmognathus ochrophaeus,* southwest Virginia, summer 2013.

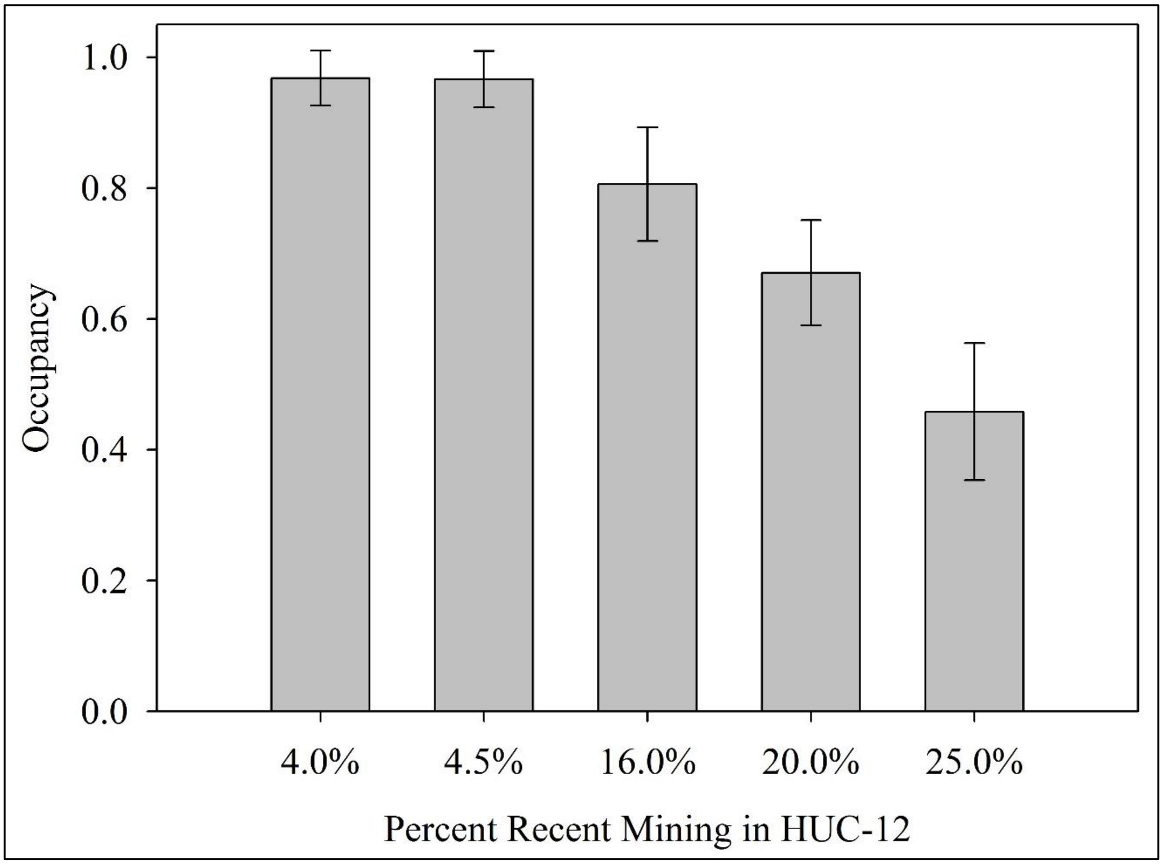

Figure A.4 Occupancy estimates (Ψ) and standard errors for the percent recent mining in each of the five HUC-12 watersheds for *Eurycea* spp., southwest Virginia, summer 2013.

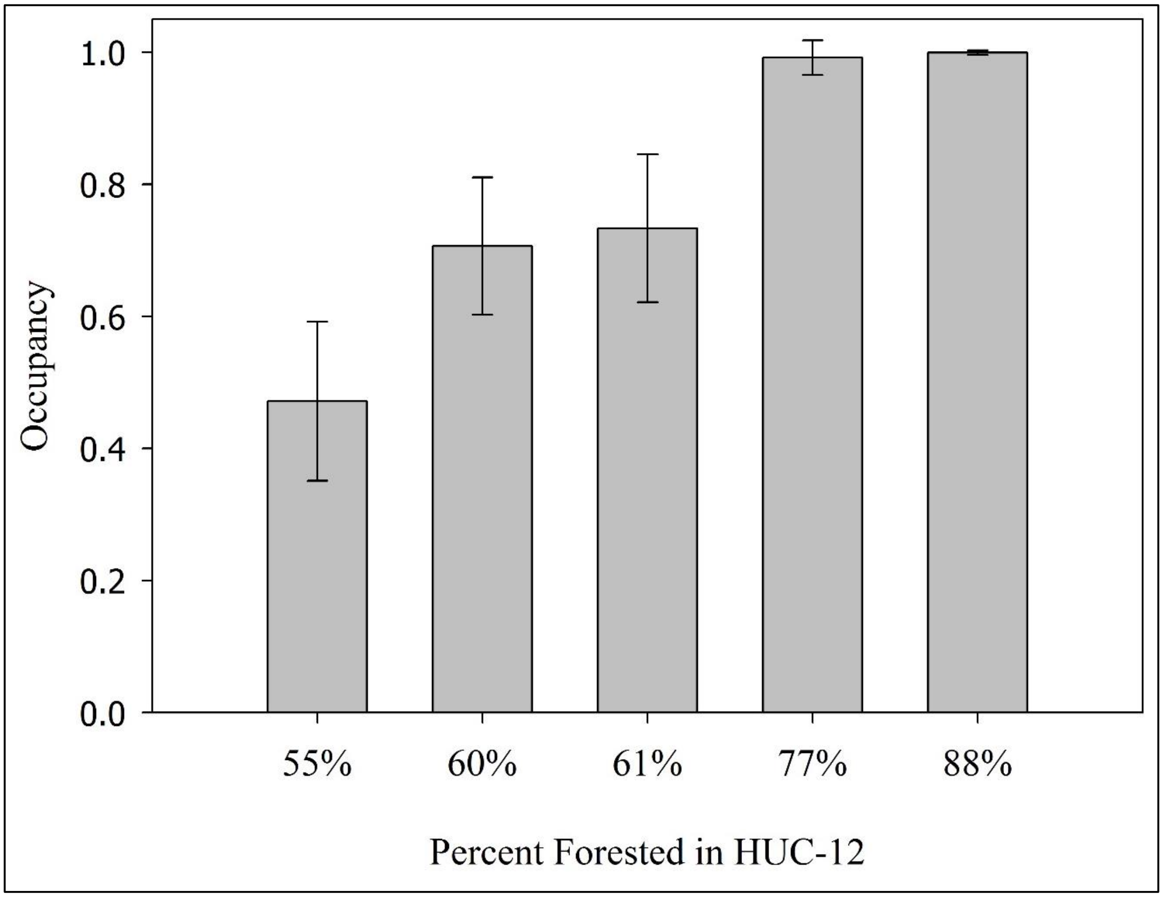

Figure A.5 Occupancy estimates (Ψ) and standard errors for the percent forested in each of the five HUC-12 watersheds for *Eurycea* spp., southwest Virginia, summer 2013.

Chapter 3

Effects of Microhabitat and Land Use on Stream Salamander Abundance in the Southwest Virginia Coalfields

Abstract

Large-scale land uses such as residential wastewater discharge and coal mining practices, particularly surface coal extraction and associated valley fills, are of particular ecological concern in central Appalachia. Identification and quantification of both large-scale land use and microhabitat alterations to ecosystems are a necessary first-step in mitigation of negative consequences to biota. In central Appalachian headwater streams absent of fish, salamanders are the dominant, most abundant vertebrate predator providing a significant intermediate trophic role. Stream salamander species are considered to be sensitive to aquatic stressors and environmental alterations, and past research has shown linkages among microhabitat parameters, large-scale land use such as urbanization and logging, and salamander abundances. However, there is little information examining these linkages in the coalfields of central Appalachia. In the summer of 2013, I visited 70 sites (sampled three times each) in the southwest Virginia coalfields to collect salamanders and quantify stream and riparian microhabitat parameters. Using an *information-theoretic* framework I compared the effects of microhabitat and large-scale land use on salamander abundances. My findings indicate that dusky salamander (*Desmognathus* spp.) abundances are more correlated to microhabitat parameters such as canopy cover than to subwatershed land uses. Brook salamander (*Eurycea* spp.) abundances show strong negative associations to the suspended sediments and stream substrate embeddedness. Neither *Desmognathus* spp. nor *Eurycea* spp. abundances were influenced by water conductivity.

Management implications of these findings include protection of riparian habitats and erosion prevention and control.

Introduction

Currently, the extraction of Appalachian coal by mountaintop mining is common as it is cost-effective and allows for mining of shallow coal seams that cannot safely be mined by underground methods (Craynon et al. 2012). Although mountaintop mining accounts for 37% of coal production in Appalachia (National Mining Association 2012), it is the most controversial because of its disproportionately large environmental footprint. Mountaintop coal extraction uses explosives to blast rock layers (overburden) that sit above relatively shallow coal seams (Griffith et al. 2012). This process causes overburden to swell and take up much more volume as haul-able rock pieces than the original compacted geologic material. The amount of overburden produced is too costly to remove from the mine site for disposal, so overburden typically is deposited into adjacent valleys. Although it is clear that small headwater streams are buried during the mining process altering physical stream habitat, water chemistry, and aquatic biota, the longitudinal spatial scale of indirect disturbance is more challenging to empirically describe. The United States Environmental Protection Agency (EPA) estimates that in central Appalachia, 10% of the region's land area (500,000 ha) has been affected by mountaintop mining, and over 2,000 km of headwater stream habitat buried by valley fills (USEPA 2011).

Along with direct physical stream loss from valley fills, typically watershed hydrology, stream chemistry, and biological communities also are altered from pre-mined conditions (Starnes and Gasper 1995; Pond et al. 2008; Griffith et al. 2012). Physical alteration of streams resulting from the mining processes such as dredging, burial of headwaters, and dam construction clearly alter stream hydrology. Changes from surface coal mining to upland terrestrial processes such as clearing of vegetation, increased impervious surfaces, and changes in topography, can also alter stream hydrology (Simmons et al. 2008; Merriam et al. 2011). As watersheds become increasingly altered from surface coal mining, vegetation is replaced by bare, impervious surfaces that causes the flow regime (hydrology) of a stream system to respond more quickly to rain events becoming “flashy”, and results in more frequent and intense stream flood events (Starnes and Gasper 1995; Price et al. 2006). These high flow events can lead to channelization of the stream along with eroded, unstable stream banks (Booth et al. 2004). Instream habitat is buried by sediment as terrestrial erosion continues. If erosion is not controlled, terrestrial processes can be significantly altered by the loss of soil and nutrients (Baker 1985). Beyond the alterations to the physical stream habitat, these terrestrial changes also both directly and indirectly influence water chemistry. All of these factors have had both acute and chronic effects on specific aquatic biota in mining-influenced streams (Northingham et al. 2011; Pond 2012).

Within the central Appalachian coalfields, efforts to minimize and mitigate environmental effects of surface coal mining have been complicated by incongruent restoration goals and poorly validated post-mine monitoring protocols that are unable to quantify ecosystem change relative to system recovery. Traditionally, research and monitoring of aquatic ecosystems has focused on the effects of land-use practice disturbances on water chemistry and macroinvertebrate assemblage changes. However, the ability to distinguish natural stream variability from the additive effects of anthropogenic influences on aquatic biotic communities is currently limited. Furthermore, more research is needed to examine the multiple temporal and spatial dynamics of the biological recovery of a degraded aquatic system (Harding et al. 1998; Adams et al. 2002; Niemi et al. 2004; Hartman et al. 2005; Milman and Short 2008; Simmons et al. 2008; Northingham et al. 2011). Adding to this is the realization that even identifying and quantifying aquatic ecosystem structure and function of undisturbed systems can be very difficult (Simmons et al. 2008). Fiscal and personnel constraints are often the limiting factors to widespread or more comprehensive sampling. It is, however, imperative to establish scientifically-backed bench marks of success for watershed remediation efforts and/or coal mine reclamation. Biological assessment using a suite of taxa may provide the most extensive view of aquatic changes including declines in stream health as well as stream recovery (Sweeten et al. 2013).

In central Appalachian headwater streams absent of fish, salamanders are the dominant, most abundant vertebrate predator (Burton and Likens 1975; Hairston 1986; Davic and Welsh 2004). In undisturbed areas salamanders have been shown to have relatively stable populations with high densities of up to 1.4/m^2 in central Appalachia (Kleeberger 1984; Hairston and Wiley 1993). Stream salamander species are considered to be sensitive to both aquatic and environmental stressors, and low stream salamander abundance has been shown to be closely linked to anthropogenic terrestrial watershed alteration (Petranka et al. 1993; Willson and Dorcas 2003). Salamanders in central Appalachian headwater streams provide significant intermediate roles in trophic cycling, feeding on small prey such as benthic macroinvertebrates and in turn, salamanders are an important prey item for larger vertebrates (Petranka et al. 1993; Davic and Welsh 2004; OHEPA 2012).

Research relating land use in the eastern United States on salamanders largely has focused on forestry practices. In the central and southern Appalachians, considerable data exist linking forest harvesting and younger stand age classes to terrestrial salamander declines or depressed abundances at least in the short-term (Ash 1988; Ash 1997; Crawford and Semlitsch 2008; Ford et al. 2002; Harper and Guynn 1999; Harpole and Haas 1999; Petranka et al. 1993; Reichenbach and Sattler 2007). Though less extensive, similar research in the central and southern Appalachians on stream salamanders also has suggested analogous outcomes between harvesting and stand age with stream salamander abundances (Moseley et al. 2008; Peterman and Semlitsch 2009). At the smaller spatial scales, logging-related alterations to riparian quality (such as decreased canopy cover, leaf litter depth and soil moisture) along with declines in physical stream conditions (such as stream substrate class, substrate embeddedness, riffle/run/pool composition) are thought to negatively influence stream salamander populations (Crawford and Semlitsch 2008; Moseley et al. 2008; Peterman and Semlitsch 2009). Additionally, low stream salamander abundance has been shown to be closely linked to terrestrial watershed alterations (Petranka et al. 1993; Willson and Dorcas 2003). Stream salamander abundance may be reduced, even to the point of local extirpation, by decreases in physiochemical stream and riparian conditions from watershed land uses such as mining, urbanization, and timber harvesting that cause changes in pH, streambed sedimentation, suspended sediments, and water and soil temperature (Gore 1983; Willson and Dorcas 2003; Welsh et al. 2005; Moseley et al. 2008). This illustrates the need for understanding the dynamics among salamander abundance, land use, and microhabitat.

Land managers and researchers have begun to examine stream salamander response to habitat alterations from either surface or underground coal mining; however, many environmental effects of coal mining on stream salamanders remain poorly quantified. Most of the amphibian research on mined lands has focused on constructed settling ponds and use and recolonization by pond salamanders such as those in the family Ambystomidae (Fowler et al. 1985; Jansen et al. 2004; Kirk 2000; Lacki et al. 1992; Loughman 2005; Turner and Fowler 1981). Some research has examined the effect of acid mine drainage on stream salamanders (Freda 1986; Middlekoop et al. 1998; Whiteleather 2001); however, acid mine drainage is not an issue throughout the entire Appalachian coalfields, such as in southwest Virginia (Herricks and Cairns 1974; Minear and Tschantz 1976). Often salamander research in the coalfields has been a small part of a larger study that examined multiple taxa (Carrozzino 2009; Loughman 2005). This research tended to be more observational and descriptive in nature.

In the southern West Virginia coalfields, Hamilton (2002) examined the effects of valley fills on stream salamander relative abundance using three streams with valley fills and two reference streams. The younger two valley fill streams (15 years and 5 years post-reclamation) were found to have lower stream salamander abundance than the reference streams. However, the oldest valley fill (18 years post-reclamation) had salamander abundances that did not differ statistically from the reference sites. Wood and Williams (2013a) examined stream salamander abundances in three reference streams compared to three streams with valley fills in southern West Virginia. They found lower relative abundances in the valley fill streams (N = 780 captures) than in the reference streams (N = 1563 captures), however species richness was the same in valley fill and reference streams. Muncy et al. (2014) compared stream salamander occupancy and species richness between 11 first-order valley fill streams and 12 first-order reference streams at the University of Kentucky’s Robinson Forest. Estimated occupancy probabilities and species richness, along with forest cover and the number of cover objects, were higher in reference streams than in valley fill streams. Still inferences were limited by the small range of study location conditions, microhabitat and riparian conditions, and lack of large-scale, landscape analysis of watershed land use and legacy land use. The results found in these three studies tend to be similar to the effects of other anthropogenic land uses i.e. forestry, agriculture, and urbanization when compared to reference stream conditions.

It is expected that many of the aquatic effects of coal mining in southwest Virginia will closely mirror known effects of forestry and urbanization land use such as increased sedimentation, changes in stream flow and thermal regimes, and decreased riparian and water quality.

However, it is important to quantify the relationship of stream salamander abundances to a gradient of environmental conditions and land uses in central Appalachia in order to investigate the possibility of developing a salamander index of biotic integrity. The main objective of my study was to investigate these critical correlations between salamander abundances and habitat parameters at both a microhabitat and a landscape-level along a gradient of conditions in order to determine if more research is warranted for the development of a salamander index of biotic integrity for central Appalachia. These foundational correlations were examined with an *information-theoretic* modeling approach using microhabitat data collected in the field along with a Geographic Information System (GIS) analysis of landscape-level land use.

Methods

Regional Description

My study sites are located in the coalfields of southwest Virginia in Wise, Russell, and Dickenson Counties. This area is part of the Cumberland Plateau (Ecoregion 69d), a sub-region of the Central Appalachian Mountains (Omernik 1987). This region is also commonly known as the Northern Cumberland Plateaus and Mountains Ecoregion (Anderson et al. 2001; Bailey 1995). Topography is characterized by steep mountains with narrow valleys with a mean peak elevation of 760 m (Woodward and Hoffman 1991). Most soils in this region are Utisols, Alfisols, and Inceptisols (McNab and Avers 1995). Mean annual precipitation is about 1150 mm with a mean temperature of 13°C (McNab and Avers 1995; Woodward and Hoffman 1991). Due to the steep topography, this region tends to have a high density of small/medium-sized streams (McNab and Avers 1995). It is estimated that 93 % of the Cumberland Plateau is forested, whereas 4 % of the region is agricultural/open area (VDGIF 2005). Coal mining (both surface and underground), forestry, agriculture, industry, and residential areas (many with inadequate sewage infrastructure) are the dominant land uses (VDGIF 2005). The forested areas are characterized by a diverse mix of hardwood and conifers (Woodward and Hoffman 1991). Common tree species include red oak (*Quercus rubra*), white oak (*Quercus alba*), pignut hickory (*Carya glabra*), red maple (*Acer rubrum*), yellow poplar (*Liriodendron tulipifera*), American beech (*Fagus grandifolia*), basswood (*Tilia Americana*), and white pine (*Pinus strobus*) (Hamilton 2002; McNab and Avers 1995).

Historical Mining

Settlers had known about the coal resources in central Appalachia as early as 1750, however, similar to logging, it was not until after the arrival of railroads in the region in the late 1880s that large-scale mining began. Mechanization processes in the 1920s increased productivity and extraction rates, and as a result mining operations became larger and altered more land in the Appalachians. In southwest Virginia, surface coal mining began in the early 1940s (Hibbart 1990). Throughout most of this time, environmental effects and reclamation of post-mined lands were largely ignored. Until 1939 when West Virginia enacted a law requiring mine land reclamation, there had been no state or federal legislation to regulate post-mine land recovery in the region (Starnes and Gasper 1995). Since the enactment of the Clean Water Act (CWA) in 1972 and the Surface Mining Control and Reclamation Act (SMCRA) in 1976, most scientific focus has been placed on monitoring efforts to regulate mining activities, environmental alterations, and reclamation of post-mined lands (Starnes and Gasper 1995). Under SMCRA, each state may establish their own regulatory and total maximum daily load (TMDL) standards as long as they are "no less effective than" Federal standards under the CWA (Craynon et al. 2012). In Virginia, the Virginia Department of Environmental Quality (VADEQ) is responsible for ensuring the compliance of mining operations under the CWA. In order to determine stream impairment, the VADEQ monitors water chemistry parameters, and developed the Virginia

Stream Condition Index (VA-SCI), a multi-metric benthic macroinvertebrate assessment protocol to identify biological impairment of streams (Burton and Gerritsen 2003).

Site Selection

Within my study area, I selected five 12-digit Hydrologic Unit Codes (HUC-12) watersheds in southwest Virginia (Figure 3.1; Table 3.1). These five watersheds are similar in area, located within the Cumberland Plateau and Mountains Region, and have active coal mining along with other land uses. I divided streams within these watersheds into segments by their stream order, and gave each stream segment in each watershed a unique identification number. I randomly selected first- or second-order stream segments for salamander sampling sites; however, in some cases I used best professional judgment to select sites that had landowner access and that I considered safe to sample. Throughout the five HUC-12 watersheds, I sampled a total of 70 sites (N = 70). Due to a lack of accessible sites on Pigeon Creek and Dumps Creek, these watersheds had 10 salamander sampling locations. Roaring Fork, and Rocky Fork had 15 salamander sampling locations, and there were 20 sites on Callahan Creek.

To accurately estimate detection rates, I visited 67 of the 70 three times in 2013 (Bailey et al. 2004; MacKenzie and Royle 2005). Due to access issues, I was only able to sample three of the sites twice in 2013. At each sampling site location, a 25 m long by 5 m wide quadrat was placed parallel to the stream with the stream center as the right or left edge of the quadrat (Figure 3.2; Hairston 1986; Jung et al. 2000; Kucken et al. 1994). Right or left quadrat placement was determined randomly using a coin flip. I hand-captured adult salamanders by overturning all rocks, detritus, and logs within the 25 m x 5 m quadrat at each sampling site. I identified all adult salamanders to species in the field and immediately released to within 2 m of capture location. A D-frame dip net was used to sample in-stream habitat (Davic 1983; Gore 1983). All larval salamanders were removed by hand from the dip net, placed in a bucket of fresh stream water for identification, and then released within 2 m of the capture location. Due to the difficulty of identifying larval salamanders to the species level, I identified larval salamanders to genus.

Field Parameters

Along with adult and larval salamander sampling, I collected physical stream attributes, water chemistry, and ambient conditions were collected from a quadrat 25 m x 10 m that combined the left and right sides of the salamander quadrat (Figure 3.2). Overall, the parameters in this study are a compilation of parameters past research has suggested to effect salamander abundance and/or occupancy. I sampled parameters thought to have high temporal variability such as stream flow condition, soil air and water temperatures, and water chemistry, using the appropriate gear at each of the three site visits (See Table 3.3 for details). I measured parameters that were likely to remain constant over the summer such as percent canopy cover, tree species richness, and large woody debris counts, once during the field season. Descriptions of all parameters collected in the field for my study are provided in Table 3.3.

GIS Analysis

I assembled multiple GIS layers including the 2011 National Land Cover Dataset (NLCD), the 2011 National Agriculture Imagery Program (NAIP images, historical topographical maps, land use cover data from Maxwell et al. (2014), mining GIS data from the Virginia Department of Mines Mineral and Energy (DMME) for analysis in ArcMap (ESRI, Redlands, California) to determine land use at both the subwatershed (the watershed from the above the sampling point) and HUC-12 scale (Table 3.4). I measured watershed area, stream order of sampled streams, length of stream segment, and stream order of receiving streams. Land use was classified as percent of each watershed with recent mining (active mining permits and areas still barren), past mining, and forested. Additionally I measured area of valley fills and kilometers of roads. I also counted the number of structures and the number of constructed ponds in each subwatershed.

Statistical Analysis

To assess salamander abundance along with detection probabilities in an *information-theoretic* framework, I used the Program PRESENCE software (available for download at www.mbr-pwrc.usgs.gov/software/presence). Program PRESENCE was developed to examine and rank multiple hypotheses using an *information-theoretic* approach with Akaike's Information Criterion selection (AIC) and maximum likelihood to determine the best-fit model for capture data (Bailey et al. 2007; Kroll et al. 2010). In particular, I used the "Royle Repeated Count" models (also known as N-mixture models) within Program PRESENCE to estimate salamander abundances from repeated site counts (Royle 2004).

The AIC approach uses species data along with habitat data to examine the best-fit relationships between a species and a suite of measured parameters MacKenzie et al. (2002; 2006). This technique best analyzes observational research by helping to avoid the over-fitting of models that is oftentimes associated when correlating a large number of habitat parameters to species abundance. N-mixture models use abundances from multiple site visits and fit the data to a Poisson distribution. The mean of the Poisson distribution is the mean estimate of abundance and is denoted as lambda (λ). The best-fit models and associated mean abundance (λ) estimates produced in the data analysis provide foundational and currently unknown estimates of abundance for a given species at a site with certain microhabitat or landscape-level conditions. N-mixture models use This N-mixture modeling technique has been widely used to estimate abundances on a number of taxa ranging from salamanders in the eastern United States to birds in Switzerland to Nile crocodiles (*Crocodulus nilticus*) in Egypt (Dodd and Dorazio 2004; Kery 2008; Maigret et al. 2014; Shirley et al. 2012).

I used a two-step method to determine which detection covariates to include for each species group in the abundance analysis (Burnham and Anderson 2002). Detection is important in N-mixture model analysis in order to get the best estimates salamander abundance by accounting for changing environmental factors (i.e., weather, stream flow, and temperature) that might influence capture rates. I ran *a priori* detection covariates for each species group against the null (intercept) model. All detection covariates that had an AIC smaller than the null were then run for all combinations to determine the best detection covariate for each species group. I then used this detection covariate for the abundance analysis. Utilizing this two-step allows a reduction in the total number of models in the final analysis. For this study, I analyzed seven detection covariates (Table 3.5).

Model Selection

I developed 18 *a priori* models for microhabitat abundance analysis (Table 3.6). These models contained the 21 parameters denoted in Table 3.3 plus the elevation parameter in Table 3.4. Incorporating 14 GIS-derived covariates (Table 3.4), I developed 15 landscape-level *a priori* models (Table 3.7). For both sets of models, I grouped covariates based on biologically relevant information or recommendations from available literature (Ford et al. 2002; Moseley et al. 2008; Petranka et al. 1993; Willson and Dorcas 2003). All combinations of the parameters were not considered due to the large number of models this would have created and the subsequent unreliable outputs (Burnham and Anderson 2002; Dillard et al. 2008). For both sets of models, I normalized continuous variables prior to analysis using Program PRESENCE.

AIC Comparison

After analysis of microhabitat and landscape-level models was completed, I did a comparison of microhabitat and landscape-level AIC values and weights in order to examine the relative importance of the two model groups. I calculated delta AIC values by subtracting the AIC of each model from the AIC of the model with the smallest AIC value. AIC weights were calculated using the equation (Burnham and Anderson 2002):

$$AIC_{Weight} = \frac{Exp(-0.5 * \Delta AIC)}{Summed\ Value\ of\ all\ models\ Exp(-0.5 * \Delta AIC)}$$

Model Averaging

In order to examine the strength of individual covariates across models, I calculated model averages for each covariate using the equation (Burnham and Anderson 2002):

$$\hat{\theta} = \sum_{i=1}^{R} w_i\, \hat{\theta_\iota}$$

where w_i is the AIC weight and $\hat{\theta}$ is the beta estimate of the covariate.

Results

Species/Genus Selection for Analysis

In my study, I detected nine species of aquatic salamanders during the 207 surveys. Due to the large number of *Eurycea* spp. larval salamanders (449 individuals) and the small number of adult *Eurycea* (39 individuals), I combined larval and adult *Eurycea longicauda* and *Eurycea bislineata cirrigera* salamanders to the genus-level for the *Eurycea* spp. group. Based on results from Sweeten (2015), the four salamander groups that had enough statistical power for occupancy analysis were also used for analysis of abundance: *D. fuscus, D. monticola, D. ochrophaeus, and Eurycea* spp. Species or genus groups that were detected too infrequently to be included in the occupancy analysis were *Desmognathus welteri*, *Desmognathus* larval, *Eurycea* adults, *Eurycea bislineata cirrigera*, *Eurycea longicauda*, *Gyrinophilus porphyriticus*, *Pseudotriton montanus*, *Pseudotriton ruber*, and the combined genus group of *Pseudotriton* and *Gyrinophilus* (Table 3.8).

Of the salamander groups used in analysis, total abundances observed across all sampling events ranged from 82 *Desmognathus fuscus* to 481 *Eurycea* spp. (Table 3.9). Mean captures per site per visit ranged from 0.396 for *D. fuscus* to 1.525 for *Eurycea* spp. (Table 3.9). As expected, estimated model averaged abundances (λ) were higher than observed mean capture rates and ranged from 1.043 (SE = 0.193) for *Desmognathus fuscus* to 4.596 (SE = 0.556) for *Eurycea* spp. (Table 3.9). These model-average abundances were larger than observed abundance due to the inclusion of detection (p) covariates.

AIC Comparison of Landscape-Level and Microhabitat Results

In comparison of microhabitat and landscape-level models, the models with empirical support (within Δ AIC of 4) for *D. fuscus* were all microhabitat models (Table 3.10). The **Future Detritus Model** showed the most empirical support with an AIC weight of 0.3177 (Table 3.10). The percent canopy cover covariate was in 4 of the 5 top models for *D. fuscus* and had beta estimates ranging from 1.37 to 1.76, which shows a strong positive relationship to abundance estimates (Table 3.11).

Both models with empirical support (within Δ AIC of 4) for *Desmognathus monticola* were microhabitat models (Table 3.10). The **Shading Model** was the top ranked model with an AIC weight of 0.8273 (Table 3.10). Both top models for *D. monticola* contained the percent canopy cover covariate, which had the largest betas (1.14 in the **Shading Model** and 1.05 in the **Trees**

Model) of all covariates in the models and showed a positive relationship to *D. monticola* abundance estimates (Table 3.12).

Desmognathus ochrophaeus results only showed empirical support for the microhabitat **Shading Model** with an AIC weight of 0.9795 (Table 3.10). The percent canopy cover covariate had the largest beta (1.20) within this model (Table 3.13).

The only model with empirical support for *Eurycea* spp. was the **Sediment Model** (a microhabitat model) with an AIC weight of 1.00 (Table 3.10). Within this model, the turbidity covariate, the stream substrate embeddedness covariate, and the stream bank erosion covariate all had negative betas (-0.8164, -0.4372, and -0.3201 respectively) showing a negative correlation to *Eurycea* spp. abundance estimates (Table 3.14).

Model Averaged Results

Results from the model averaging showed the strongest support for a positive correlation to the percent canopy cover covariate for the three *Desmognathus* species (Table 3.15). Percent herbaceous cover had a negative relationship to both *D. monticola* and *D. ochrophaeus* abundances. *Eurycea* spp. showed a negative relationship to turbidity, stream substrate embeddedness, and stream bank erosion (Table 3.15).

Discussion

I found that *Desmognathus* species had similar results with all top models being microhabitat models. Within the microhabitat models, most top models (within a Δ AIC of 4) contained canopy cover as a covariate. Four models best explained abundances of the three *Desmognathus* species. These models were the **Shading Model,** the **Trees Model**, the **Future Detritus Model**, and the **Canopy Model**. Multiple parameters made up these models including canopy cover, DBH, tree species richness, herbaceous cover, sapling/shrub density, and tree snags. Because data were normalized prior to analysis, it is possible to compare the betas, or effect sizes, of these different parameters and examine correlation trends among models and species groups.

Canopy cover was a parameter in all four of the models. All three *Desmognathus* species showed a strong positive correlation to increased canopy cover. This was also confirmed in both the model averaging analysis where canopy cover was the covariate with the largest support for the three *Desmognathus* species. Additionally, beta values for canopy cover in the top models were large indicating a strong effect size. Past research has also shown strong associations between canopy cover and salamander abundance (Crawford and Semlitsch 2008; Davic and Welsh 2004; Ward et al. 2008). *Desmognathus* salamanders are lungless and therefore are required to constantly have moist skin in order for oxygen exchange across the skin membrane (Petranka 1998). Lungless salamanders have a high risk of desiccation particularly when foraging away from the stream. Canopy cover not only provides cover from solar exposure lowering temperature, it also increases other microhabitat parameters associated with maintaining a cool, moist environment such as detritus cover. Crawford and Semlitsch (2008) found a positive correlation between detritus depth and *D. monticola* and *E. b. cirrigera*. In the southern Appalachians of North Carolina, Harper and Guynn (1999) found more salamanders including *D. ochrophaeus* and *D. aeneus* in moist microhabitats with increased detritus depths.

A closed canopy cover is also often an indication of mature, less-disturbed or more recovered forest conditions. Although forest stand age was not a covariate in this study, *Desmognathus* were positively correlated characteristics often found in more mature forests such as high canopy cover, native tree species dominance, and increased sapling/shrub densities. In the central Appalachians of West Virginia, Moseley et al. (2008) found a positive relationship between *Desmognathus* spp. abundance and time since forest harvest. Ford et al. (2002) observed that

Desmognathus spp. abundance was most correlated to increased basal area within a forest stand in southern Appalachia. Additionally, in a review of North American literature on amphibian ecology and forest management deMaynadier and Hunter (1995) suggested more salamanders in older forests functionally are an indirect measure of microhabitat conditions such as LWD, detritus cover, and canopy.

Results from my study show that *Desmognathus* were negatively correlated to disturbed, open area parameters such as increased herbaceous cover and areas dominated by invasive species. Walz (2002) found decreased abundances of *D. fuscus* and *D. ochrophaeus* in agricultural fields and pastures. Wood and Williams (2013b) found lower abundances of *Desmognathus* in reclaimed grassland and shrubland where there was less detritus, lower stem densities, less LWD, less canopy cover, and an increase in invasive herbaceous species such as *Lespedeza* as compared to forested or partially forested sites. Invasive herbaceous species may just indicate recent disturbance, or it may be that invasive plant species do not produce the necessary microhabitat (i.e. leaf litter, cover, and LWD) to provide the cool, moist habitat needed for salamanders (Lemke et al. 2012).

Large-scale land use models were not significant for the *Desmognathus*. This indicates that local riparian effects and conditions are more important for *Desmognathus* than watershed or subwatershed land uses. In southeastern Kentucky, Maigret et al. (2014) found that *Desmognathus* spp. abundances were lower in areas without a timber harvest stream buffer zone than in undisturbed areas or areas with a 7.6 m stream buffer zone. Pearl et al. 2005 found that in the Pacific Northwest wetland and riparian conditions most influenced native salamander species and that land use within a 1000 m radius had little effect on salamanders. In the central

Appalachians, Ward et al. (2008) found that riparian habitat conditions were highly influential on stream salamander richness.

Eurycea spp. abundance was negatively correlated to factors associated with sedimentation including stream embeddedness, bank erosion, and turbidity. Previous research has shown decreases in stream salamander abundance caused by effects of sedimentation, particularly increases in stream substrate embeddedness (Brannon and Purvis 2008; Lowe and Bolger 2002; Peterman and Semlitsch 2009). For example, Brannon and Purvis (2008) found a significant ($p \leq 0.01$) negative correlation between sediment depth and salamander abundances in the southern Appalachians. Stream embeddedness is thought to reduce the amount of interstitial space available for breeding, feeding, and cover from predators (Lowe and Bolger 2002; Lowe et al. 2004). In New Hampshire, adult *G. porphyriticus* decreased with increased substrate embeddedness, particularly with the presence of predatory fish such as the brook trout (*Salvelinus fontinalis*) indicating the importance of habitat created from interstitial spaces that is often lost when a stream substrate becomes embedded (Lowe and Bolger 2002; Lowe et al. 2004). In urban streams in Georgia, Orser and Shure (1972) found that increases in both substrate embeddedness and stream bank erosion caused decreases in stream salamander densities. Still, the direct effects of suspended sediment or turbidity on stream salamanders has not been examined (Keitzer and Goforth 2012). Because stream salamanders are sight-feeders, it is thought that suspended sediment will have similar negative impacts on stream salamanders as found with other site-feeding taxa such as smallmouth bass larvae (*Micropterus dolomieu*). In a laboratory study, Sweeten and McCreedy (2002) found decreased growth and survival rates of smallmouth bass fry in suspended sediment concentrations as low as 11.4 mg/l. However, it should be a research priority to determine the effects of suspended sediment of both larval and adult salamanders as elevated levels of suspended sediment are undoubtedly a universal problem across land uses.

Although the **Sediment Model** is a microhabitat model, both turbidity and stream substrate embeddedness are often a result of erosion from upstream watershed land uses such as agriculture, urbanization, and surface coal mining (Orser and Shure 1972: Price et al. 2011). Surface coal mining, at least in the short-term, creates vast areas of bare, impervious surfaces that leads to increased sedimentation of streams (Minear and Tschantz 1976; Mitsch et al. 1983). This may explain differences in results between the abundance estimates and the occupancy analysis where the **HUC-12 Recent Mining Model** had the most empirical support (Sweeten 2015). Research has documented decreases in *Eurycea* abundances with increased impervious surface from urbanization similar to the results with the occupancy **HUC-12 Recent Mining Model (Sweeten 2015b)** as none of the other potential mining-related parameters, such as the **Chemistry Model**, the **Valley Fill Model**, the **Recent Mining Model**, and the **Surface Mining Model,** had empirical support in either the occupancy or abundance analyses (Miller et al. 2007; Minton 1968; Orser & Shure 1972; Willson and Dorcas 2003). Miller et al. (2007) and Willson and Dorcas (2003) both found that watershed disturbance caused declines in *Eurycea* abundances, but that riparian buffer size and quality was not correlated to relative abundance.

This difference in response of *Eurycea* and *Desmognathus* to microhabitat and landscape-level land use may be a function of movement and habitat selection. Adult *Eurycea* are more terrestrial than the *Desmognathus* species, and *E. bislineata cirrigera* have been shown to migrate over 100 m away from the stream whereas *D. fuscus* are largely sedentary having home ranges of 1.4 m^2 to 3 m^2 (Ashton 1975; Barthalmus and Bellis 1972; MacCulloch and Bider 1975; Pertranka 1998). Larval *Eurycea bislineata* have been shown to move instream. Small first-year larvals tend to drift downstream whereas larger second-year larvals will move both upstream and downstream equally (Bruce 1986; Petranka 1998).

Overall, stream salamanders in this study seemed to be either disturbance avoiders (*Desmognathus*) or disturbance tolerant (*Eurycea*). Disturbance avoiders are generally long-lived, large-bodied salamanders that are dependent on forests and are sensitive to riparian disturbances. Disturbance tolerant species often can be characterized as short-lived, small bodied, microhabitat generalists that can withstand riparian land uses (Surasinghe and Baldwin 2015). Species that comprise these groups may change in different ecoregions. For example Surasinghe and Baldwin (2015) found that *D. quadramaculatus* and *G. porphyriticus* to be disturbance avoiders in the Blue Ridge ecoregion, however, in the Piedmont ecoregion these species were considered to be disturbance tolerant. Based on results from my study, *Desmognathus* spp. seem to be disturbance avoiders whereas *Eurycea* spp. were disturbance tolerant. Research throughout Appalachia proper has found that in undisturbed areas *Desmognathus* spp. is the dominant stream salamander group whereas in disturbed areas *Eurycea* spp. is the dominant stream salamander (Hamilton, 2002; Hyde and Simons, 2001; Resetarits, 1997). In central Appalachia, Ward et al. (2008) found that abundances of disturbance tolerant salamanders (*E. b. cirrigera*) were higher at roadside sites as compared to forested control sites whereas *Desmognathus* spp. had the inverse response. These riparian disturbances that have a short-term acute effects and then longer, subtle but continuing effects, such as roads, may cause stream salamander communities to shift to disturbance tolerant species without changes in overall abundance (Ward et al. 2008). This indicates the need for research to separate stream salamanders to species instead of looking at total salamander abundance. It is important to note however that even disturbance tolerant species are not totally disturbance-adapted, and these species could still be vulnerable to localized extirpations above some land use threshold.

Predation and competition may also play a role in differences in results between *Eurycea* and *Desmognathus*. Predation may have both direct mortality and indirect behavioral effects on stream salamanders, particularly on *Eurycea* which is the smallest stream salamander with the most predatory salamander species in this study. For example, Barr and Babbitt (2002) found that densities of *E. b. bislineata* were dramatically lower (9.4/m^2 as compared to 54/m^2) when the predatory *Salvelinus fontinalis* (brook trout) were present. Stomach content analysis of predatory salamander species, *G. porphyriticus*, showed that up to 46% of content was other stream salamanders (Bruce 1972). *Eurycea* may be more susceptible to predation from other stream salamanders more so than from fish predation. In a laboratory study, Resetarits (1997) found survival rates of *Eurycea b. wilderae* were highest when *Salvelinus fontinalis* was present and lowest when *G. porphyriticus* was present. Predation from other stream salamanders may not only reduce survival rates, but may also influence microhabitat selection and behavior. Without predators *E. b. wilderae* were evenly distributed in pools and riffles, however, when *D. quadramaculatus*, a large predator species that prefers riffles, were present *E. b. wilderae* were more abundant in pools than in riffles (Beachy 1993). Keen (1982) found when *D. monticola*, a predator of *D. fuscus*, was present activity levels of *D. fuscus* dropped significantly. Others have also hypothesized that decreases in some salamander species abundances when predatory fish and/or salamander species are present may not come from predation but rather from competition (Barr and Babbitt 2002; Bruce 2011; Davic 1983; Hairston 1980). However, much of the research that has examined the influence of competition have been field-based without the ability to control for natural variability. *Eurycea* are opportunistic generalists with diets largely consists of pollutant tolerant benthic macroinvertebrates such as *Chironomids* (Barrett et al. 2012; Burton 1976; Muenz et al. 2008; Petranka 1984). I posit that *Eurycea* perhaps is better able to tolerate poor water quality and riparian habitat conditions as even in degraded conditions prey items are often readily available.

Overall my research found equivocal effects of coal mining on stream salamanders; however this is not to imply that mining does not affect stream salamanders. None of the species in my abundance analysis had strong empirical support for any of the direct mining-related parameters such as the percent active or past mining in the subwatershed, area of valley fill, conductivity, and the number of settling ponds. This was shown not only in the model ranking, but also in the model average results. Research has also shown negative effects of surface mining and valley fills on stream salamander abundance or relative species richness (Hamilton 2002; Muncy et al. 2014; Wood and Williams 2013a; Wood and Williams 2013b). However, these studies all examined the effects of mining versus reference conditions, and the resulting decreases in salamander assemblages may be a reflection of general riparian disturbance and not factors particularly unique to coal mining. My research helped differentiate effects of coal mining and general riparian disturbance on stream salamanders by using a gradient of land uses and disturbances. This gradient of sites showed poor riparian quality lead to declines in *Desmognathus* abundance regardless of the type of large-scale land use associated with that riparian disturbance. Nonetheless, results from my research confirms that riparian habitat quality and erosion prevention and control are important for stream salamanders.

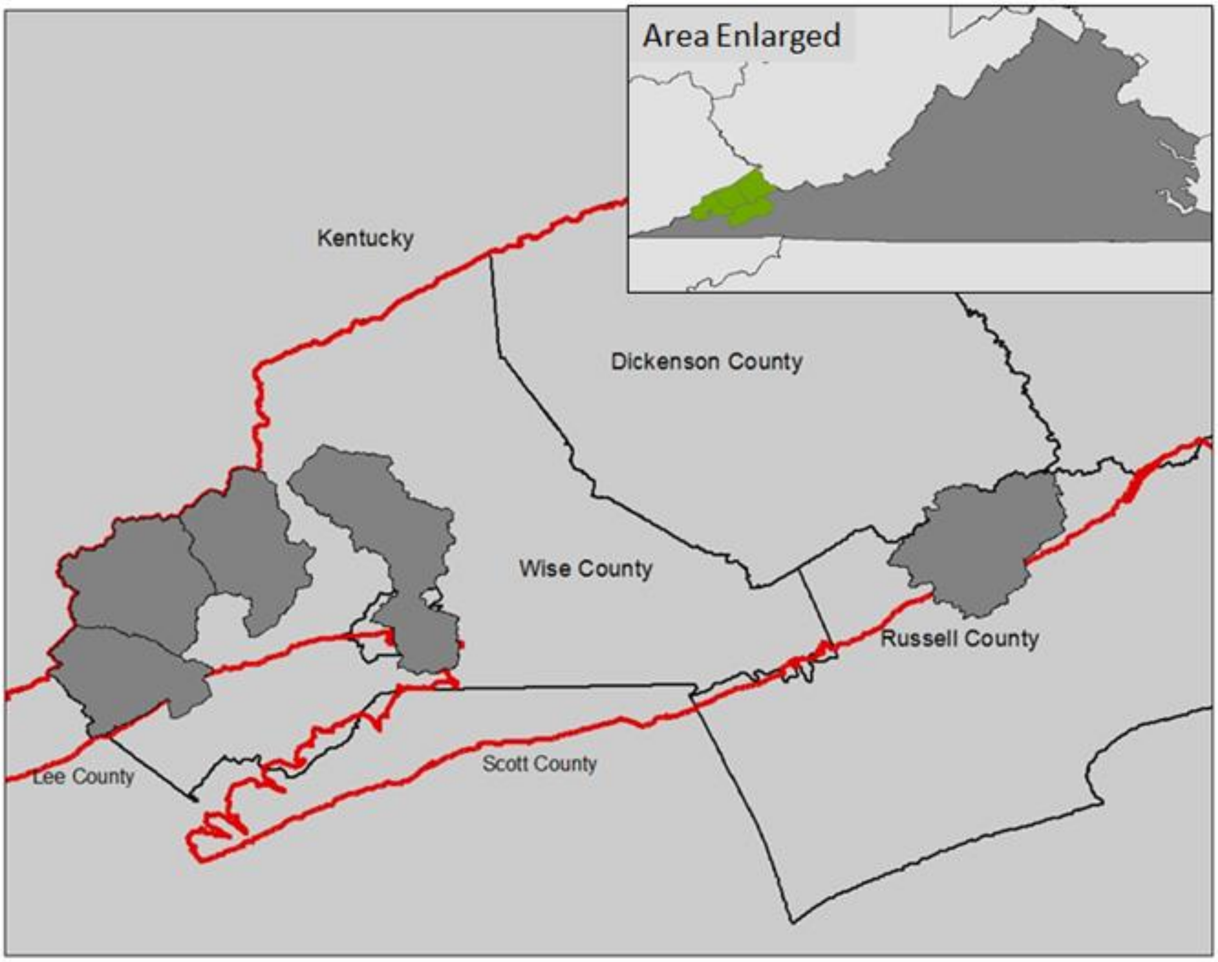

Figure 3.1 Map of the five study watersheds in southwest Virginia (2013), and the watersheds relative location within the state. The red line shows the southwest coalfield boundary. With the exception of a few square kilometers, all the study watersheds are inside the coalfield boundary of the Cumberland Plateau and Mountains ecoregion.

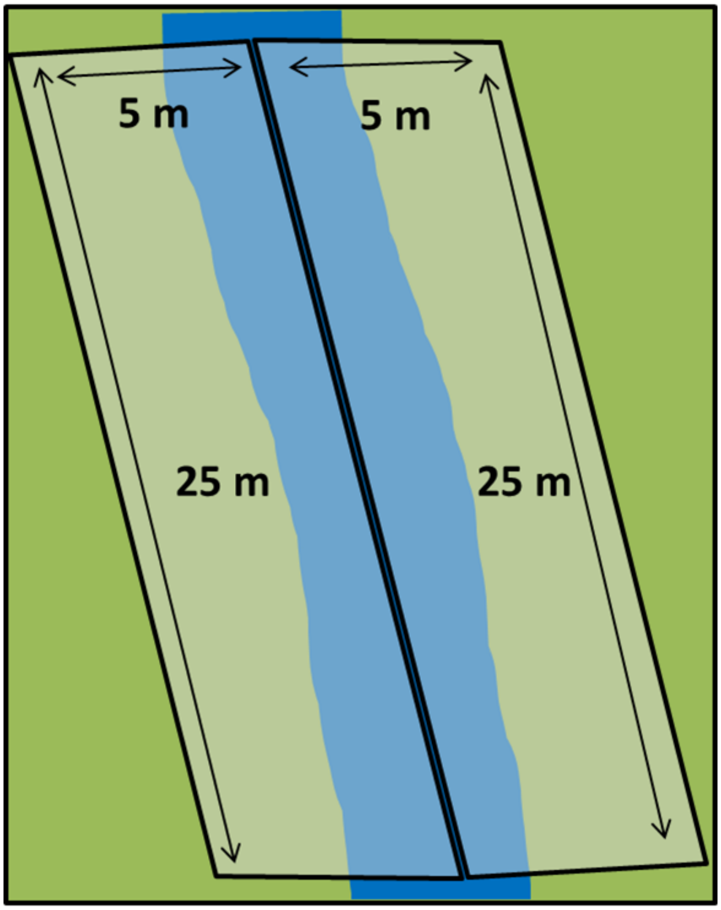

Figure 3.2 Salamander and riparian habitat quadrat placement used in southwest Virginia, 2013. Salamander sampling was done on either the left or right quadrat (determined by a coin flip at each visit). Habitat measurements were taken from the full 25 x 10 m area.

Table 3.1 Information for the five HUC-12 study watersheds used in 2013 including watershed name, HUC-12 identification number, Virginia County(s) where the watershed is located and the area of the watershed.

Watershed Name	HUC-12	County	Watershed area (sq. km)
Callahan Creek	60102060103	Wise	54.7
Dumps Creek	60102050402	Dickenson; Russell	82.3
Pigeon Creek	60102060104	Wise	58.9
Roaring Fork	60102060101	Wise	66
Rocky Fork	60102050501	Wise	91

Table 3.2 Common and scientific names for the stream salamander species within geographic ranges of the study watersheds (southwest Virginia, 2013). Terrestrial and uncommon salamander species were not included.

Scientific Name	Common Name
Notophthalmus viridescens viridescens	Red-spotted Newt
Desmognathus fuscus fuscus	Northern Dusky Salamander
Desmognathus monticola	Seal Salamander
Desmognathus ochrophaeus	Allegheny Mountain Dusky Salamander
Desmognathus welteri	Black Mountain Dusky Salamander
Eurycea bislineata cirrigera	Southern Two-lined Salamander
Eurycea longicauda longicauda	Long-tailed Salamander
Gyrinophilus porphyriticus porphyriticus	Northern Spring Salamander
Pseudotriton montanus diasticus	Midland Mud Salamander
Pseudotriton ruber ruber	Northern Red Salamander

Table 3.3 Parameters measured in the field during summer 2013, southwest Virginia.

Field Parameters	Units	Sampling Events	Additional Description
1. Stream Flow	Categorical	3	Flow Above; Base Flow; Flow Below
2. Soil Temperature	ºC	3	
3. Air Temperature	ºC	3	
4. Water Temperature[1]	ºC	3	
5. Current Weather	Categorical	3	Clear; Cloudy; Rain
6. Past 24-Hour Weather	Categorical	3	Clear; Cloudy; Rain
7. Fish Presence[1]	Categorical	3	1 = Observed; 0 = Not Observed
8. Conductivity[1]	μs/cm	3	Hach water meter (Hach Company, Loveland, CO)
9. Turbidity[1]	cm	3	Depth of secchi tube visibility
10. Stream Width[1]	m	1	Mean of 3 measurements
11. Maximum Stream Depth[1]	cm	1	At base flow
12. Aspect[1]	Degree	1	Compass reading
13. Gradient[1]	Percent	1	Clinometer reading
14. Canopy Cover[1]	Percent	1	Mean of 3 measurements (Noble et al. 2010)
15. Detritus Cover[1]	Percent	1	Mean of 6 measurements (Noble et al. 2010)
16. Large Woody Debris (LWD)[1]	Count	1	(Noble et al. 2010)
17. Stream Bank Erosion[1]	Percent	1	Mean of left and right banks (Noble et al. 2010)
18. Pool Composition[1]	Percent	1	Estimation
19. Substrate Size[1]	cm	1	Mean of 30 measurements (Noble et al. 2010)
20. Stream Embeddedness[1]	Score 1-5	1	Mean of 30 measurements (Noble et al. 2010)
21. Tree Species Richness[1]	Score 1-5	1	(Noble et al. 2010)
22. Tree Diameter (DBH)[1]	cm	1	Mean DBH of trees (Noble et al. 2010)
23. Herbaceous Cover[1]	Percent	1	Mean of 6 measurements (Noble et al. 2010)
24. Sapling/Shrub[1]	Count	1	# Stems with DBH < 10 cm (Noble et al. 2010)
25. Tree Snags[1]	Count	1	# Down, dead trees (Noble et al. 2010)
26. Cobble/Boulder[1]	Percent	1	Estimate of stream substrate cobble and boulder size

[1] Parameters used in microhabitat occupancy and abundance models

Table 3.4 Parameters derived from GIS analysis, southwest Virginia, summer 2013.

Landscape Covariate	Unit	GIS Data Used
1. Forested[1]	Percent	Maxwell et al. 2014
2. Past mining[1]	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
3. Recent mining[1]	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
4. Structures[1]	Count	NAIP 2011
5. Roads[1]	Kilometers	NAIP 2011
6. Valley fill[1]	Hectares	Maxwell et al. 2014; DMME 2013
7. Ponds in segment[1]	Count	NAIP 2011
8. Ponds downstream	Count	NAIP 2011
9. HUC12 forested	Percent	Maxwell et al. 2014
10. HUC12 past mining	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
11. HUC12 recent mining	Percent	Maxwell et al. 2014; NAIP 2011; DMME 2013; NLCD 2011
12. Subwatershed area	Hectares	NED
13. Stream order		NHD
14. Receiving stream order		NHD
15. Elevation[2]	Meters	NED

[1] Calculated within subwatershed from the salamander sampling location upstream

[2] Elevation was used for microhabitat models. All other covariates in this table were used in the landscape-level analyses

Table 3.5 Detection covariates and data type of covariate used in abundance analysis of stream salamanders in southwest Virginia, 2013.

Detection Covariate	Covariate Name	Covariate Type	Additional Description
Stream Flow Above Base	Flow Above	Binomial	1 = Above; 0 = Not Above
Stream Flow Below Base	FlowBelow	Binomial	1 = Below; 0 = Not Below
Soil Temperature	SoilT	Continuous	ºC
Air Temperature	AirT	Continuous	ºC
Water Temperature	WaterT	Continuous	ºC
Current Weather Rainy	Rain	Binomial	1 = Rainy; 0 = Not Rainy
Rain in Past 24 Hours	Rain24	Binomial	1= Rainy; 0 = Not Rainy

Table 3.6 Final *a priori* abundance models for microhabitat covariates used to examine the influence of fine-spatial scale conditions on stream salamanders in southwest Virginia, 2013.

	Predictive Model	**Microhabitat Environmental Parameters**
1.	Null	Intercept
2.	Microhabitat Global	All microhabitat parameters
3.	Size	Stream width; Maximum stream depth
4.	Stream Location	Stream aspect; Gradient; Elevation
5.	Macrohabitat	Aspect; Gradient; Elevation; Canopy cover
6.	Microhabitat	Detritus cover; LWD Count; Stream bank erosion
7.	Stream Comp.	Percent pool
8.	Predation	Fish presence
9.	Chemistry	Conductivity; Water temperature
10.	Sediment	Turbidity; Stream embeddedness; Stream bank erosion
11.	Substrate	Average substrate size; % Cobble and boulder; Stream embeddedness
12.	Canopy	Canopy cover
13.	Trees	Tree species richness; DBH; Canopy cover
14.	Herbaceous	Herbaceous cover; Saplings/shrubs
15.	Shading	Canopy cover; Herbaceous cover; Sapling/shrubs; DBH
16.	Detritus	Detritus cover
17.	Future Detritus	Tree snag count; Canopy cover
18.	Cover	LWD count; Detritus cover; % Cobble and boulder; Stream bank erosion

Table 3.7 *A priori* abundance models with large-scale land use covariates used to examine the influence of landscape-level conditions on stream salamanders in southwest Virginia, 2013.

	Predictive Models	**Large-scale Environmental Parameters**
1.	Null	Intercept
2.	Global	All landscape-level parameters
3.	Forested	Subwatershed forested
4.	Residential	Structures
5.	Non-Mining	Structures; Roads
6.	Past Mining	Subwatershed past mining
7.	Recent Mining	Subwatershed recent mining
8.	Surface Mining	Subwatershed recent mining; Valley fill; Ponds in segment; Ponds downstream
9.	Valley Fill	Valley fill
10.	Pond	Ponds in segment; Ponds downstream
11.	Mining Streams	Valley fill; Ponds in segment
12.	HUC-12 Forested	HUC-12 watershed forested
13	HUC-12 Past Mining	HUC-12 watershed past mining
14	HUC-12 Recent Mining	HUC-12 watershed recent mining
15	Physical	Subwatershed area; Stream order; Receiving stream order

Table 3.8 Total number of salamanders caught by species along with the number of sampling sites and sampling events the species were found in summer 2013, southwest Virginia.

Scientific Name	**Total Captured**	**# of Sampling Sites (n = 70)**	**# Sampling Events (n = 207)**
Desmognathus fuscus	82	27	45
Desmognathus monticola	291	40	81
Desmognathus ochrophaeus	227	37	75
Desmognathus welteri	41	18	21
Desmognathus Larval	6	6	6
Eurycea b. cirrigera	24	15	19
Eurycea longicauda	15	7	8
Eurycea Larval	442	42	85
Gyrinophilus porphyriticus	11	10	11
Pseudotriton montanus	5	3	3
Pseudotriton ruber	7	6	6

Table 3.9 Total salamanders observed, model-averaged estimations of abundance per site along with standard errors and 95 % confidence intervals of lambda (λ)for the four groups of stream salamanders (*Desmognathus fuscus*, *Desmognathus monticola*, *Desmognathus ochrophaeus*, and *Eurycea* spp.) used for full analysis of abundance at the microhabitat and landscape-level scales, southwest Virginia, 2013.

Species Group	**Total Observed**	**Mean Observed** Per Site, Per Visit	**λ Estimate**	**λ SE**	**95% Confidence Interval**
D. fuscus	82	0.396	1.043	0.193	0.726 – 1.498
D. monticola	291	1.406	3.488	0.357	2.854 – 4.263
D. ochrophaeus	273	1.319	3.018	0.374	2.368 – 3.847
Eurycea spp.	481[1]	1.525	4.596	0.556	3.626 – 5.826

[1] This includes 169 *Eurycea* Larval and 1 *Eurycea* Adult found at the outlier site (ROA1-8) which was removed from the abundance analysis.

Table 3.10 Top models for abundance estimates (λ) including the number of parameters in each model (K), Akaike's Information Criterion (AIC) rankings, Δ AIC, and AIC weight (ω_i) for the four groups of stream salamanders, southwest Virginia, 2013.

Abundance Models	**K**	**AIC**	**Δ AIC**[1]	**ω_i**
Desmognathus fuscus				
λ(FutureDetritus), p(SoilT*FlowAbove)	6	305.40	0.00	0.3177
λ(Shading), p(SoilT*FlowAbove)	8	305.74	0.34	0.2680
λ(Canopy), p(SoilT*FlowAbove)	5	306.06	0.66	0.2284
λ(Trees), p(SoilT*FlowAbove)	7	308.99	3.59	0.0528
λ(Microhabitat), p(SoilT*FlowAbove)	7	309.17	3.77	0.0482
λ(Macrohabitat), p(SoilT*FlowAbove)	8	309.47	4.07	0.0415
Desmognathus monticola				
λ(Shading), p(SoilT*FlowBelow)	8	660.83	0.00	0.8273
λ(Trees), p(SoilT*FlowBelow)	7	663.97	3.14	0.1721
λ(FutureDetritus), p(SoilT*FlowBelow)	6	676.01	15.18	0.0004
Desmognathus ochrophaeus				
λ(Shading), p(AirT*Rain)	8	593.97	0.00	0.9795
λ(Trees), p(AirT*Rain)	7	601.78	7.81	0.0197
Eurycea spp.				
λ(Sediment), p(SoilT*Rain)	7	690.20	0.00	1.0000
λ(Location), p(SoilT*Rain)	7	760.72	70.52	0.0000

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

Table 3.11 Beta estimates and standard errors for each covariate in the top abundance models (Δ AIC < 4) for *Desmognathus fuscus,* southwest Virginia, 2013.

Abundance Models and Covariates	**Beta**	**Standard Error**
λ(FutureDetritus), p(SoilT*FlowAbove)		
λ	-0.5571	0.3638
λ.TreeSnags	-0.2613	0.1936
λ.CanopyCover	1.7612	0.5204
p	-0.6409	0.3333
p.SoilT	-0.3803	0.1827
p.FlowAbove	-1.4219	0.5934
λ(Shading), p(SoilT*FlowAbove)		
λ	-0.4556	0.3285
λ.CanopyCover	1.3709	0.4805
λ.HerbaceousCover	-0.2934	0.1668
λ.Sapling/Shrub	0.3544	0.1754
λ.TreeDiameter	-0.1840	0.1733
p	-0.6291	0.3312
p.SoilT	-0.3719	0.1807
p.FlowAbove	-1.4974	0.5897
λ(Canopy), p(SoilT*FlowAbove)		
λ	-0.5104	0.3460
λ.CanopyCover	1.6003	0.4849
p	-0.6034	0.3242
p.SoilT	-0.3782	0.1836
p.FlowAbove	-1.4241	0.5928
λ(Trees), p(SoilT*FlowAbove)		
λ	-0.4793	0.3463
λ.TreeSppRichness	0.1736	0.1716
λ.TreeDiameter	-0.0539	0.1604
λ.CanopyCover	1.4605	0.5382
p	-0.6246	0.3296
p.SoilT	-0.3808	0.1830
p.FlowAbove	-1.4405	0.5913
λ(Microhabitat), p(SoilT*FlowAbove)		
λ	-0.1158	0.2430
λ.DetritusCover	0.6509	0.1489
λ.LWD	0.1775	0.1278
λ.BankErosion	-0.4546	0.2036
p	-0.6565	0.3293
p.SoilT	-0.3378	0.1809
p.FlowAbove	-1.6656	0.5830

Table 3.12 Beta estimates and standard errors for each covariate in the top abundance models (Δ AIC < 4) for *Desmognathus monticola*, southwest Virginia, 2013.

Abundance Models and Covariates	**Beta**	**Standard Error**
λ(Shading), p(SoilT*FlowBelow)		
λ	0.7809	0.1638
λ.CanopyCover	1.1434	0.2328
λ.HerbaceousCover	-0.4549	0.0958
λ.Sapling/Shrub	0.1412	0.1004
λ.TreeDiameter	-0.1327	0.0950
p	-0.7275	0.1648
p.SoilT	-0.4453	0.1146
p.FlowBelow	1.5379	0.4286
λ(Trees), p(SoilT*FlowBelow)		
λ	0.7634	0.1710
λ.TreeSppRichness	0.4201	0.0972
λ.TreeDiameter	-0.0497	0.0906
λ.CanopyCover	1.0517	0.2589
p	-0.7301	0.1611
p.SoilT	-0.4721	0.1153
p.FlowBelow	1.4802	0.4305

Table 3.13 Beta estimates and standard errors for each covariate in the top abundance models (Δ AIC < 4) for *Desmognathus ochrophaeus*, southwest Virginia, 2013.

Abundance Model and Covariates	Beta	Standard Error
λ(Shading), p(AirT*Rain)		
λ	0.8246	0.2028
λ.CanopyCover	1.1991	0.2325
λ.HerbaceousCover	-0.3843	0.1054
λ.Sapling/Shrub	-0.2916	0.1265
λ.TreeDiameter	-0.1265	0.1080
p	-0.8855	0.2439
p.AirT	-0.3530	0.1252
p.Rain	-1.1445	0.3739

Table 3.14 Beta estimates and standard errors for each covariate in the top abundance models (Δ AIC < 4) for *Eurycea* spp., southwest Virginia, 2013.

Abundance Model and Covariates	**Beta**	**Standard Error**
λ(Sediment), p(WaterT*Rain)		
λ	1.4025	0.1852
λ.Turbidity	-0.8164	0.0818
λ.StreamEmbeddedness	-0.4372	0.0682
λ.StreamBankErosion	-0.3201	0.0856
p	-1.1794	0.2352
p.SoilT	0.3252	0.0745
p.Rain	0.7918	0.2754

Table 3.15 Model averaged values (> 0.1) and direction of relationship for individual covariates for stream salamander species abundance, southwest Virginia, summer 2013. Landscape-level covariates are in italics.

Abundance Covariate	***D. fuscus***	***D. monticola***	***D. ochrophaeus***	***Eurycea* spp.**
Stream Width	-	-	-	-
Stream Depth	-	-	-	-
Aspect	-	-	-	-
Gradient	-	-	-	-
Elevation	-	-	-	-
Canopy Cover	1.327(+)	1.128(+)	1.196(+)	-
Detritus Cover	-	-	-	-
LWD	-	-	-	-
Bank Erosion	-	-	-	0.320(-)
Pool Composition	-	-	-	-
Fish Presence	-	-	-	-
Conductivity	-	-	-	-
Water Temperature	-	-	-	-
Turbidity	-	-	-	0.816(-)
Substrate Size	-	-	-	-
Embeddedness	-	-	-	0.437(-)
Tree Spp. Richness	-	-	-	-
DBH	-	0.118(-)	0.126(-)	-
Herbaceous Cover	-	0.376(-)	0.376(-)	-
Sapling/Shrub	-	0.117(+)	0.286(-)	-
Tree Snags	-	-	-	-
Cobble/Boulder	-	-	-	-
Forested	-	-	-	-
Past Mining	-	-	-	-
Recent Mining	-	-	-	-
Structures	-	-	-	-
Roads	-	-	-	-
Valley Fill	-	-	-	-
PondsIN	-	-	-	-
PondsDS	-	-	-	-
HUC12-Forested	-	-	-	-
HUC12-Past Mining	-	-	-	-
HUC12-Recent Mining	-	-	-	-
Subwatershed Area	-	-	-	-
Stream Order	-	-	-	-
Receiving Stream Order	-	-	-	-

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Appendix B

Table B.1 Habitat variables thought to vary temporally that were measured at each salamander sampling event (n = 3) in southwest Virginia, 2013 along with justification/rationale and supporting literature for each parameter.

Parameter	Justification	Supporting Literature
Current Weather	Covariate thought to influence detection	Bailey et al. 2004; Kroll et al. 2010; Williams 2003
Weather: Past 24 hrs.	Covariate thought to influence detection	Bailey et al. 2004; Kroll et al. 2010; Williams 2003
Fish Presence	Fish presence linked to decreases in salamander abundance	Lowe et al. 2004; Petranka 1998; Sih et al. 1992
Stream Flow Condition	Covariate thought to influence detection	
Turbidity	Most work has looked at the effect of sedimentation on salamanders, but limited data on suspended sediment.	Barbour et al. 1999
Air Temperature	Covariate that may influence detection and occupancy	Kroll et al. 2010; Miller et al. 2007; Pawlik 2008; Whiteleather 2001; Williams 2003
Water Temperature	Covariate that may influence detection and occupancy	Bury and Corn 1989; Kroll et al. 2010; Miller et al. 2007; Pawlik 2008; Whiteleather 2001
Soil Temperature	Covariate that may influence detection and occupancy	Crawford 2007; Pawlik 2008; Williams 2003
Conductivity	Conductivity affects are largely unknown for salamanders	Gore 1983; Miller et al. 2007; Whiteleather 2001

Table B.2 Habitat parameters thought to be relatively stabled and sampled once during the 2013 field season (southwest Virginia) along with justification/rationale and supporting literature for each parameter.

Parameter	Justification	Supporting Literature
Wetted Stream Width	Stream size is thought to affect salamander occupancy	Bury and Corn 1989; Miller et al. 2007; Moseley et al. 2008; Stoddard 2001; Williams 2003
Maximum Stream Depth	Stream size and pools are thought to affect salamander occupancy	Bury and Corn1989: Miller et al. 2007
Riffle/Run/Pool Composition	Positive correlation between salamanders and pool habitat	Moseley et al. 2008; Murphy and Hall 1981; Sih et al. 1992; Stoddard 2001; Welsh and Ollivier 1998
Stream Aspect	Influences heat load/UV levels.	Hairston 1949; Stoddard 2001; Whiteleather 2001
Stream Gradient	Research has shown gradient influences sedimentation and salamander abundance	Bury and Corn 1989; Crawford 2007; Hall et al. 1978; Moseley et al. 2008; Stoddard 2001; Whiteleather 2001
Stream Substrate Size	Part of USACE protocol.	Noble et al. 2010
Stream Substrate Composition	Salamander abundances have been correlated to rock/cobble substrate	Bury and Corn 1989; Davic and Orr 1987; Miller et al. 2007; Mitchell 1999; Moseley et al. 2008; Pawlik 2008; Petranka 1998; Redmond 1980; Welsh and Ollivier 1998
Stream Bank Erosion	Indication of unstable stream banks and sedimentation	Miller et al. 2007; Orser and Shure 1972; Noble et al. 2010
Large Woody Debris	Important cover object for salamanders.	Miller et al. 2007; Moseley et al. 2008; Murphy and Hall 1981; Stoddard 2001; Noble et al. 2010
Tree Snags	Part of USACE protocol. Successional stage	deMaynadier and Hunter 1998; Noble et al. 2010
DBH of Riparian Trees	Part of USACE protocol. Successional stage	Moseley et al. 2008; Noble et al. 2010
Riparian Tree Species Richness	Part of USACE protocol. Help to determine riparian vegetation quality	Noble et al. 2010

Table B.2 (Continued)

Parameter	Justification	Supporting Literature
Saplings, Shrubs	Part of USACE protocol.	Stoddard 2001; Noble et al. 2010
Riparian Herbaceous Cover	Part of USACE protocol.	Stoddard 2001; Noble et al. 2010
Detritus Cover	Part of USACE protocol. Important for salamander microhabitat	Miller et al 2007; Noble et al. 2010
Substrate Embeddedness	Research has shown sedimentation to cause declines in salamander abundance.	Bury and Corn1989; Hawkins et al. 1983; Lowe and Bolger2002; Lowe et al. 2004; Moseley et al 2008;Noble et al. 2010;Welsh and Ollivier 1998
Canopy Cover	Research has shown relationship between salamander abundance and canopy cover	Bury and Corn 1989; Crawford 2007; Miller et al. 2007; Stoddard 2001; Noble et al. 2010; Whiteleather 2001

Table B.3 AIC comparison of all landscape-level and microhabitat abundance models for *Desmognathus fuscus*, southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Abundance Models _Desmognathus fuscus_	**K**	**AIC**	**Δ AIC**	**ω_i**
λ(FutureDetritus), p(SoilT*FlowAbove)	6	305.40	0.00	0.2940
λ(Shading), p(SoilT*FlowAbove)	8	305.74	0.34	0.2481
λ(Canopy), p(SoilT*FlowAbove)	5	306.06	0.66	0.2114
λ(Trees), p(SoilT*FlowAbove)	7	308.99	3.59	0.0488
λ(Microhabitat), p(SoilT*FlowAbove)	7	309.17	3.77	0.0446
λ(Macrohabitat), p(SoilT*FlowAbove)	8	309.47	4.07	0.0384
λ(SurfaceMining), p(SoilT*FlowAbove)	_8_	_309.61_	_4.21_	_0.0358_
λ(Cover), p(SoilT*FlowAbove)	8	309.86	4.46	0.0316
λ(NonMining), p(SoilT*FlowAbove)	_6_	_310.81_	_5.41_	_0.0197_
λ(Ponds), p(SoilT*FlowAbove)	_6_	_312.51_	_7.11_	_0.0084_
λ(Detritus), p(SoilT*FlowAbove)	5	312.64	7.24	0.0079
λ(Forested), p(SoilT*FlowAbove)	_5_	_313.55_	_8.15_	_0.0050_
λ(MiningStreams), p(SoilT*FlowAbove)	_6_	_313.85_	_8.45_	_0.0043_
λ(Herbaceous), p(SoilT*FlowAbove)	6	318.54	13.14	0.0004
λ(RecentMining), p(SoilT*FlowAbove)	_5_	_318.83_	_13.43_	_0.0004_
λ(Physical), p(SoilT*FlowAbove)	_7_	_318.90_	_13.50_	_0.0003_
λ(Chemistry), p(SoilT*FlowAbove)	6	319.47	14.07	0.0003
λ(HUC12Forested), p(SoilT*FlowAbove)	_5_	_319.61_	_14.21_	_0.0002_
λ(ValleyFill), p(SoilT*FlowAbove)	_5_	_319.89_	_14.49_	_0.0002_
λ(HUC12RecentMining), p(SoilT*FlowAbove)	_5_	_321.58_	_16.18_	_0.0001_
λ(HUC12PastMining), p(SoilT*FlowAbove)	_5_	_323.22_	_17.82_	_0.0000_
λ(Predation), p(SoilT*FlowAbove)	5	328.22	22.82	0.0000
λ(Size), p(SoilT*FlowAbove)	6	328.77	23.37	0.0000
λ(StreamComp), p(SoilT*FlowAbove)	5	328.87	23.47	0.0000
λ(Residential), p(SoilT*FlowAbove)	_5_	_329.60_	_24.20_	_0.0000_
λ(PastMining), p(SoilT*FlowAbove)	_5_	_331.18_	_25.78_	_0.0000_
λ(MicrohabitatNull), p(SoilT*FlowAbove)	4	332.53	27.13	0.0000
λ(LandscapeNull), p(SoilT*FlowAbove)	_4_	_332.53_	_27.13_	_0.0000_
λ(Sediment), p(SoilT*FlowAbove)	7	333.92	28.52	0.0000
λ(Substrate), p(SoilT*FlowAbove)	7	336.28	30.88	0.0000
λ(Location), p(SoilT*FlowAbove)	7	336.29	30.89	0.0000

Table B.4 AIC comparison of all landscape-level and microhabitat abundance models for *Desmognathus monticola*, southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Abundance Models _Desmognathus monticola_	**K**	**AIC**	**Δ AIC**	**ω_i**
λ(Shading), p(SoilT*FlowBelow)	8	660.83	0.00	0.8273
λ(Trees), p(SoilT*FlowBelow)	7	663.97	3.14	0.1721
λ(FutureDetritus), p(SoilT*FlowBelow)	6	676.01	15.18	0.0004
λ(Canopy), p(SoilT*FlowBelow)	5	679.67	18.84	0.0001
λ(Macrohabitat), p(SoilT*FlowBelow)	8	680.06	19.23	0.0001
λ(Physical), p(SoilT*FlowBelow)	_7_	_688.09_	_27.26_	_0.0000_
λ(NonMining), p(SoilT*FlowBelow)	_6_	_688.55_	_27.72_	_0.0000_
λ(Microhabitat), p(SoilT*FlowBelow)	7	690.50	29.67	0.0000
λ(Cover), p(SoilT*FlowBelow)	8	691.29	30.46	0.0000
λ(Detritus), p(SoilT*FlowBelow)	5	696.12	35.29	0.0000
λ(Herbaceous), p(SoilT*FlowBelow)	6	705.21	44.38	0.0000
λ(SurfaceMining), p(SoilT*FlowBelow)	_8_	_717.52_	_56.69_	_0.0000_
λ(Chemistry), p(SoilT*FlowBelow)	6	717.64	56.81	0.0000
λ(ValleyFill), p(SoilT*FlowBelow)	_5_	_727.97_	_67.14_	_0.0000_
λ(MiningStreams), p(SoilT*FlowBelow)	_6_	_729.35_	_68.52_	_0.0000_
λ(Ponds), p(SoilT*FlowBelow)	_6_	_736.69_	_75.86_	_0.0000_
λ(Size), p(SoilT*FlowBelow)	6	741.57	80.74	0.0000
λ(PastMining), p(SoilT*FlowBelow)	_5_	_749.49_	_88.66_	_0.0000_
λ(Predation), p(SoilT*FlowBelow)	5	752.68	91.85	0.0000
λ(Residential), p(SoilT*FlowBelow)	_5_	_753.24_	_92.41_	_0.0000_
λ(Forested), p(SoilT*FlowBelow)	_5_	_758.66_	_97.83_	_0.0000_
λ(Location), p(SoilT*FlowBelow)	7	760.89	100.06	0.0000
λ(Sediment), p(SoilT*FlowBelow)	7	763.65	102.82	0.0000
λ(StreamComp), p(SoilT*FlowBelow)	5	765.06	104.23	0.0000
λ(RecentMining), p(SoilT*FlowBelow)	_5_	_765.12_	_104.29_	_0.0000_
λ(MicrohabitatNull), p(SoilT*FlowBelow)	4	767.56	106.73	0.0000
λ(Substrate), p(SoilT*FlowBelow)	7	767.56	106.73	0.0000
λ(LandscapeNull), p(SoilT*FlowBelow)	_4_	_767.56_	_106.73_	_0.0000_
λ(HUC12Forested), p(SoilT*FlowBelow)	_5_	_769.39_	_108.56_	_0.0000_
λ(HUC12PastMining), p(SoilT*FlowBelow)	_5_	_769.42_	_108.59_	_0.0000_
λ(HUC12RecentMining), p(SoilT*FlowBelow)	_5_	_769.45_	_108.62_	_0.0000_

Table B.5 AIC comparison of all landscape-level and microhabitat abundance models for *Desmognathus ochrophaeus*, southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Abundance Models *Desmognathus ochrophaeus*	**K**	**AIC**	**Δ AIC**	**ω_i**
λ(Shading), p(AirT*Rain)	8	593.97	0.00	0.9795
λ(Trees), p(AirT*Rain)	7	601.78	7.81	0.0197
λ(Canopy), p(AirT*Rain)	5	609.09	15.12	0.0005
λ(FutureDetritus), p(AirT*Rain)	6	610.76	16.79	0.0002
λ(Herbaceous), p(AirT*Rain)	6	642.26	48.29	0.0000
λ(Microhabitat), p(AirT*Rain)	7	644.82	50.85	0.0000
λ(Cover), p(AirT*Rain)	8	646.43	52.46	0.0000
*λ(NonMining), p(AirT*Rain)*	*6*	*648.47*	*54.50*	*0.0000*
λ(Detritus), p(AirT*Rain)	5	650.48	56.51	0.0000
*λ(Ponds), p(AirT*Rain)*	*6*	*652.87*	*58.90*	*0.0000*
*λ(MiningStreams), p(AirT*Rain)*	*6*	*653.71*	*59.74*	*0.0000*
*λ(SurfaceMining), p(AirT*Rain)*	*8*	*654.52*	*60.55*	*0.0000*
*λ(Residential), p(AirT*Rain)*	*5*	*660.36*	*66.39*	*0.0000*
*λ(ValleyFill), p(AirT*Rain)*	*5*	*661.46*	*67.49*	*0.0000*
λ(Sediment), p(AirT*Rain)	7	663.98	70.01	0.0000
*λ(Physical), p(AirT*Rain)*	*7*	*664.09*	*70.12*	*0.0000*
λ(Chemistry), p(AirT*Rain)	6	667.05	73.08	0.0000
*λ(HUC12Forested), p(AirT*Rain)*	*5*	*667.75*	*73.78*	*0.0000*
λ(Substrate), p(AirT*Rain)	7	668.22	74.25	0.0000
*λ(HUC12RecentMining), p(AirT*Rain)*	*5*	*669.05*	*75.08*	*0.0000*
*λ(HUC12PastMining), p(AirT*Rain)*	*5*	*671.15*	*77.18*	*0.0000*
λ(Size), p(AirT*Rain)	6	672.98	79.01	0.0000
λ(Predation), p(AirT*Rain)	5	673.23	79.26	0.0000
*λ(RecentMining), p(AirT*Rain)*	*5*	*674.29*	*80.32*	*0.0000*
λ(Location), p(AirT*Rain)	7	674.47	80.50	0.0000
λ(MicrohabitatNull) P(AirT*Rain)	4	675.88	81.91	0.0000
*λ(LandscapeNull), p(AirT*Rain)*	*4*	*675.88*	*81.91*	*0.0000*
λ(Macrohabitat), p(AirT*Rain)	8	676.47	82.50	0.0000
λ(StreamComp), p(AirT*Rain)	5	676.96	82.99	0.0000
*λ(PastMining), p(AirT*Rain)*	*5*	*677.20*	*83.23*	*0.0000*
*λ(Forested), p(AirT*Rain)*	*5*	*677.21*	*83.24*	*0.0000*

Table B.6 AIC comparison of all landscape-level and microhabitat abundance models for *Eurycea* spp., southwest Virginia, 2013. Also includes the number of parameters (K), Δ AIC, and AIC weight (ω_i) for each model. Landscape-level models are in italics.

Abundance Models *Eurycea* spp.	**K**	**AIC**	**Δ AIC**	**ω_i**
λ(Sediment), p(SoilT*Rain)	7	690.20	0.00	1.0000
λ(Location), p(SoilT*Rain)	7	760.72	70.52	0.0000
λ(Macrohabitat), p(SoilT*Rain)	8	762.66	72.46	0.0000
λ(HUC12RecentMining), p(SoilT*Rain)	_5_	_770.27_	_80.07_	_0.0000_
λ(Cover), p(SoilT*Rain)	8	773.05	82.85	0.0000
λ(HUC12Forested), p(SoilT*Rain)	_5_	_785.31_	_95.11_	_0.0000_
λ(Physical), p(SoilT*Rain)	_7_	_802.34_	_112.14_	_0.0000_
λ(SurfaceMining), p(SoilT*Rain)	_8_	_804.18_	_113.98_	_0.0000_
λ(HUC12PastMining), p(SoilT*Rain)	_5_	_805.12_	_114.92_	_0.0000_
λ(RecentMining), p(SoilT*Rain)	_5_	_807.06_	_116.86_	_0.0000_
λ(Substrate), p(SoilT*Rain)	7	807.60	117.40	0.0000
λ(Microhabitat), p(SoilT*Rain)	7	808.07	117.87	0.0000
λ(Forested), p(SoilT*Rain)	_5_	_814.16_	_123.96_	_0.0000_
λ(Residential), p(SoilT*Rain)	_5_	_817.50_	_127.30_	_0.0000_
λ(NonMining), p(SoilT*Rain)	_6_	_819.50_	_129.30_	_0.0000_
λ(Herbaceous), p(SoilT*Rain)	6	821.94	131.74	0.0000
λ(Size), p(SoilT*Rain)	6	823.21	133.01	0.0000
λ(Shading), p(SoilT*Rain)	8	823.99	133.79	0.0000
λ(Predation), p(SoilT*Rain)	5	825.27	135.07	0.0000
λ(PastMining), p(SoilT*Rain)	_5_	_825.52_	_135.32_	_0.0000_
λ(Detritus), p(SoilT*Rain)	5	825.63	135.43	0.0000
λ(MiningStreams), p(SoilT*Rain)	_6_	_827.46_	_137.26_	_0.0000_
λ(Ponds), p(SoilT*Rain)	_6_	_827.49_	_137.29_	_0.0000_
λ(StreamComp), p(SoilT*Rain)	5	828.26	138.06	0.0000
λ(MicrohabitatNull), p(SoilT*Rain)	4	829.60	139.40	0.0000
λ(LandscapeNull), p(SoilT*Rain)	_4_	_829.60_	_139.40_	_0.0000_
λ(Canopy), p(SoilT*Rain)	5	831.21	141.01	0.0000
λ(ValleyFill), p(SoilT*Rain)	_5_	_831.37_	_141.17_	_0.0000_
λ(FutureDetritus), p(SoilT*Rain)	6	831.98	141.78	0.0000
λ(Chemistry), p(SoilT*Rain)	6	832.41	142.21	0.0000
λ(Trees), p(SoilT*Rain)	7	833.41	143.21	0.0000

Chapter 4

Validation of a Stream and Riparian Habitat Assessment Protocol using Stream Salamanders in the Southwest Virginia Coalfields

Abstract

Within the central Appalachia coalfields, the aquatic effects of large-scale land use changes are of particular ecological concern. Coal mining practices, particularly surface coal extraction and associated valley fills, both directly and indirectly affect physical stream habitat and water quality. Additionally, many residential areas in this region are concentrated in flood plains and riparian areas whereby they alter stream and riparian habitat. For example, runoff and untreated sewage into area streams continue to degrade water quality. Identification and quantification of land use alterations to ecosystems are a necessary first-step to aid in mitigation of negative consequences to biota. However, oftentimes quantifying physical environmental quality such as stream and riparian habitat can be quite difficult, particularly when there are time or fiscal limitations. Standard protocols such as the U.S. Environmental Protection Agency (EPA) Stream Habitat Rapid Bioassessment Protocol have been established to be cost- and time-effective (Barbour et al. 1999). This protocol estimates ten different stream and riparian conditions on a scale of 0 to 20. However, using estimations can be problematic due to large potential variation in the scoring, depending on differences in training, experience, and opinion of the personnel doing the estimations. In order to help negate this and provide a simplified process, the U.S. Army Corps of Engineers (USACE) developed a functional assessment for streams that measures 11 stream and riparian parameters along with watershed land use to calculate three different scores: a hydrology score, biogeochemical score, and habitat score

(Noble et al. 2010). My research examined the correlation of stream salamander occupancy and abundance to the three USACE scores. In the summer of 2013, I visited 70 sites (sampled three times each) in the southwest Virginia coalfields to collect salamanders and quantify stream and riparian microhabitat parameters. Using an *information-theoretic* framework for analysis I found strong relationships for both occupancy and abundance of three *Desmognathus spp.* and the USACE Habitat FCI score. My findings support use of the USACE protocol for stream and riparian habitat assessment.

Introduction

Central Appalachia has high levels of terrestrial and aquatic biodiversity and species richness, with many endemic species (Morse et al. 1993). The New River catchment in Virginia and West Virginia alone is estimated to have 23 endemic plant and animal species (Hocutt et al. 1978). The aquatic systems in this region contain some of the highest levels of biodiversity in North America, including ten percent of global salamander and freshwater mussel diversity (Bernhardt and Palmer 2011; Stein et al. 2000). Unfortunately, many of these species are listed as threatened or endangered (Bernhardt and Palmer 2011). For example, 46 species of freshwater mussels once inhabited the Powell River system in southwest Virginia and northeast Tennessee yet only 36 species, seven of which are listed as federally endangered, remain (Ahlstedt et al. 2005; Ortmann 1918). In the United States, the biggest threat to biodiversity is declining habitat conditions and habitat loss (Wilcove et al. 1998).

Many stream systems and their associated habitats in the eastern United States, including those in central Appalachia, have been significantly altered by past or current anthropogenic disturbances (Karr 1991). From the time of European settlement of central Appalachia in the 1700s through the mid-1900s, central Appalachian streams have been used as unchecked drainages for residential wastewater, mining operation byproducts, and industrial discharges. In particular, these habitats have been altered by factors associated with coal mining, and over the past three decades, these stressors may have increased exponentially with the advent of large-scale mountaintop removal mining. Although mountaintop mining accounts for 37% of coal production in Appalachia (National Mining Association 2012), it is the most controversial because of the associated disproportionately large environmental footprint. The United States Environmental Protection Agency (EPA) estimates that in central Appalachia, ten percent of the region’s land area (500,000 ha) have been affected by mountaintop mining, and over 2,000 km of headwater stream habitat buried by valley fills (US EPA 2011). However, mountaintop mining extraction methods are common in central Appalachia as it is cost-effective and allows for mining of shallow coal seams that cannot safely be mined by underground methods (Craynon et al. 2012).

Along with direct physical stream loss from valley fills, typically the watershed hydrology, stream chemistry, biological communities, and instream and riparian habitats also are altered substantially from pre-mined conditions (Griffith et al. 2012; Pond et al. 2008; Starnes and Gasper 1995). Physical alteration of streams resulting from the mining process such as dredging, burial of headwaters, and dam construction alter stream hydrology in ways that are analogous to vast geologic change. Changes from surface coal mining on upland terrestrial processes such as clearing of vegetation, increased impervious surfaces, and changes in topography, are additional stressors on these aquatic systems that vary from temporary to permanent in duration (Merriam et al. 2011; Simmons et al. 2008). Of course, many of these hydrological changes are not unique to coal mining land use per se, and are often seen in streams with urban development, and to a lesser extent agricultural, and forestry land uses. For example, research in both urban and mining watersheds suggests that as watersheds become increasingly altered, vegetation is replaced by impervious surfaces such as parking lots or coal processing areas. This causes the flow regime (hydrology) of a stream system to respond more quickly to rain events becoming “flashy” resulting in more frequent and intense stream flood events (Price et al. 2006; Starnes and Gasper 1995). These high flow events can lead to channelization of the stream along with eroded, unstable stream banks whereby instream habitat is buried by sediment as terrestrial erosion continues (Booth et al. 2004). If erosion is not controlled, terrestrial processes can be significantly altered by the loss of soil and nutrients (Baker 1985). Additionally, these dramatic terrestrial changes influence water chemistry, both directly and indirectly. All of these factors have had both acute and chronic effects on specific aquatic biota in mining-influenced streams (Northingham et al. 2011; Pond 2012).

Nonetheless, much is still unknown about many of the terrestrial and aquatic ecosystem dynamics and interactions caused by dramatic landscape alterations from coal mining practices (Simmons et al. 2008; Stout and Wallace 2005). Physical stream and riparian habitat quality provides critical areas for aquatic organisms to feed, reproduce, and take refuge from both predators as well as high-flow events (Hynes 1968; Maddock 1999). Without good instream and riparian habitat, mitigation of water chemistry parameters alone will not allow for subsequent aquatic biota recovery. However, accurately assessing and measuring physical stream and riparian habitat can be difficult due to fiscal and personnel constraints. Consequently regulators and mangers are accordingly challenged by inability to understand current conditions, restoration needs and proper management priorities and directions.

In Virginia, the Virginia Department of Environmental Quality (VADEQ) is responsible for ensuring the compliance of mining operations under the Clean Water Act (CWA) and the Surface Mining Control and Reclamation Act (SMCRA). In order to determine stream health, the VADEQ requires coal companies to monitor water chemistry parameters as well as benthic macroinvertebrates using the Virginia Stream Condition Index (VA-SCI), a multi-metric benthic macroinvertebrate assessment protocol (Burton and Gerritsen 2003). Additionally, the EPA Rapid Bioassessment Protocol (RBP) habitat assessment is used to visibly estimate instream habitats as well as some limited riparian habitat parameters such as bank stability and bank vegetation cover (Barbour et al. 1999). Although this habitat assessment method is time- and financially-effective, overall it is a qualitative estimation that can easily be biased due to inexperienced or improperly trained personnel. A more quantitative method that takes into account surrounding riparian quality would provide more accurate, reliable results.

In 2010 the U.S. Army Corps of Engineers (USACE) developed the *Operational Draft Regional Guidebook for the Functional Assessment of High-gradient Ephemeral and Intermittent Headwater Streams in Western West Virginia and Eastern Kentucky* using the Hydrogeomorphic approach (HGM) in order to provide a cost- and time-effective stream and riparian habitat assessment that is quantitative in nature (Noble et al. 2010). The HGM protocol calculates three Functional Capacity Index (FCI) scores: Hydrology, Habitat, and Biogeochemical. Hydrological function is defined as "the ability of the high-gradient headwater stream to dissipate energy associated with flow velocity and transport water downstream" (Noble et al. 2010). The Hydrological FCI incorporates substrate embeddedness, substrate size, large woody debris (LWD), stream bank erosion, and subwatershed land use (Table 4.1). Habitat function is defined as "the capacity of the high-gradient headwater stream ecosystem to provide critical life requisites to selected components of the vertebrate and invertebrate wildlife community" (Noble et al. 2010). The Habitat FCI uses the following parameters: canopy cover, substrate embeddedness, substrate size, LWD, riparian tree diameter, tree snag density, sapling and shrub density, riparian tree species richness, detritus cover, herbaceous cover, and subwatershed land use (Table 4.2). Biogeochemical function is defined as "the ability of high-gradient headwater stream ecosystem to retain and transfer inorganic materials needed for biological processes into organic forms and to oxidize those organic molecules back into elemental forms through respiration and decomposition" (Noble et al. 2010). Substrate embeddedness, LWD, riparian tree diameter, sapling and shrub density, detritus cover, herbaceous cover, and subwatershed land use are the parameters that comprise the Biogeochemical FCI (Table 4.3). Final scores for all three FCI components range from 0 – 1.0 where a score of 1.0 indicates the function to be equal to that of a reference site.

Whereas the FCI habitat assessment approach was designed for a region adjacent to the Virginia coalfields, the protocol does not contain parameters specific to the western West Virginia and eastern Kentucky region that are perhaps more similar than dissimilar to the Virginia coalfields physically and biologically, i.e., eastern Kentucky is in the same ecoregion as the Virginia coalfields (the Cumberland Plateau and Mountains ecoregion). Stream salamanders such as *Desmognathus* spp. are thought to be good indicators of riparian and instream habitat quality (Welsh and Ollivier 1998; Welsh et al. 2005). A small validation study (N = 10) of this approach showed positive correlations between stream salamander abundance and the Habitat FCI score (Noble et al. 2014). Additionally, my occupancy and abundance analyses of stream salamanders had strong relationships to riparian and instream habitat parameters that are covariates that are also used in the FCI scores such as canopy cover, stream substrate embeddedness, and stream bank erosion (Sweeten 2015a, Sweeten 2015b). I therefore decided to do two *post hoc* analyses (one using occupancy, one using abundance) to examine and possibly validate the relationship among the three FCI scores and stream salamanders.

Methods

Regional Description

My study sites are located in the coalfields of southwest Virginia in Wise, Russell, and Dickenson Counties. This area is part of the Cumberland Plateau (Ecoregion 69d), a sub-region of the central Appalachian Mountains (Omernik 1987). This region is also commonly known as the Northern Cumberland Plateaus and Mountains Ecoregion (Anderson et al. 2001; Bailey 1995). Topography is characterized by steep mountains with narrow valleys with a mean peak elevation of 760 m (Woodward and Hoffman 1991). Most soils in this region are Utisols, Alfisols, and Inceptisols (McNab and Avers 1995). Mean annual precipitation is about 1150 mm with a mean temperature of 13°C (McNab and Avers 1995; Woodward and Hoffman 1991). Due to the steep topography, this region tends to have a high density of small/medium-sized streams (McNab and Avers 1995). It is estimated that 93 % of the Cumberland Plateau is forested, whereas 4 % of the region is agricultural/open area (VDGIF 2005). Coal mining (both surface and underground), forestry, agriculture, industry, and residential areas (many with inadequate sewage infrastructure) are the dominant land uses (VDGIF 2005). The forested areas are characterized by a diverse mix of hardwood and conifers (Woodward and Hoffman 1991). Common tree species include red oak (*Quercus rubra*), white oak (*Quercus alba*), pignut hickory (*Carya glabra*), red maple (*Acer rubrum*), yellow poplar (*Liriodendron tulipifera*), American beech (*Fagus grandifolia*), basswood (*Tilia americana*), and white pine (*Pinus strobus*) (Hamilton 2002; McNab and Avers 1995).

Site Selection

For my study area, I selected five 12-digit Hydrologic Unit Codes (HUC-12) watersheds in southwest Virginia (Table 4.4). These five watersheds are similar in area, located within the Cumberland Plateau and Mountains Region, and have active coal mining along with other land uses. I divided streams within these watersheds into segments by their stream order, and gave each stream segment in each watershed a unique identification number. Based on the identification numbers, I randomly selected first- or second-order stream segments for salamander sampling sites; however, in some cases I used best professional judgment to select sites that had landowner access and that I considered safe to sample. Throughout the five HUC-12 watersheds, I sampled a total of 70 sites (N = 70). Due to a lack of accessible sites on Pigeon Creek and Dumps Creek, these watersheds had 10 salamander sampling locations. Roaring Fork, and Rocky Fork had 15 salamander sampling locations, and there were 20 sites on Callahan Creek.

To accurately estimate detection rates, I visited 67 of the 70 three times in 2013 (Bailey et al. 2004; MacKenzie and Royle 2005). Due to access issues, I was only able to sample three of the sites twice in 2013. At each sampling site location a 25 m long by 5 m wide quadrat was placed parallel to the stream with the stream center as the right or left edge of the quadrat (Hairston 1986; Jung et al. 2000; Kucken et al. 1994). Right or left quadrat placement was determined randomly using a coin flip. I hand-captured adult salamanders by overturning all rocks, detritus, and logs within the 25 m x 5 m quadrat at each sampling site. I identified all adult salamanders to species in the field and immediately released to within 2 m of capture location. A D-frame dip net was used to sample in-stream habitat (Davic 1983; Gore 1983). All larval salamanders were removed by hand from the dip net, placed in a bucket of fresh stream water for identification, and then released within 2 m of the capture location. Due to the difficulty of identifying larval salamanders to the species level, I identified larval salamanders to genus.

FCI Assessment Protocol

A full description of the original field protocols for the FCI protocol can be found in Noble et al. (2010). Though largely following the FCI protocol, I made some modifications for my study: first, habitat measurements were taken from a 25 m x 10 m quadrat centered on the salamander quadrat instead of a 30.5 m x 15.2 m quadrat suggested by the FCI protocol. Secondly, I took three canopy cover measurements, six detritus measurements, and six herbaceous cover measurements, not the ten canopy cover, eight detritus, and eight herbaceous measurements recommended by the FCI protocol.

The Functional Capacity Index (FCI) protocol calculates three scores: Hydrology, Habitat, and Biogeochemical. The equation used for calculating the Hydrology FCI is:

$$FCI = \left\{ \frac{V_{WLUSE} + \frac{\left[V_{LWD} + min\left(V_{SUBSTRATE,}V_{EMBED,}V_{BERO}\right)\right]}{2}}{2} \right\}$$

where V_{LWD} = the number of down woody stems per 25 m of stream reach, $V_{SUBSTRATE}$ = the median stream substrate particle size (n = 30), V_{EMBED} = the mean embeddedness of the stream channel (n = 30), V_{BERO} = the total percent of eroded stream channel bank, and V_{WLUSE} was calculated using land cover data from Maxwell et al. (2014), in ArcMap (ESRI, Redlands, California). Details of sampling methods can be found in Table 4.1.

One of two equations is used to calculate the Habitat FCI score. If there is an average channel canopy cover of ≥ 20 percent, then the equation is:

$$FCI = \langle \frac{\left[V_{CCANOPY} + \min\left(V_{SUBSTRATE,}V_{EMBED}\right)\right]}{2} X \left\{ \frac{\frac{(V_{LWD} + V_{DETRITUS})}{2} + \frac{\left[\left(\frac{V_{SNAG} + V_{TDBH} + V_{SRICH}}{3}\right) + V_{WLUSE}\right]}{2}}{2} \right\} \rangle$$

where V_{CANOPY} = mean percent canopy cover (n = 3), $V_{SUBSTRATE}$ = the median stream substrate particle size (n = 30), V_{EMBED} = the mean embeddedness of the stream channel (n = 30), V_{LWD} = the number of down woody stems per 25 m of stream reach, $V_{DETRITUS}$ = the mean percent detritus cover (n = 6), V_{SNAG} = the number of standing, dead snags per 25 m of stream, V_{TDBH} = the mean diameter at breast height (DBH) of trees with DBH ≥ 10 cm, V_{SRICH} = riparian vegetation species richness score per 25 m of stream reach, and V_{WLUSE} was calculated using land cover data from Maxwell et al. (2014), in ArcMap (ESRI, Redlands, California).

If there is ≤ 20 percent canopy cover, then the Habitat FCI is calculated using this equation:

$$FCI = \left\langle \min\left(V_{EMBED,}V_{SUBSTRATE}\right) X \, \frac{\left\{\left(\frac{V_{LWD} + V_{DETRITUS}}{2}\right) + \left[\frac{\left(\frac{V_{SNAG} + V_{SSD} + V_{HERB} + V_{SRICH}}{6}\right) + V_{WLUSE}}{4}\right]\right\}}{2} \right\rangle$$

where V$_{\text{EMBED}}$ = the mean embeddedness of the stream channel (n = 30), V$_{\text{SUBSTRATE}}$ = the median stream substrate particle size (n = 30), V$_{\text{LWD}}$ = the number of down woody stems per 25 m of stream, V$_{\text{DETRITUS}}$ = the mean percent detritus cover (n = 6), V$_{\text{SNAG}}$ = the number of standing, dead snags per 25 m of stream, V$_{\text{SSD}}$ = the number of saplings and shrubs per 25 m of stream reach, V$_{\text{HERB}}$ = the mean percent cover of herbaceous vegetation (n = 6), V$_{\text{SRICH}}$ = riparian vegetation species richness score per 25 m of stream reach, and V$_{\text{WLUSE}}$ was calculated using land cover data from Maxwell et al. (2014), in ArcMap (ESRI, Redlands, California). See Table 4.2 for parameter sampling methods.

One of two equations is used to calculate the Biogeochemical FCI score. If there is a mean channel canopy cover of ≥ 20 percent, then the equation is:

$$FCI = \left\{ V_{EMBED} \; X \sqrt{\left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{TBDH}}{3}\right) + V_{WLUSE}}{2}\right]} \right\}$$

where V$_{\text{EMBED}}$ = the mean embeddedness of the stream channel (n = 30), V$_{\text{LWD}}$ = the number of down woody stems per 25 m of stream, V$_{\text{DETRITUS}}$ = the mean percent detritus cover (n = 6), V$_{\text{TDBH}}$ = the mean diameter at breast height (DBH) of trees with DBH ≥ 10 cm, and V$_{\text{WLUSE}}$ was calculated using land cover data from Maxwell et al. (2014), in ArcMap (ESRI, Redlands, California).

If the mean channel canopy cover is ≤ 20 percent, then the equation is:

$$FCI = \left\{ V_{EMBED}\ X \sqrt{\left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{SSD} + V_{HERB}}{4} \right) + V_{WLUSE}}{4} \right]} \right\}$$

where V_{EMBED} = the mean embeddedness of the stream channel (n = 30), V_{LWD} = the number of down woody stems per 25 m of stream, $V_{DETRITUS}$ = the mean percent detritus cover (n = 6), V_{SSD} = the number of saplings and shrubs per 25 m of stream reach, V_{HERB} = the mean percent cover of herbaceous vegetation (n = 6), and V_{WLUSE} was calculated using land cover data from Maxwell et al. (2014), in ArcMap (ESRI, Redlands, California). Sampling methods for the parameters can be found in Table 4.3.

Occupancy and Abundance Modeling

To assess salamander occupancy and detection probabilities and abundance estimates in an *information-theoretic* framework, I used the Program PRESENCE software (available for download at www.mbr-pwrc.usgs.gov/software/presence). Program PRESENCE was developed to examine the relationship between species presence/abundance and measured parameters. Program PRESENCE fits of these relationships using Akaike's Information Criterion (AIC) and maximum likelihood to determine the best-fit model for capture data (Bailey et al. 2007; Kroll et al. 2010). In particular, within Program PRESENCE, I used "Single-Season" models to examine occupancy and the "Royle Repeated Count" models (also known as N-mixture models) to estimate salamander abundances from repeated site visits (Royle et al. 2004).

The AIC approach uses species data along with habitat data to examine the best-fit relationships between a species and a suite of measured parameters (MacKenzie et al. 2002; 2006). This technique best analyzes observational research by helping to avoid the over-fitting of models that is oftentimes associated when correlating a large number of habitat parameters to species occupancy or abundance. The best-fit models and associated occupancy probabilities (Ψ) and abundance (λ) estimates produced in the data analyses provide foundational and currently unknown probability of occurrence or estimates of abundance for a given species at a site with certain microhabitat or landscape-level conditions.

Additionally, I estimated detection probabilities for each species to determine which environmental conditions most influence detection. Detection is important in the AIC approach to model selection in order to get the best estimates occupancy or abundance. Without considering detection in data analysis, a species true presence may be misclassified as absent when the species was present but not detected. Often this will then underestimate the occupancy probability and abundance estimate (Dorazio et al. 2006; MacKenzie 2006). I used a two-step method to determine which detection covariates to include for each species group in the occupancy analysis (Burnham and Anderson 2002). I ran seven *a priori* detection covariates for each species against the null (intercept) model. The detection covariates included stream flow above base flow, stream flow below base flow, water temperature, air temperature, soil temperature, current weather, and weather in the past 24 hours. Prior to analysis I normalized continuous detection covariates as well as FCI scores using Program PRESENCE in order to compare beta (effect size) values among models.

Results

In my study, I captured 1,145 stream salamanders consisting of nine species during the 207 surveys in the 2013 collection (Table 4.5). Due to the large number of *Eurycea* spp. larval salamanders and the small number of adult *Eurycea,* I combined larval and adult *Eurycea longicauda* and *Eurycea bislineata cirrigera* salamanders to the genus-level for the *Eurycea* spp. group (Table 4.5). Based on results from Sweeten (2015a), the four salamander groups that had sufficient data for occupancy analysis were also used for analysis of abundance: *Desmognathus fuscus, Desmognathus monticola, Desmognathus ochrophaeus, and Eurycea* spp.

Across the 70 sites, Habitat FCI scores ranged from 0.23 to 0.98 with a mean of 0.70 (SD = 0.22). Biogeochemical FCI scores ranged from 0.25 to 0.99 with a mean of 0.72 (SD = 0.21). Hydrology FCI scores ranged from 0.31 to 0.98 with a mean of 0.70 (SD = 0.08).

Occupancy

The **Habitat FCI Model** was the best occupancy model for *Desmognathus fuscus* with an AIC weight of 0.85 (Table 4.6). The beta estimate for the Habitat FCI score was 9.4 (SE = 2.90) and showed a strong positive correlation to *D. fuscus* presence (Table 4.7). Individual site occupancy probabilities for *D. fuscus* ranged from 0.01 to 0.89 (Figure 4.1). The **Hydrology FCI Model** also had empirical support with a Δ AIC of 3.58 and an AIC weight of 0.14 (Table 4.6). There was a strong positive relationship between *D. fuscus* presence and the Hydrology FCI score with a beta estimate of 10.66 (SE = 3.38; Table 4.7). With a Hydrology FCI score of 0.31, *D. fuscus* occupancy estimates were 0.01. Occupancy estimates increased to 0.93 when the Hydrology FCI score improved to 0.98 (Figure 4.2).

The only occupancy model with empirical support for *Desmognathus monticola* was the **Biogeochemical FCI Model** with an AIC weight of 0.97 (Table 4.6). *D. monticola* showed a strong positive correlation to the Biogeochemical FCI score with a beta estimate of 10.89 (SE = 2.95; Table 4.7). Individual site occupancy probabilities ranged from 0.04 to 0.94 (Figure 4.3).

The top occupancy model for *Desmognathus ochrophaeus* was the **Habitat FCI Model,** which had an AIC weight of 0.63 (Table 4.6). Occupancy of *D. ochrophaeus* was positively correlated to the Habitat FCI scores (Beta estimate = 4.43; SE = 1.40; Table 4.7), and individual site occupancy estimates ranged from 0.14 to 0.82 (Figure 4.1). There was also strong empirical support for the **Biogeochemical FCI Model** with a Δ AIC of 1.11 and an AIC weight of 0.36 (Table 4.6). Beta estimates (4.41; SE = 1.47) for the Biogeochemical FCI score showed similar effect size to that of the Habitat FCI score (Table 4.7). Occupancy estimates for individual sites ranged from 0.15 to 0.82 for *D. ochrophaeus* (Figure 4.3).

The top occupancy model for *Eurycea* spp. was the **Hydrology FCI Model** with an AIC weight of 0.47 (Table 4.6). The beta estimate of 2.90 (SE = 1.77) shows a positive correlation between *Eurycea* spp. occupancy and Hydrology FCI scores (Table 4.7). Individual site occupancy estimations ranged from 0.50 to 0.87 (Figure 4.2). The **Biogeochemical FCI Model** also had strong empirical support for *Eurycea* spp. with a Δ AIC of 0.91 (Table 4.6). However, the beta estimate for the Biogeochemical FCI score was low at 0.45 with a large standard error of 1.43. Individual site estimates of occupancy for *Eurycea* spp. only ranged from0.70 to 0.76 (Figure 4.3). Additionally, there was moderate empirical support for the **Habitat FCI Model** that had a Δ AIC of 2.91 and an AIC weight of 0.11 (Table 4.6). The beta estimate for the Habitat FCI score was small at 0.11 (SE = 1.30), and individual site occupancy estimations ranged from 0.73 to 0.74 (Table 4.7; Figure 4.1).

Abundance

The top abundance model for *Desmognathus fuscus* is the **Habitat FCI Model** with an AIC weight of 0.60 (Table 4.8), and a beta estimate of 5.04 (SE = 1.11; Table 4.9). Individual site abundance estimates for *D. fuscus* range from 0.1 to 3.1 salamanders per site (Figure 4.4). The **Hydrology FCI Model** also had strong empirical support for *D. fuscus* abundance with a Δ AIC of 0.83 and an AIC weight of 0.40 (Table 4.8). The beta estimate for the Hydrology FCI score was 5.58 (SE = 1.16) indicating a positive relationship between the Hydrology FCI score and *D.*

fuscus abundance (Table 4.9). Individual site estimates of abundance range from 0.1 to 4.0 (Figure 4.5).

The only abundance model with empirical support for *Desmognathus monticola* was the **Habitat FCI Model** which had an AIC weight of 0.9999 (Table 4.8). The beta estimate was 5.4 with a standard error of 0.63 (Table 4.9). Abundance estimates by site ranged from 0.2 to 10.4 for *D. monticola* (Figure 4.4).

Desmognathus ochrophaeus had one abundance model with empirical support, the **Habitat FCI Model** (AIC weight = 0.9994; Table 4.8). The beta estimate for the Habitat FCI score was 4.23 (SE = 0.58) and showed a positive correlation to *D. ochrophaeus* abundances (Table 4.9). Individual site estimates of abundance ranged from 0.4 to 8.5 (Figure 4.4).

The **Hydrology FCI Model** was the only abundance model with empirical support for *Eurycea* spp. with an AIC weight of 0.88 (Table 4.8). The beta estimate was 1.23 (SE = 0.42) and individual site abundance estimates were 2.7 to 6.2 *Eurycea* spp. per site (Table 4.9; Figure 4.5).

Discussion

Overall, my study found the three *Desmognathus* species were most correlated to the **Habitat FCI Model** with this model performing best for *D. fuscus* and *D. ochrophaeus* in the occupancy analysis. In the abundance analysis, the **Habitat FCI Model** was the top model for all three *Desmognathus* species. With 11 parameters, the Habitat FCI score takes into account more parameters than either the Biogeochemical FCI score or the Hydrology FCI score. Many of the Habitat FCI parameters measure riparian conditions such as canopy cover, riparian vegetation species richness, detritus cover, herbaceous cover, and LWD counts.

The relationship between *Desmognathus* spp. and the Habitat FCI score may be a function of direct correlation to the parameters in the Habitat FCI. Past research has shown strong associations between canopy cover and *Desmognathus* abundance (Crawford and Semlitsch 2008; Sweeten 2015b; Ward et al. 2008). *Desmognathus* salamanders are lungless and therefore are required to constantly have moist skin in order for oxygen exchange across the skin membrane (Petranka 1998). Lungless salamanders have a high risk of desiccation particularly when foraging away from the stream. Canopy cover not only provides cover from solar exposure lowing soil, stream, and air temperatures, it also increases other microhabitat parameters associated with maintaining a cool, moist environment such as detritus cover. Crawford and Semlitsch (2008) found a positive correlation between detritus depth and *D. monticola* and *E. b. cirrigera*. In the southern Appalachians of North Carolina, Harper and Guynn (1999) found more salamanders including *D. ochrophaeus* and *D. aeneus* in moist microhabitats with increased detritus depths.

Habitat FCI scores closer to 1.0 indicate forest-like, less-disturbed conditions. *Desmognathus* spp. have been shown to be positively correlated with characteristics often found in more mature forests such as high canopy cover, native tree species dominance, high detritus cover, and more LWD. In the central Appalachians of West Virginia, Moseley et al. (2008) found a positive relationship between *Desmognathus* spp. abundance and time since forest harvest. Ford et al. (2002) observed that *Desmognathus* spp. abundance was most correlated to increasing basal area (as a surrogate for stand-age/time since disturbance) within a forest stand in southern Appalachians of northern Georgia. Additionally, in a review of North American literature on amphibian ecology and forest management deMaynadier and Hunter (1995) suggested that more salamanders in older forests functionally were an indirect measure of microhabitat conditions such as LWD, detritus cover, and canopy.

Lower Habitat FCI scores indicate riparian habitat that is disturbed with increased herbaceous cover and areas dominated by invasive species. Walz (2002) found decreased abundances of *D. fuscus* and *D. ochrophaeus* in agricultural fields and pastures. Wood and Williams (2013) found lower abundances of *Desmognathus* in reclaimed grassland and shrubland where there was less detritus, lower stem densities, less LWD, less canopy cover, and an increase in invasive herbaceous species such as *Lespedeza* as compared to forested or partially forested sites. Invasive herbaceous species may just indicate recent disturbance and grassland conditions, or it may be that invasive plant species do not produce the necessary forest-like microhabitat (i.e. leaf litter, cover, and LWD) to provide the cool, moist habitat needed for salamanders (Lemke et al. 2012).

Unlike the results for *Desmognathus* spp., none of the three FCI models worked well with *Eurycea* spp. in either the occupancy or abundance analysis. The occupancy analysis of *Eurycea* spp. showed empirical support for all three FCI models; however, the betas for the FCI scores were low with large standard errors.

The differences found in this study between *Desmognathus* and *Eurycea* may be explained by the theory that stream salamanders can be grouped as either disturbance avoiders (i.e. *Desmognathus*) or disturbance tolerant (i.e. *Eurycea*) (Surasinghe and Baldwin 2015). Disturbance avoiders are generally long-lived, large-bodied salamanders that are dependent on forests and are sensitive to riparian disturbances. Disturbance tolerant species often can be characterized as short-lived, small bodied, microhabitat generalists that can withstand riparian land uses (Surasinghe and Baldwin 2015). Species that comprise these groups may change in different ecoregions. For example Surasinghe and Baldwin (2015) found that *D. quadramaculatus* and *G. porphyriticus* to be disturbance avoiders in the Blue Ridge ecoregion of the southern Appalachians, however, in the Piedmont ecoregion these species were considered to be disturbance tolerant. Based on results from my study, *Desmognathus* spp. seem to be disturbance avoiders whereas *Eurycea* spp. were disturbance tolerant. Research throughout Appalachia proper has found that in undisturbed areas *Desmognathus* spp. is the dominant stream salamander group whereas in disturbed areas *Eurycea* spp. is the dominant stream salamander (Hamilton 2002; Hyde and Simons 2001; Resetarits 1997). Ward et al. (2008) also had similar findings in central Appalachia. Roads seemed to benefit disturbance tolerant salamanders (*E. b. cirrigera*) as abundances of this species was higher at roadside sites as compared to forested control sites while *Desmognathus* spp. had the inverse response (Ward et al. 2008). Riparian disturbances, such as roads, may cause stream salamander communities to shift to disturbance tolerant species without changes in overall abundance (Ward et al. 2008). This indicates the need for research to separate stream salamanders to species rather than examining the total, grouped salamander abundance. *Eurycea* are opportunistic generalists with diets largely consisting of pollutant tolerant benthic macroinvertebrates such as *Chironomids* (Barrett et al. 2012; Burton 1976; Muenz et al. 2008; Petranka 1984). One reason *Eurycea* is disturbance tolerant may be that *Eurycea* are better able to tolerate poor water quality and riparian habitat conditions because prey items (*Chironomids*) are often readily available even in degraded conditions. It is important to note however that even disturbance tolerant species are not disturbance-adapted, and these species are not immune to localized extirpations if certain environmental thresholds are exceeded (Surasinghe and Baldwin 2015).

The management implications of my study show that the Habitat FCI score is a good measure of physical instream and riparian conditions and is reflected in both *Desmognathus* spp. occupancy and abundance. My results show that *Desmognathus* spp. are a good indicator of physical stream and riparian conditions. Although the Habitat FCI individual site estimate line slopes for occupancy and abundance are slightly different the relationship to *Desmognathus* is similar (Figure 4.1; Figure 4.5). These results support the use of a presence/absence sampling design of *Desmognathus* to examine physical stream and riparian conditions. My findings also show that *Eurycea* spp. should not be used as an indicator taxa.

Table 4.1 Instream and riparian habitat measurements for the Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) along with sampling gear used and sampling details, southwest Virginia, summer 2013.

Components of Hydrology FCI	Sampling Gear	Sampling Method
Substrate Embeddedness	Estimation	Mean embeddedness rating (scale of 1 to 5) of 30 pieces of stream substrate
Substrate Size	Ruler	Mean size (cm) of 30 pieces of stream substrate
Stream Bank Erosion	Estimation	Total percent of left and right stream bank erosion along the 25 m quadrat (up to 200 % possible)
Large Woody Debris (LWD) Density	Count	Number of pieces of dead, down wood (>10 cm diameter and >90 cm long) in 25 m by 10 m quadrat[1]
Land Use	GIS	Weighted average of runoff score for subwatershed

[1] Extrapolated out to calculate number for 30.5 m stream length

Table 4.2 Instream and riparian habitat measurements for the Habitat Functional Capacity Index (FCI; see Noble et al. 2010) along with sampling gear used and sampling details, southwest Virginia, summer 2013.

Components of Habitat FCI	Sampling Gear	Sampling Method
Canopy Cover[1]	Spherical Convex Densiometer	Mean of three measurements taken from center of stream at meter 0, 12.5, and 25
Substrate Embeddedness	Estimation	Mean embeddedness rating (scale of 1 to 5) of 30 pieces of stream substrate
Substrate Size	Ruler	Mean size (cm) of 30 pieces of stream substrate
Large Woody Debris (LWD) Density	Count	Number of pieces of dead, down wood (>10 cm diameter and >90 cm long) in 25 m by 10 m quadrat[3]
Tree DBH[1]	Tree Calipers	Mean DBH of all trees (>10 cm DBH) in 25 m by 10 m quadrat
Tree Snag Density	Count	Number of standing, dead trees (>10 cm DBH) in 25 m by 10 m quadrat[3]
Sapling & Shrub Density[2]	Count	Number of saplings and shrubs (<10 cm DBH) in 25 m by 10 m quadrat[3]
Vegetation Species Richness	Count	Number of species in Group 1 (in tallest strata) and Group 2 (in any strata) present in 25 m by 10 m quadrat
Detritus Cover	Estimation	Mean percent of 6 estimations of cover from leaves, sticks, and organic material taken 5 m from stream center on right and left side at meter 0, 12.5, and 25
Herbaceous Cover[2]	Estimation	Mean percent 6 estimations of herbaceous cover taken 5 m from stream center on right and left side at meter 0, 12.5, and 25
Land Use	GIS	Weighted average of runoff score for subwatershed

[1] Only used if canopy cover ≥ 20 percent
[2] Only used if canopy cover ≤ 20 percent
[3] Extrapolated out to calculate density for 30.5 m stream length

Table 4.3 Instream and riparian habitat measurements for the Biogeochemical Functional Capacity Index (FCI; see Noble et al. 2010) along with sampling gear used and sampling details, southwest Virginia, summer 2013.

Components of Biogeochemical FCI	Sampling Gear	Sampling Method
Substrate Embeddedness	Estimation	Mean embeddedness rating (scale of 1 to 5) of 30 pieces of stream substrate
Large Woody Debris (LWD) Density	Count	Number of pieces of dead, down wood (>10 cm diameter and >90 cm long) in 25 m by 10 m quadrat[3]
Tree DBH[1]	Tree Calipers	Mean DBH of all trees (>10 cm DBH) in 25 m by 10 m quadrat
Sapling & Shrub Density[2]	Count	Number of saplings and shrubs (<10 cm DBH) in 25 m by 10 m quadrat[3]
Detritus Cover	Estimation	Mean percent of 6 estimations of cover from leaves, sticks, and organic material taken 5 m from stream center on right and left side at meter 0, 12.5, and 25
Herbaceous Cover[2]	Estimation	Mean percent 6 estimations of herbaceous cover taken 5 m from stream center on right and left side at meter 0, 12.5, and 25
Land Use	GIS	Weighted average of runoff score for subwatershed

[1] Only used if canopy cover ≥ 20 percent

[2] Only used if canopy cover ≤ 20 percent

[3] Extrapolated out to calculate density for 30.5 m stream length

Table 4.4 Information for the five HUC-12 study watersheds used in 2013 including watershed name, HUC-12 identification number, Virginia County(s) where the watershed is located and the area of the watershed.

Watershed Name	HUC-12	County	Watershed area (sq. km)
Callahan Creek	60102060103	Wise	54.7
Dumps Creek	60102050402	Dickenson; Russell	82.3
Pigeon Creek	60102060104	Wise	58.9
Roaring Fork	60102060101	Wise	66
Rocky Fork	60102050501	Wise	91

Table 4.5 Total number of salamanders caught by species along with the number of sampling sites and sampling events the species were found in summer 2013, southwest Virginia.

Scientific Name	**Total Captured**	**# of Sampling Sites (n = 70)**	**# Sampling Events (n = 207)**
Desmognathus fuscus	82	27	45
Desmognathus monticola	291	40	81
Desmognathus ochrophaeus	227	37	75
Desmognathus welteri	41	18	21
Eurycea b. cirrigera	24	15	19
Eurycea longicauda	15	7	8
Eurycea Larvae	442	42	85
Gyrinophilus porphyriticus	11	10	11
Pseudotriton montanus	5	3	3
Pseudotriton ruber	7	6	6

Table 4.6 Model results for occupancy probabilities of Functional Capacity Index scores (FCI; see Noble et al. 2010) including the number of parameters in each model (K), Akaike's Information Criterion (AIC) rankings, Δ AIC, and AIC weight (ω_i) for the four stream salamander groups, southwest Virginia, 2013.

FCI Occupancy Models	**K**	**AIC**	**Δ AIC**[1]	**ω_i**
Desmognathus fuscus				
Ψ(HabitatFCI), p(Rain24)[2]	4	173.53	0.00	0.8516
Ψ(HydrologyFCI), p(Rain24)	4	177.11	3.58	0.1422
Ψ(BiogeochemicalFCI), p(Rain24)	4	183.38	9.85	0.0062
Ψ(Null), p(Rain24)	3	198.22	24.69	0.0000
Desmognathus monticola				
Ψ(BiogeochemicalFCI), p(SoilT)[3]	4	210.86	0.00	0.9707
Ψ(HabitatFCI), p(SoilT)	4	217.87	7.01	0.0292
Ψ(HydrologyFCI), p(SoilT)	4	228.97	18.11	0.0001
Ψ(Null), p(SoilT)	3	242.39	31.53	0.0000
Desmognathus ochrophaeus				
Ψ(HabitatFCI), p(SoilT*Rain)	5	210.25	0.00	0.6263
Ψ(BiogeochemcialFCI), p(SoilT*Rain)[4]	5	211.36	1.11	0.3595
Ψ(HydrologyFCI), p(SoilT*Rain)	5	218.61	8.36	0.0096
Ψ(Null), p(SoilT*Rain)	4	220.09	9.84	0.0046
Eurycea spp.				
Ψ(HydrologyFCI), p(SoilT*Rain)	5	260.06	0.00	0.4732
Ψ(Null), p(SoilT*Rain)	4	260.97	0.91	0.3002
Ψ(BiogeochemicalFCI), p(SoilT*Rain)	5	262.87	2.81	0.1161
Ψ(HabitatFCI), p(SoilT*Rain)	5	262.97	2.91	0.1104

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support. Models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

[2] Rain24 is a binomial of rain in the past 24 hours

[3] SoilT is Soil Temperature

[4] Rain is a binomial for current weather rainy

Table 4.7 Beta estimates and standard errors for each Functional Capacity Index (FCI; see Noble et al. 2010) covariate in the top occupancy models (Δ AIC < 4) for four salamander groups, southwest Virginia, 2013.

Top Occupancy FCI Models	**Beta**	**Standard Error**
Desmognathus fuscus		
Ψ.HabitatFCI	9.4156	2.8996
Ψ.HydrologyFCI	10.6577	3.3821
Desmognathus monticola		
Ψ.BiogeochemicalFCI	10.8927	2.9531
Desmognathus ochrophaeus		
Ψ.HabitatFCI	4.4261	1.3975
Ψ.BiogeochemicalFCI	4.4126	1.4660
Eurycea spp.		
Ψ.HydrologyFCI	2.8970	1.7669
Ψ.BiogeochemicalFCI	0.4496	1.4295
Ψ.HabitatFCI	0.1112	1.2998

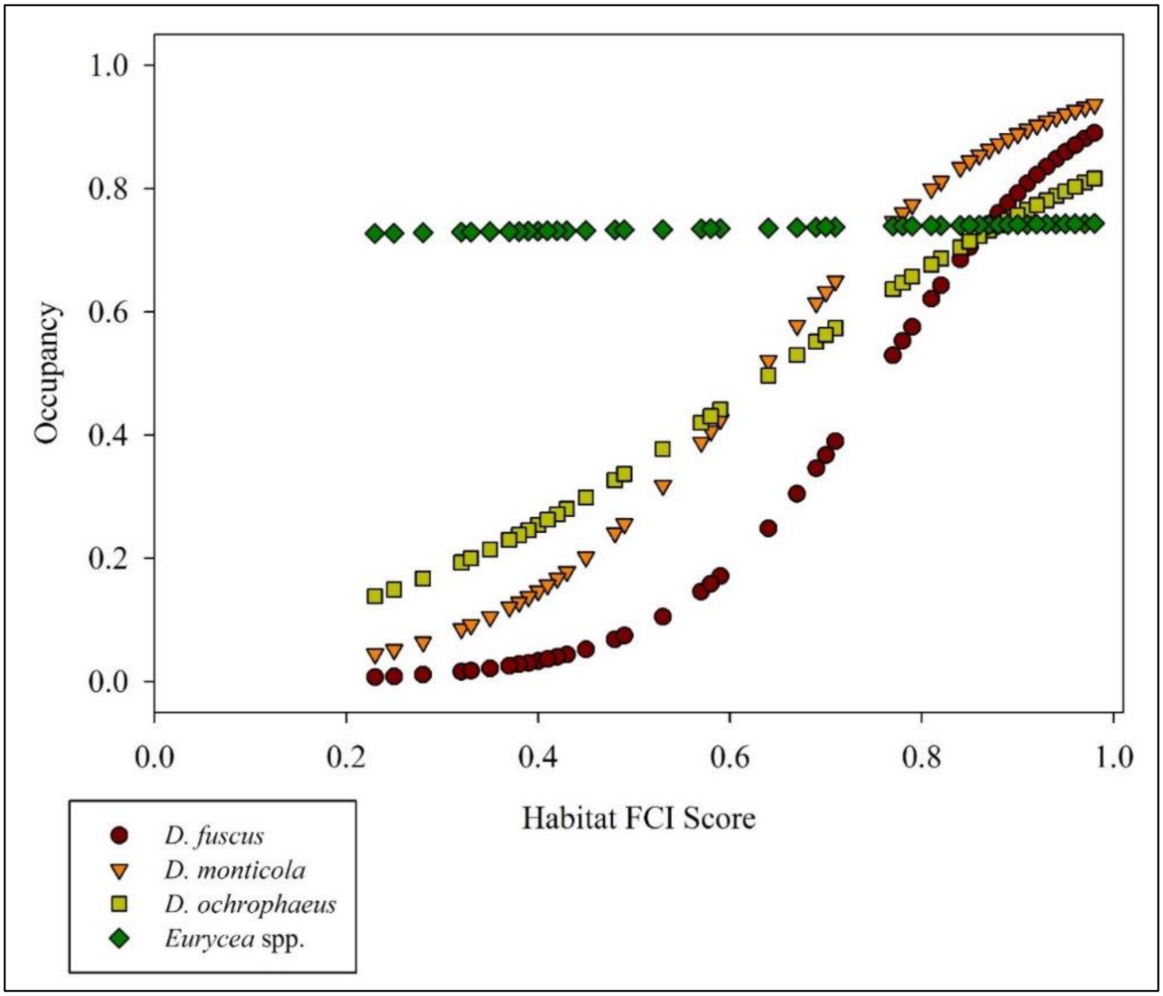

Figure 4.1 Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimations of occupancy (Ψ) for the four salamander groups, southwest Virginia, summer 2013.

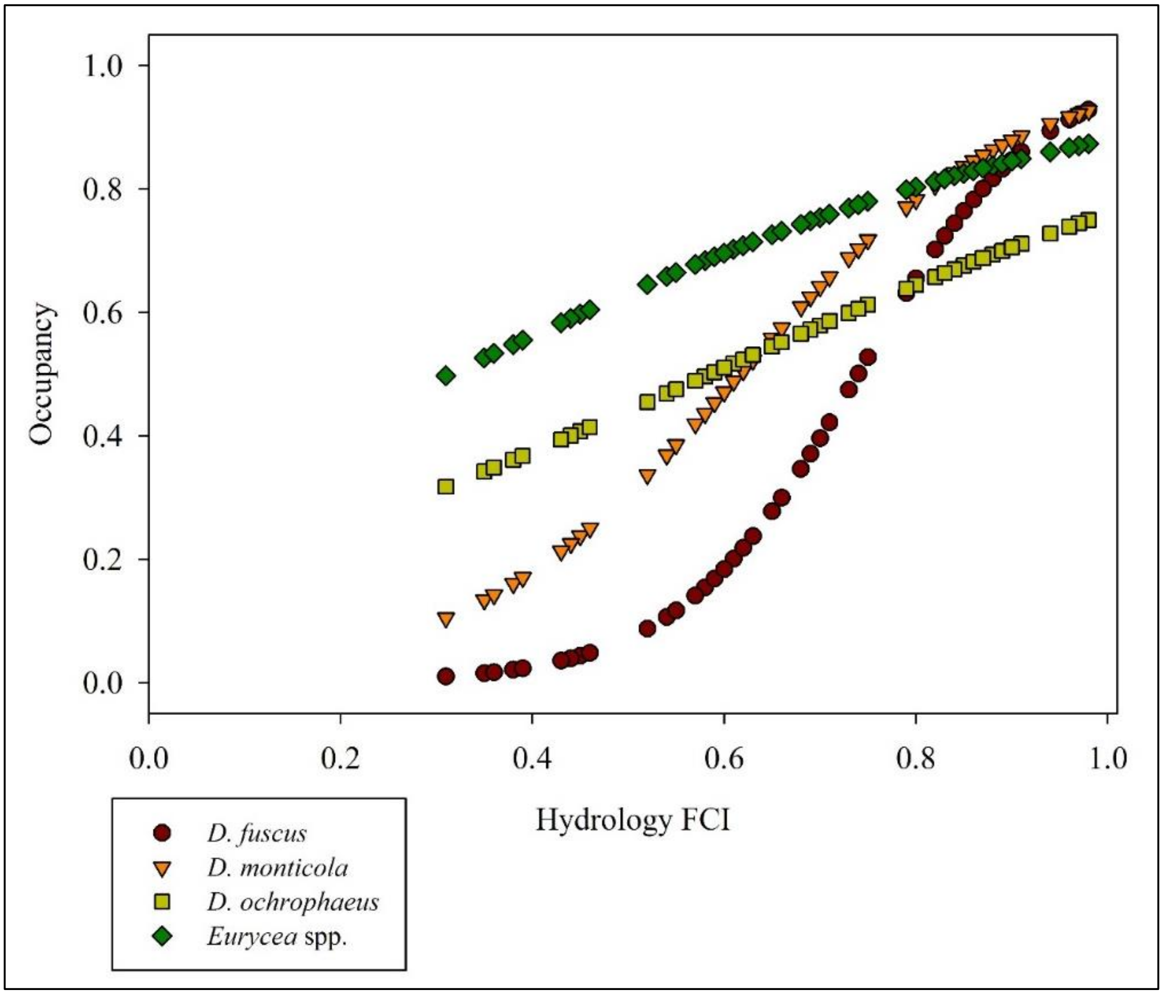

Figure 4.2 Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimations of occupancy (Ψ) for the four salamander groups, southwest Virginia, summer 2013.

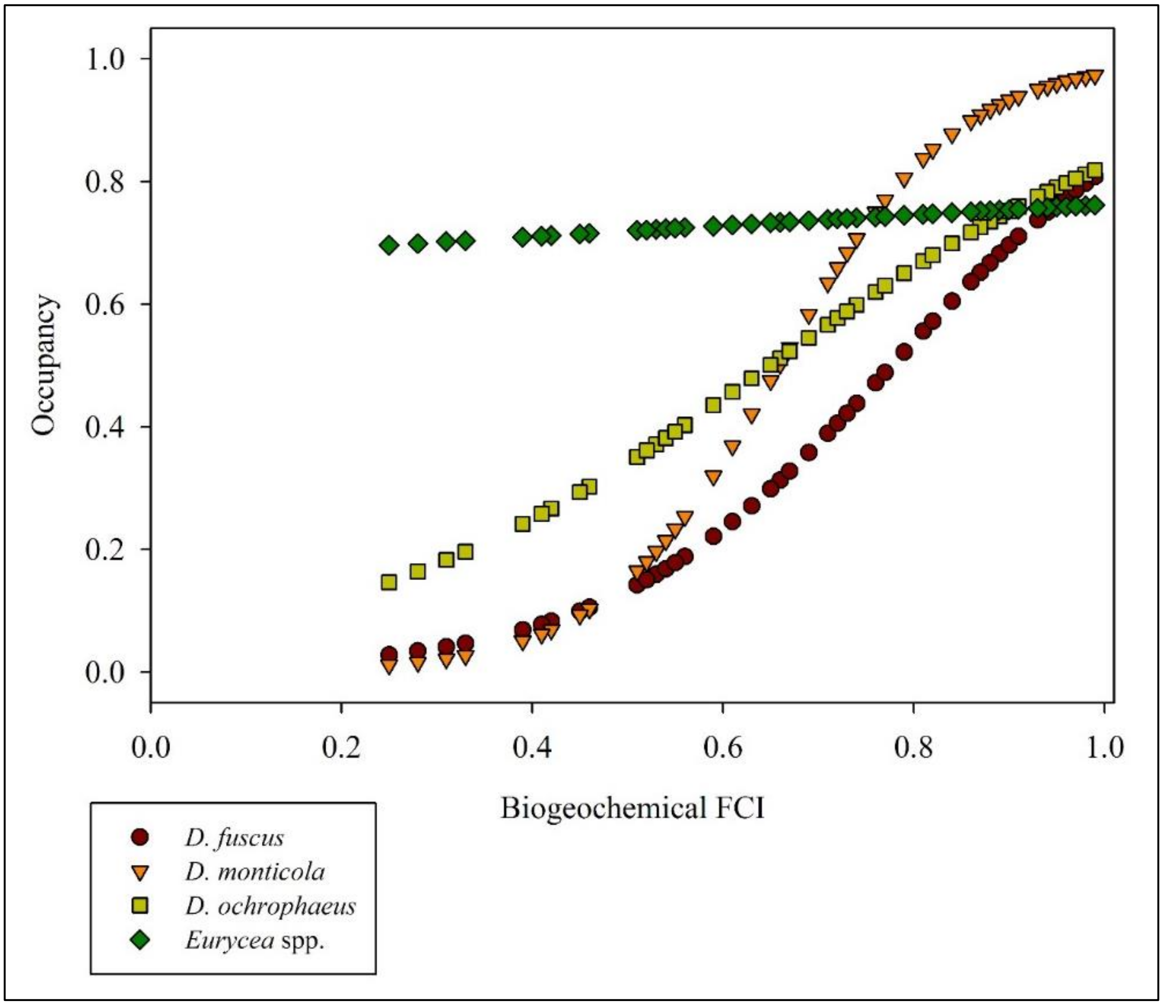

Figure 4.3 Biogeochemical Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimations of occupancy (Ψ) for the four salamander groups, southwest Virginia, summer 2013.

Table 4.8 Model results for abundance estimates (λ) of Functional Capacity Index (FCI; see Noble et al. 2010) scores including the number of parameters in each model (K), Akaike's Information Criterion (AIC) rankings, Δ AIC, and AIC weight (ω_i) for the four stream salamander groups, southwest Virginia, 2013.

FCI Abundance Models	K	AIC	Δ AIC[1]	ω_i
Desmognathus fuscus				
λ(HabitatFCI), p(SoilT*FlowAbove)[2,3]	5	300.68	0.00	0.6022
λ (HydrologyFCI), p(SoilT*FlowAbove)	5	301.51	0.83	0.3976
λ(BiogeochemicalFCI), p(SoilT*FlowAbove)	5	316.89	16.21	0.0002
λ(Null), p(SoilT*FlowAbove)	4	332.52	31.84	0.0000
Desmognathus monticola				
λ(HabitatFCI), p(SoilT*FlowBelow)[4]	5	644.06	0.00	0.9999
λ(BiogeochemicalFCI), p(SoilT*FlowBelow)	5	663.39	19.33	0.0001
λ(HydrologyFCI), p(SoilT*FlowBelow)	5	718.45	74.39	0.0000
λ(Null), p(SoilT*FlowBelow)	4	767.56	123.50	0.0000
Desmognathus ochrophaeus				
λ(HabitatFCI), p(AirT*Rain)[5]	5	600.84	0.00	0.9994
λ(BiogeochemicalFCI), p(AirT*Rain)	5	615.60	14.76	0.0006
λ(HydrologyFCI), p(AirT*Rain)	5	664.87	64.03	0.0000
λ(Null), p(AirT*Rain)	4	675.88	75.04	0.0000
Eurycea spp.				
λ(HydrologyFCI), p(SoilT*Rain)	5	822.64	0.00	0.8798
λ(BiogeochemicalFCI), p(SoilT*Rain)	5	827.36	4.72	0.0831
λ(Null), p(SoilT*Rain)	4	829.60	6.96	0.0271
λ(HabitatFCI), p(SoilT*Rain)	5	831.59	8.95	0.0100

[1] Models with a ΔAIC < 2 are considered to have a substantial level of empirical support while models with a ΔAIC of 2 – 4 are considered to have a moderate level of empirical support.

[2] SoilT is soil temperature

[3] FlowAbove is a binomial for flow above base flow

[4] FlowBelow is a binomial for flow below base flow

[5] AirT is air temperature

Table 4.9 Beta estimates and standard errors for each Functional Capacity Index (FCI; see Noble et al. 2010) covariate in the top abundance models (Δ AIC < 4) for the four salamander groups, southwest Virginia, 2013.

Top Abundance FCI Models	**Beta**	**Standard Error**
Desmognathus fuscus		
λ.HabitatFCI	5.0402	1.1057
λ.HydrologyFCI	5.5785	1.1638
Desmognathus monticola		
λ.HabitatFCI	5.3807	0.6274
Desmognathus ochrophaeus		
λ.HabitatFCI	4.2315	0.5802
Eurycea spp.		
λ.HydrologyFCI	1.2330	0.4193

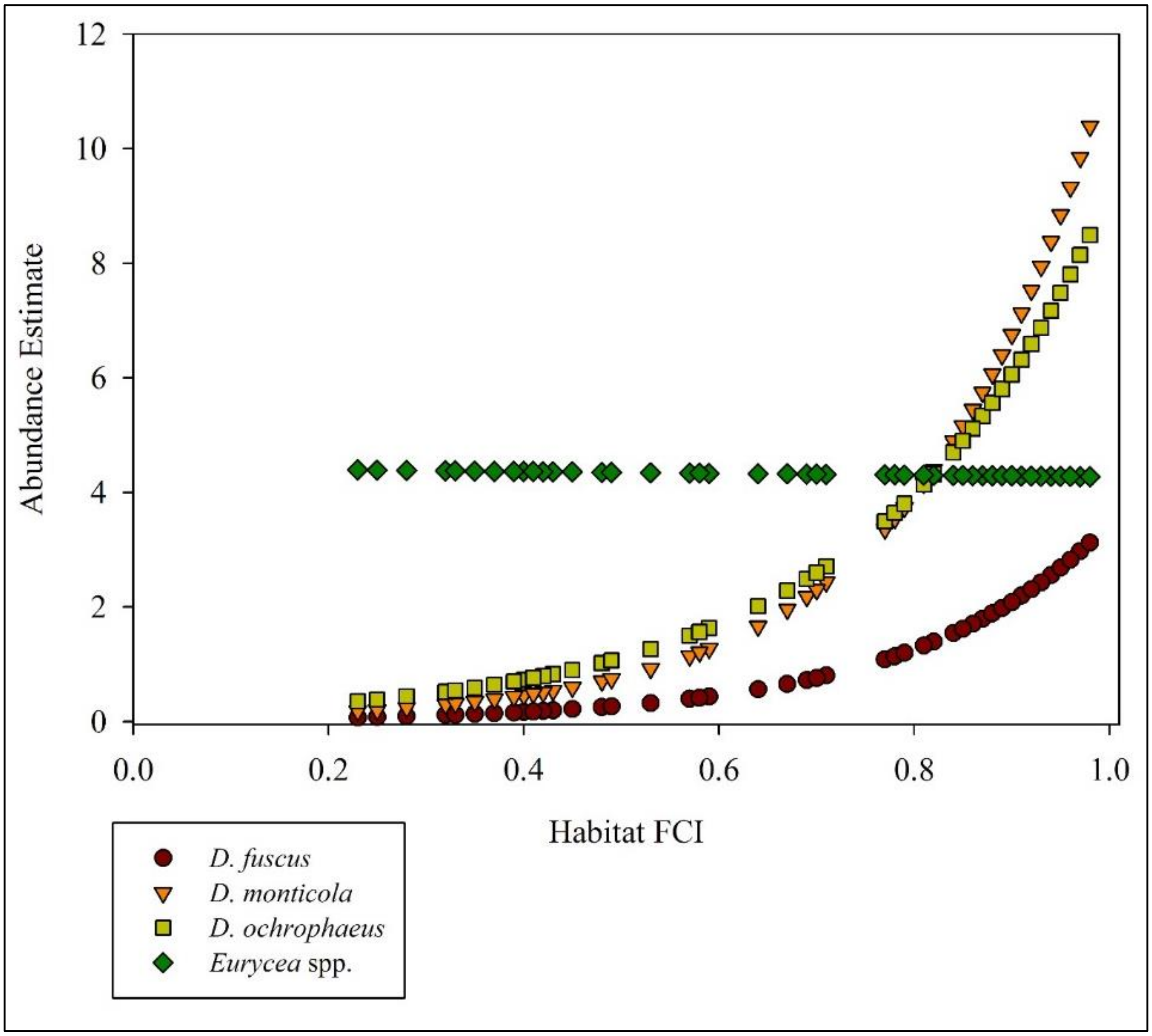

Figure 4.4 Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimations of abundance (λ) per 25m x 5 m quadrat for the four salamander groups, southwest Virginia, summer 2013.

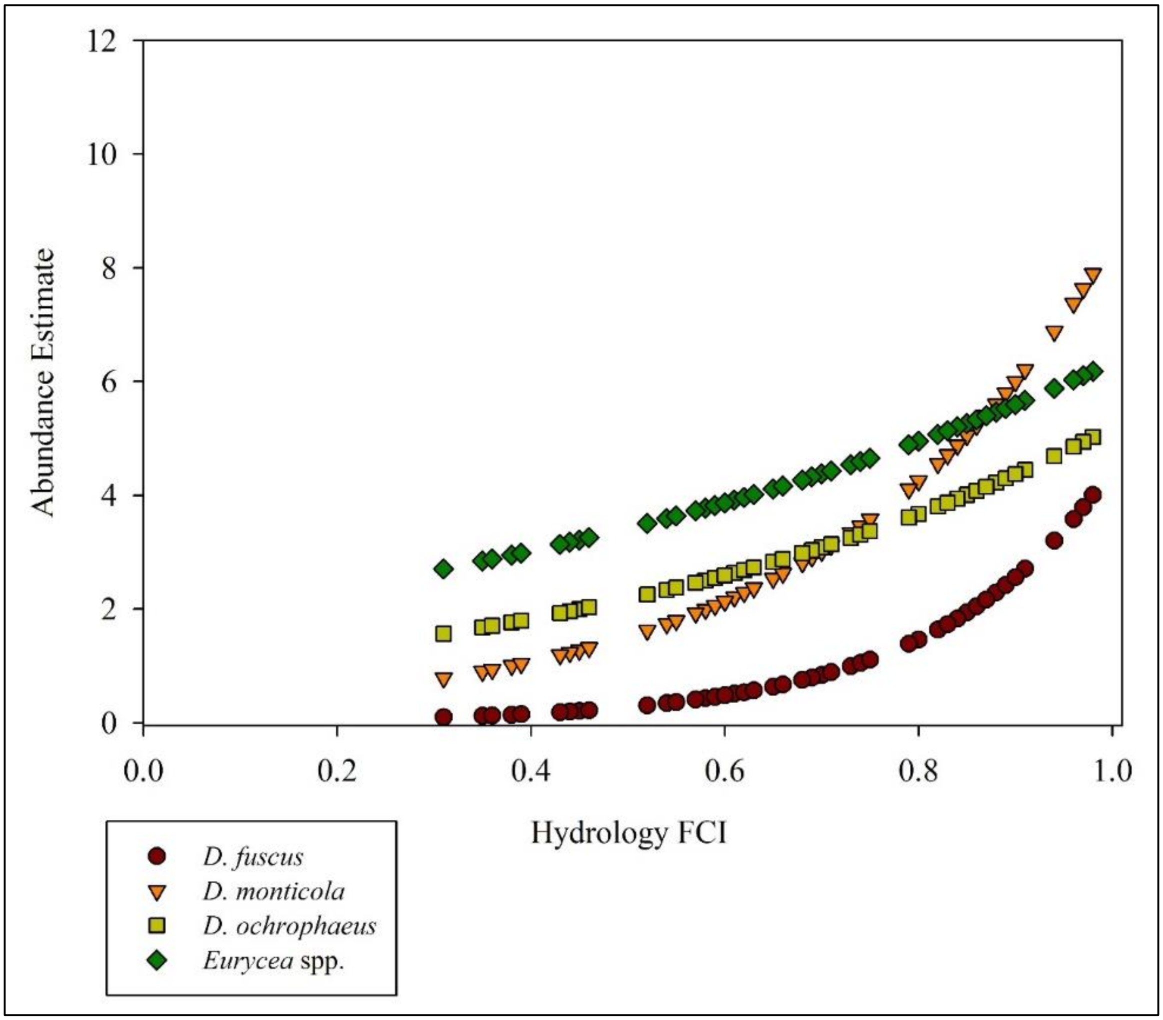

Figure 4.5 Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimations of abundance (λ) per 25 m x 5 m quadrat for the four salamander groups, southwest Virginia, summer 2013.

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Appendix C

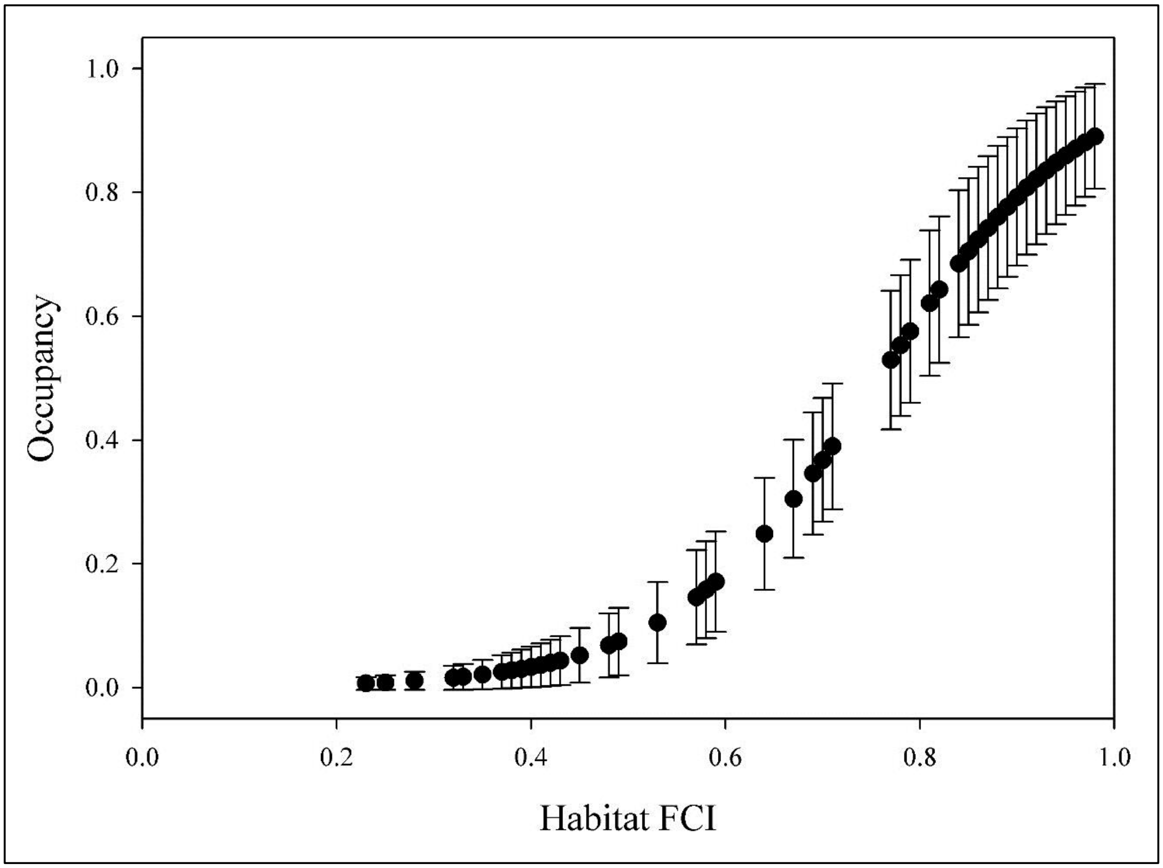

Figure C.1 Individual site estimates of occupancy (Ψ) and standard errors of the Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Desmognathus fuscus*, southwest Virginia, summer 2013.

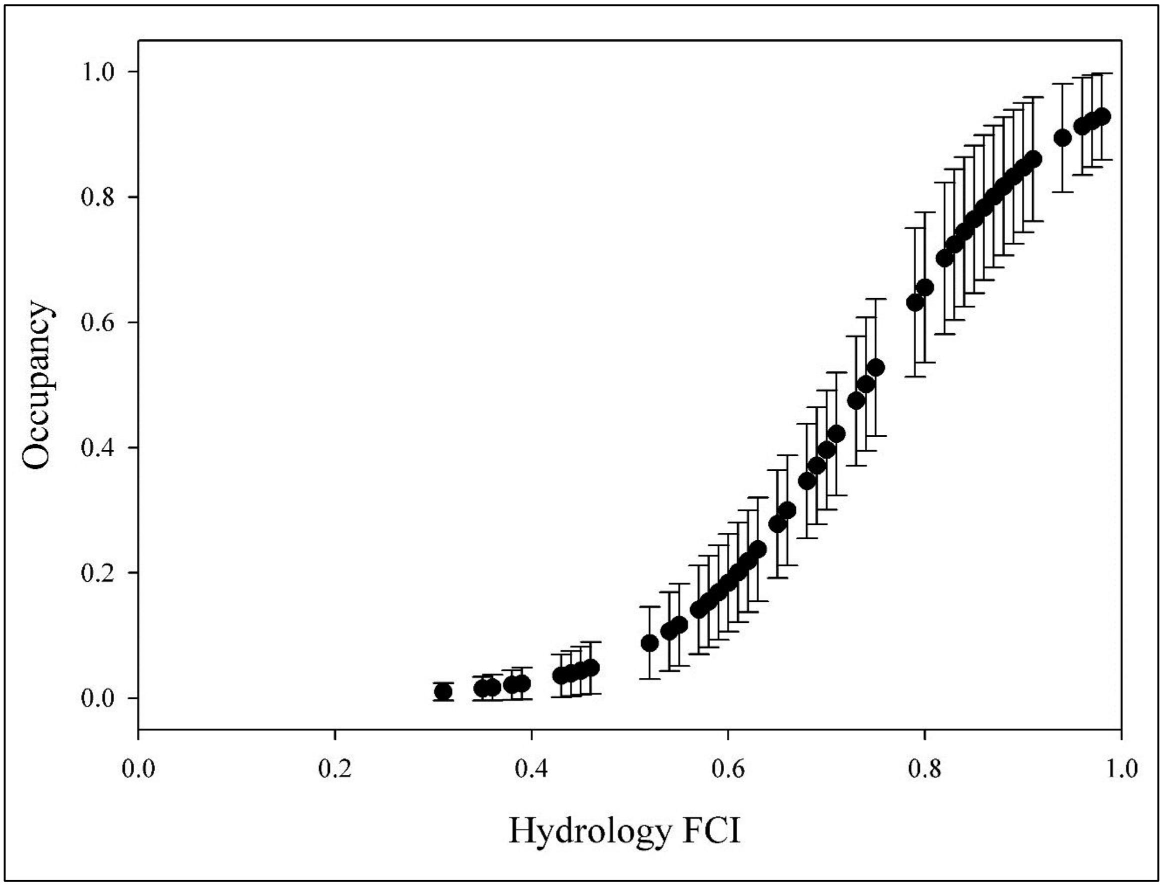

Figure C.2 Individual site estimates of occupancy (Ψ) and standard errors of the Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Desmognathus fuscus*, southwest Virginia, summer 2013.

Table C.1 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) occupancy models for *Desmognathus fuscus*, southwest Virginia, summer 2013.

FCI Occupancy Models and Covariates for *D. fuscus*	**Beta**	**Standard Error**
Ψ(HabitatFCI), p(Rain24)		
Ψ	-7.1332	2.1309
Ψ.HabitatFCI	9.4156	2.8996
p	0.6553	0.5095
p.Rain24	-1.0295	0.5096
Ψ(HydrologyFCI), p(Rain24)		
Ψ	-7.8818	2.3895
Ψ.HydrologyFCI	10.6577	3.3821
p	0.6706	0.5107
p.Rain24	-1.0558	0.5169
Ψ(BiogeochemicalFCI), p(Rain24)		
Ψ	-5.2329	1.5661
Ψ.BiogeochemicalFCI	6.7369	2.0575
p	0.7470	0.5010
p.Rain24	-1.0715	0.5250
Ψ(Null), p(Rain24)		
Ψ	-0.2417	0.2861
p	0.7403	0.5091
p.Rain24	-1.0653	0.5331

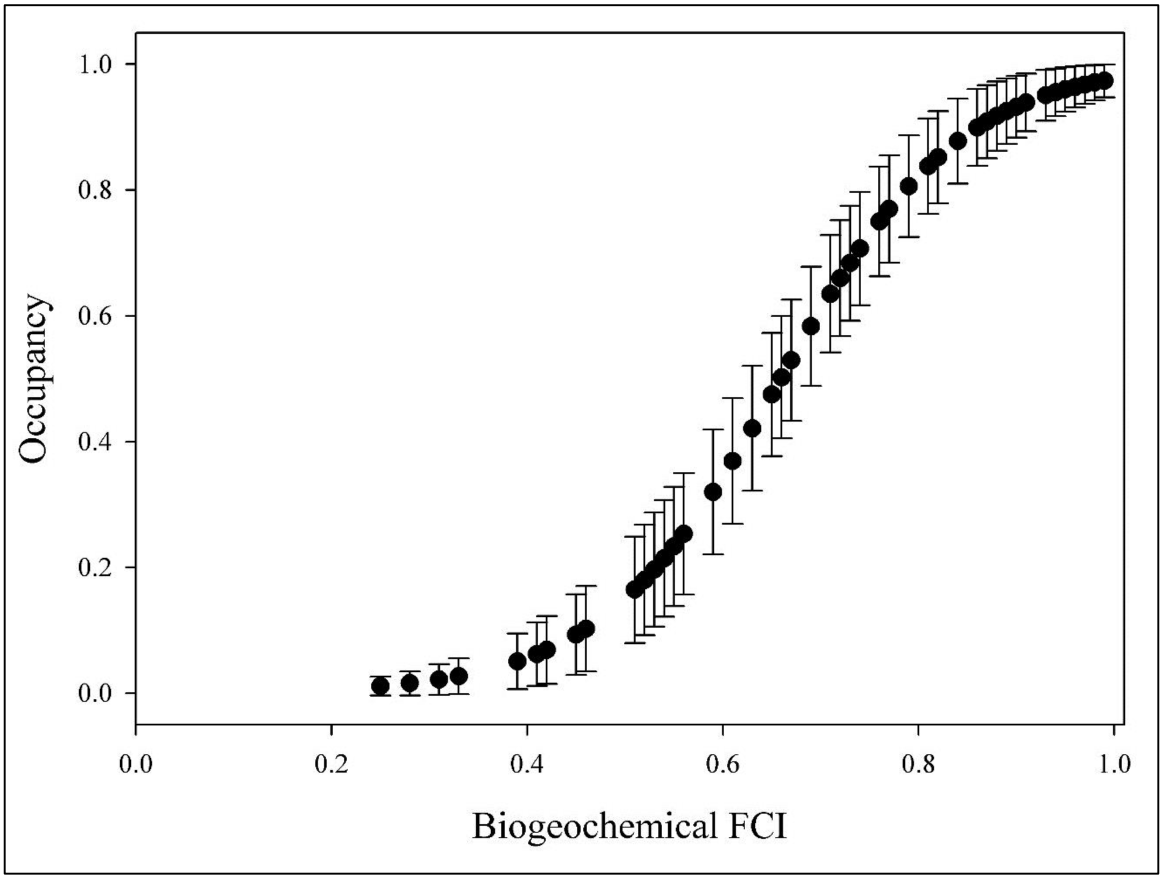

Figure C.3 Individual site estimates of occupancy (Ψ) and standard errors of the Biogeochemical Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Desmognathus monticola*, southwest Virginia, summer 2013.

Table C.2 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) occupancy models for *Desmognathus monticola*, southwest Virginia, summer 2013.

FCI Occupancy Models and Covariates for *D. monticola*	**Beta**	**Standard Error**
Ψ(BiogeochemicalFCI), p(SoilT)		
Ψ	-7.1809	2.0239
Ψ.BiogeochemicalFCI	10.8927	2.9531
p	0.5574	0.2138
p.SoilT	-0.3951	0.2431
Ψ(HabitatFCI), p(SoilT)		
Ψ	-4.8253	1.3427
Ψ.HabitatFCI	7.6658	1.9616
p	0.5741	0.2161
p.SoilT	-0.4062	0.2432
Ψ(HydrologyFCI), p(SoilT)		
Ψ	-4.3121	1.4790
Ψ.HydrologyFCI	6.9923	2.2297
p	0.5107	0.2262
p.SoilT	-0.4841	0.2439
Ψ(Null), p(SoilT)		
Ψ	0.4739	0.2799
p	0.5098	0.2265
p.SoilT	-0.5415	0.2474

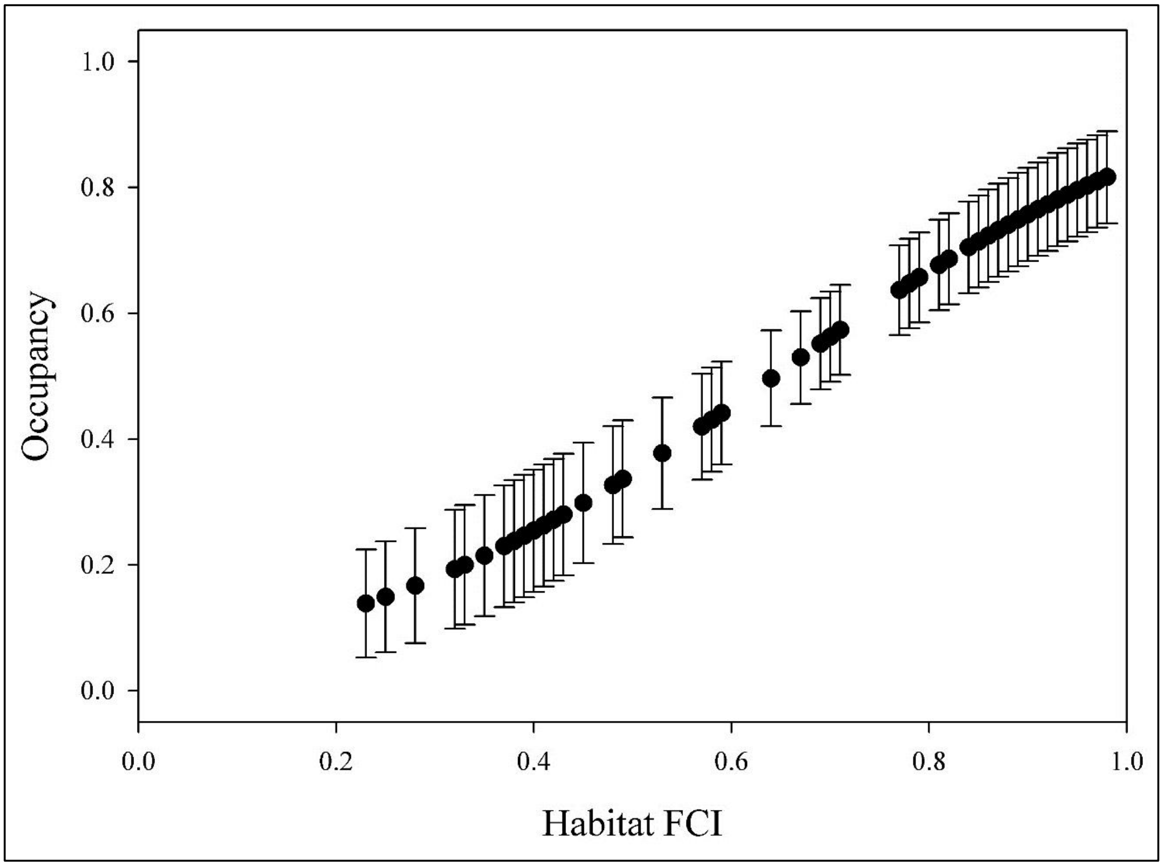

Figure C.4 Individual site estimates of occupancy (Ψ) and standard errors of the Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Desmognathus ochrophaeus*, southwest Virginia, summer 2013.

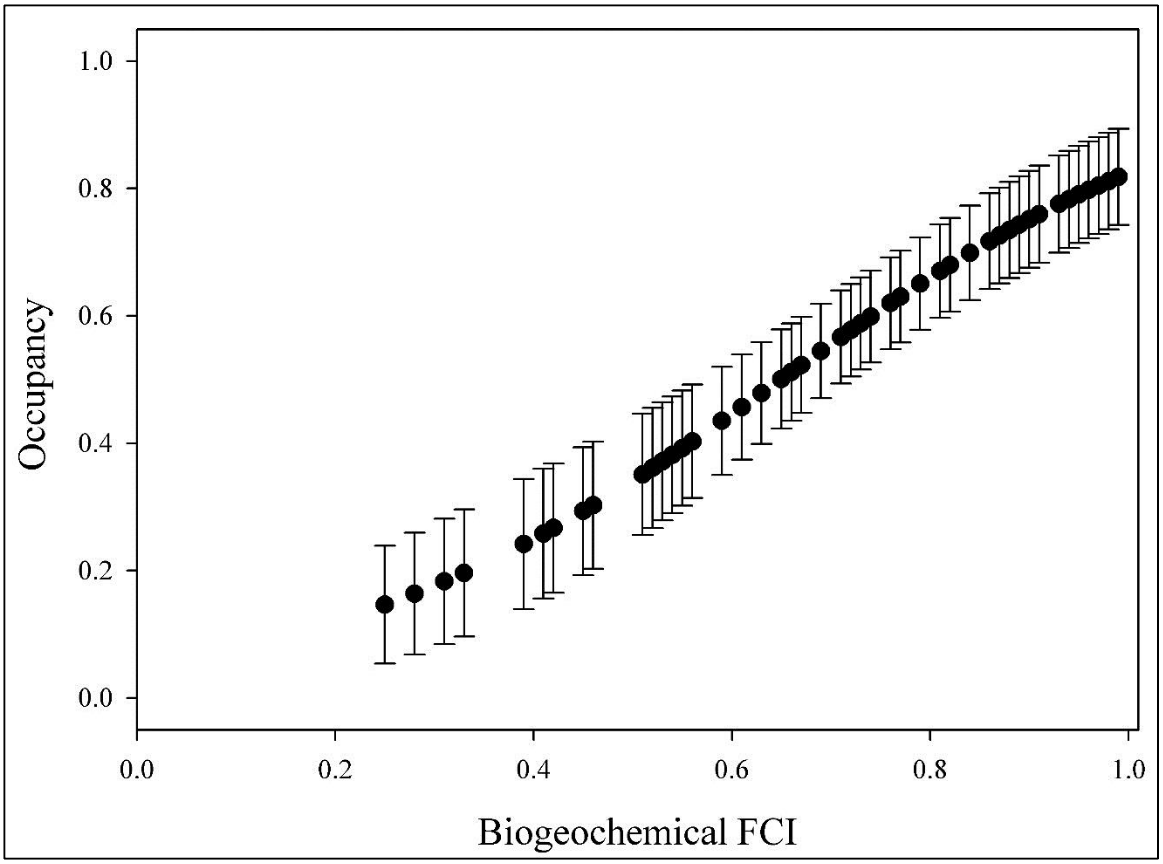

Figure C.5 Individual site estimates of occupancy (Ψ) and standard errors of the Biogeochemical Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Desmognathus ochrophaeus*, southwest Virginia, summer 2013.

Table C.3 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) occupancy models for *Desmognathus ochrophaeus*, southwest Virginia, summer 2013.

FCI Occupancy Models and Covariates for *D. ochrophaeus*	**Beta**	**Standard Error**
Ψ(HabitatFCI), p(SoilT*Rain)		
Ψ	-2.8471	1.0227
Ψ.HabitatFCI	4.4261	1.3975
p	0.7618	0.2601
p.SoilT	-0.5866	0.2563
p.Rain	-1.9545	0.7407
Ψ(BiogeochemicalFCI), p(SoilT*Rain)		
Ψ	-2.8649	1.0834
Ψ.BiogeochemicalFCI	4.4126	1.4660
p	0.7278	0.2627
p.SoilT	-0.6352	0.2615
p.Rain	-2.0163	0.7328
Ψ(HydrologyFCI), p(SoilT*Rain)		
Ψ	-1.6253	1.0904
Ψ.HydrologyFCI	2.7780	1.5228
p	0.6841	0.2677
p.SoilT	-0.6754	0.2649
p.Rain	-2.0141	0.7312
Ψ(Null), p(SoilT*Rain)		
Ψ	0.3339	0.2834
p	0.6581	0.2697
p.SoilT	-0.7113	0.2683
p.Rain	-2.0138	0.7309

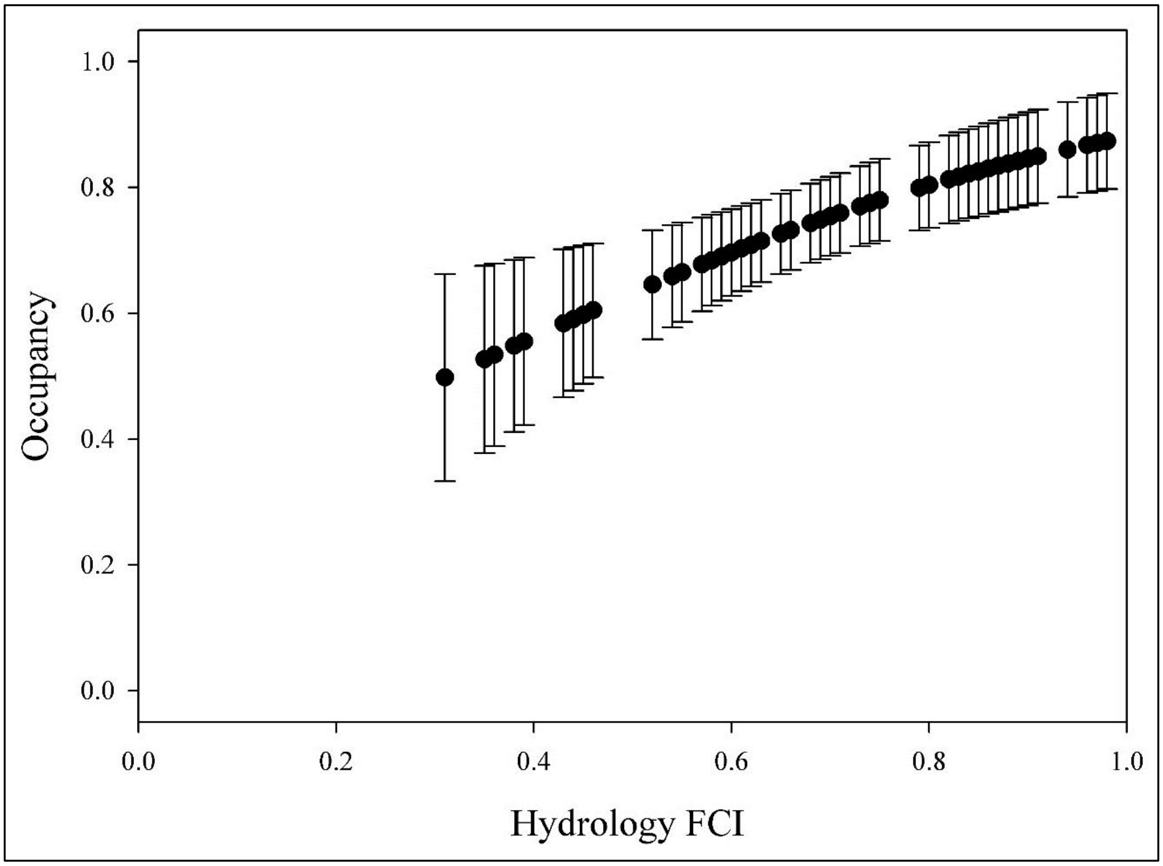

Figure C.6 Individual site estimates of occupancy (Ψ) and standard errors of the Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Eurycea* spp., southwest Virginia, summer 2013.

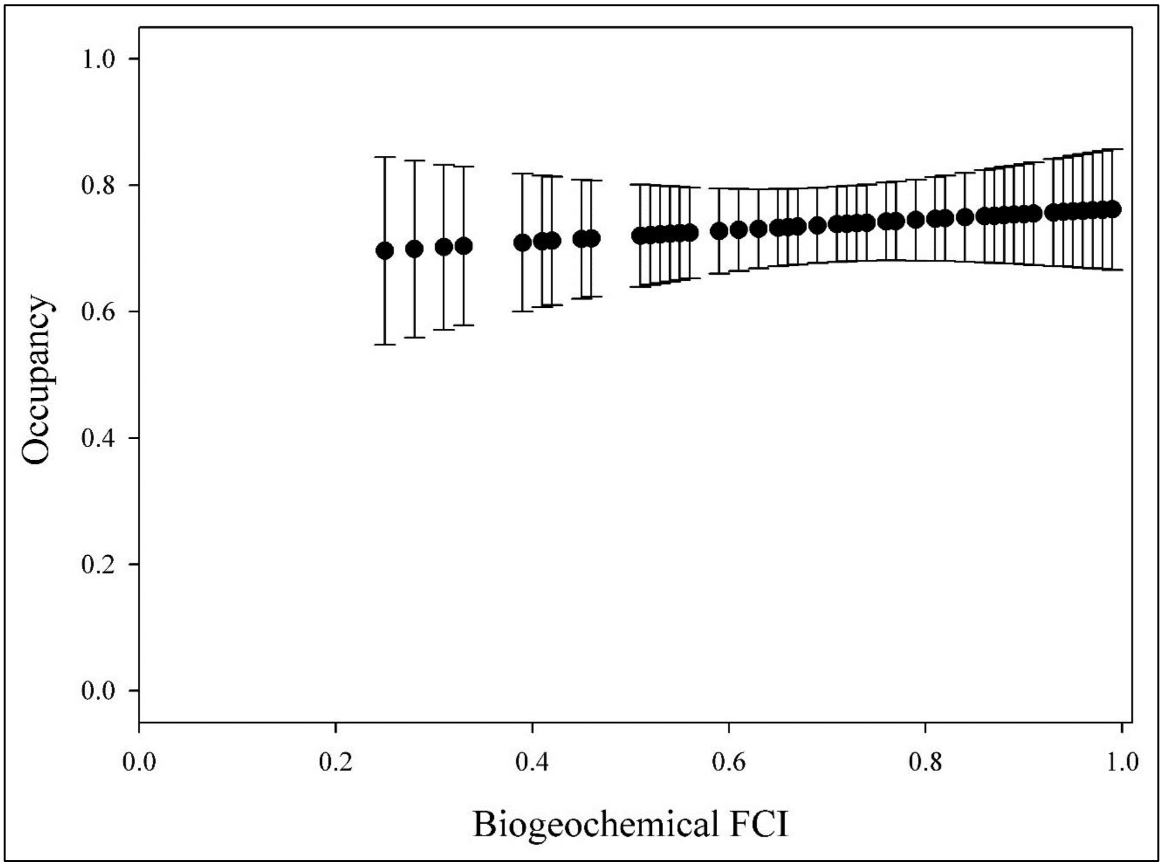

Figure C.7 Individual site estimates of occupancy (Ψ) and standard errors of the Biogeochemical Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Eurycea* spp., southwest Virginia, summer 2013.

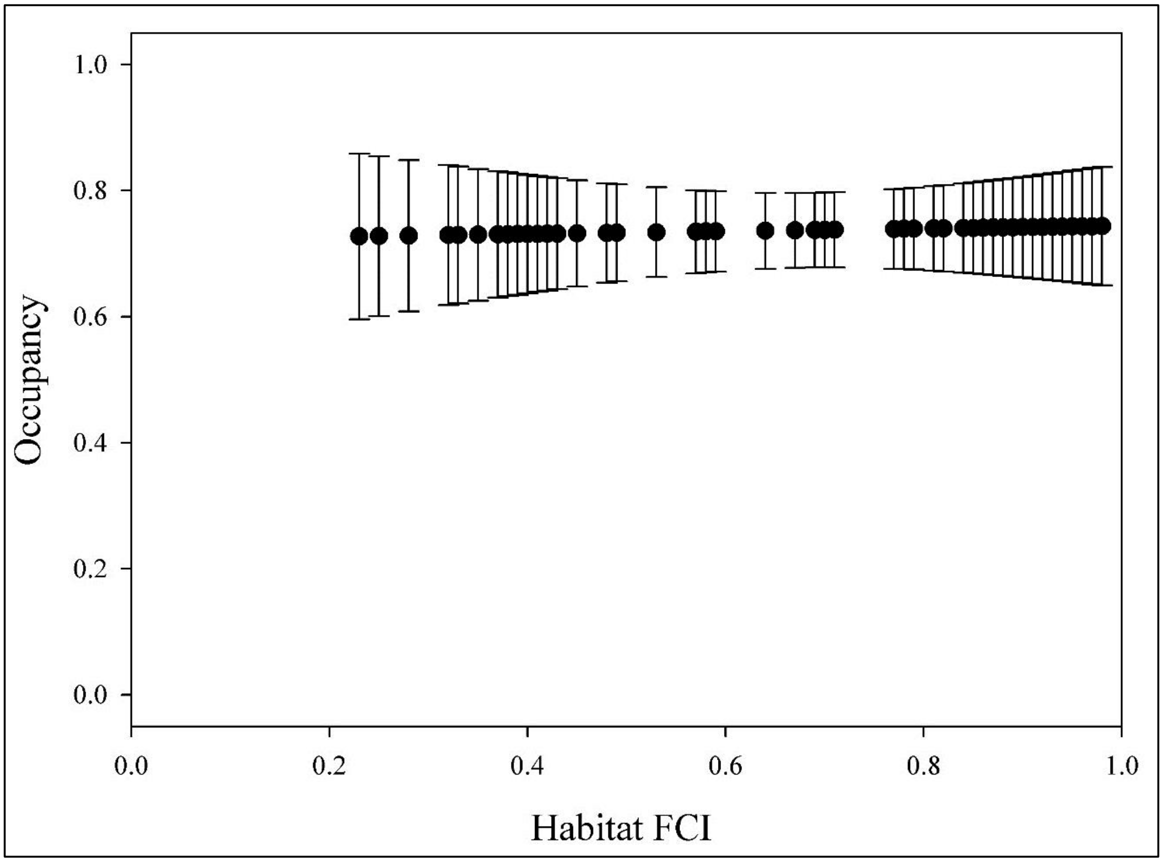

Figure C.8 Individual site estimates of occupancy (Ψ) and standard errors of the Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores for *Eurycea* spp., southwest Virginia, summer 2013.

Table C.4 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) occupancy models for *Eurycea* spp., southwest Virginia, summer 2013.

FCI Occupancy Models and Covariates for *Eurycea* spp.	**Beta**	**Standard Error**
Ψ(HyrdologyFCI), p(SoilT*Rain)		
Ψ	-0.9079	1.1675
Ψ.HydrologyFCI	2.8970	1.7669
p	0.5972	0.2148
p.SoilT	0.3508	0.1974
p.Rain	1.0412	0.6654
Ψ(Null), p(SoilT*Rain)		
Ψ	1.0310	0.3054
p	0.6114	0.2149
p.SoilT	0.3239	0.1964
p.Rain	1.0260	0.6649
Ψ(BiogeochemicalFCI), p(SoilT*Rain)		
Ψ	0.7167	1.0353
Ψ.BiogeochemicalFCI	0.4496	1.4295
p	0.6084	0.2152
p.SoilT	0.3295	0.1974
p.Rain	1.0280	0.6650
Ψ(HabitatFCI), p(SoilT*Rain)		
Ψ	0.9548	0.9382
Ψ.HabitatFCI	0.1112	1.2998
p	0.6106	0.2151
p.SoilT	0.3252	0.1969
p.Rain	1.0265	0.6650

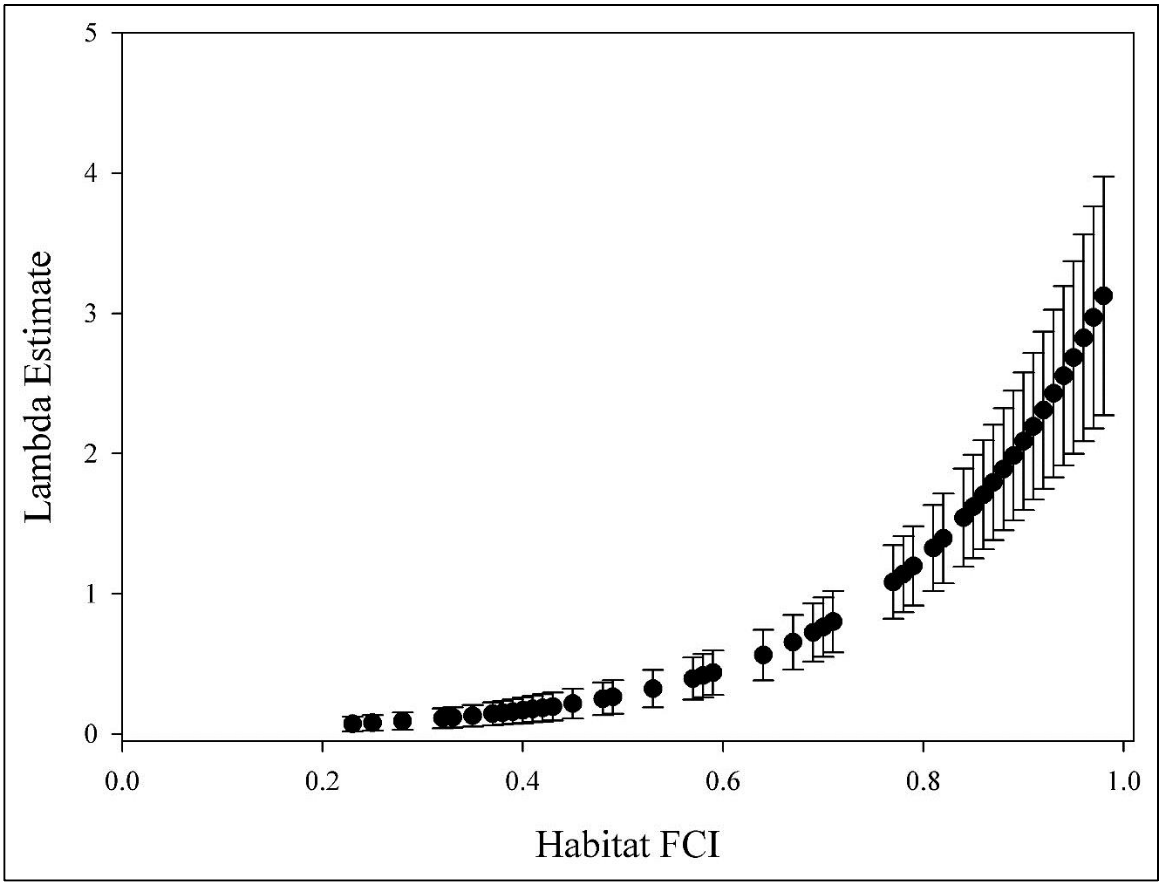

Figure C.9 Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimates of abundance (λ) per 25 m x 5m quadrat along with standard errors for *Desmognathus fuscus*, southwest Virginia, summer 2013.

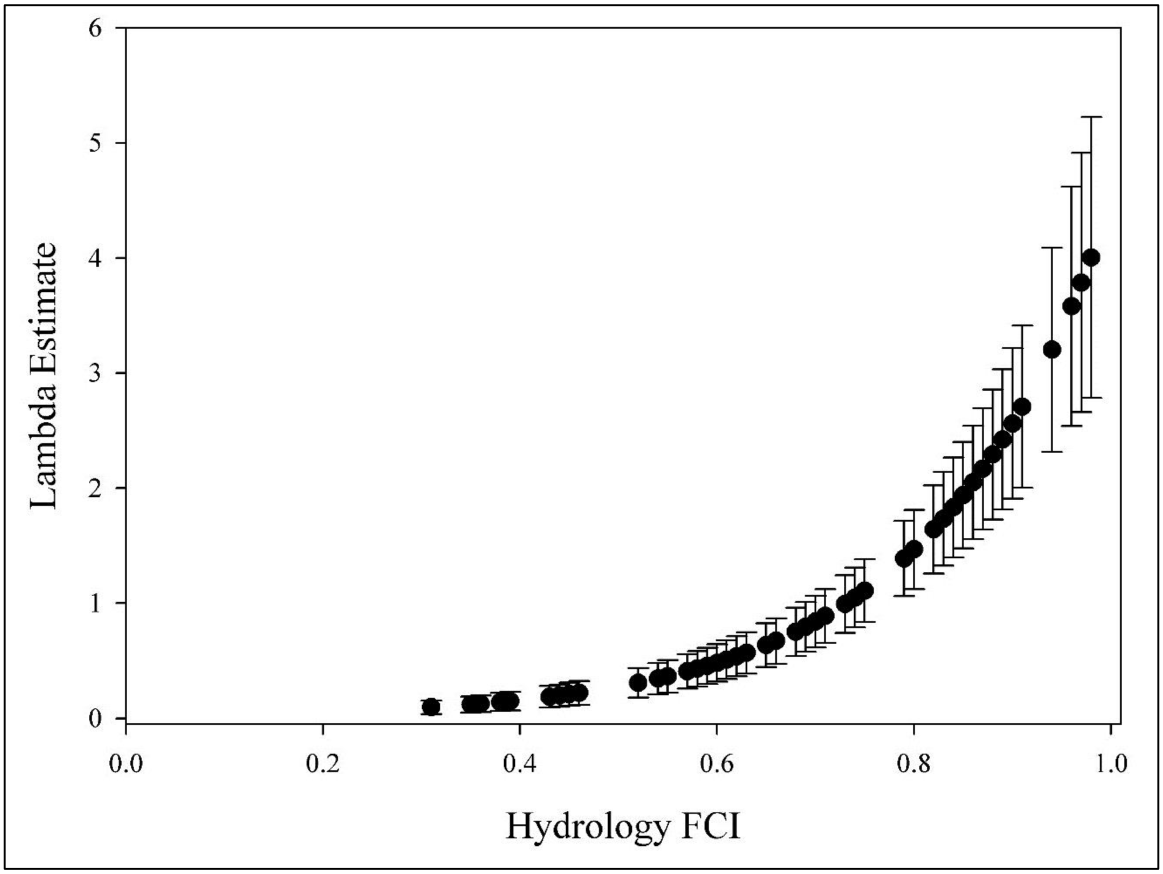

Figure C.10 Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimates of abundance (λ) per 25 m x 5m quadrat along with standard errors for *Desmognathus fuscus*, southwest Virginia, summer 2013.

Table C.5 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) abundance models for *Desmognathus fuscus*, southwest Virginia, summer 2013.

FCI Abundance Models and Covariates for *D. fuscus*	**Beta**	**Standard Error**
λ(HabitatFCI), p(SoilT*FlowAbove)		
λ	-3.8005	0.9609
λ.HabitatFCI	5.0402	1.1057
p	-0.6755	0.3400
p.SoilT	-0.3574	0.1826
p.FlowAbove	-1.5359	0.5850
λ(HydrologyFCI), p(SoilT*FlowAbove)		
λ	-4.0799	0.9739
λ.HydrologyFCI	5.5785	1.1638
p	-0.7223	0.3422
p.SoilT	-0.3947	0.1819
p.FlowAbove	-1.6377	0.5786
λ(BiogeochemicalFCI), p(SoilT*FlowAbove)		
λ	-2.4927	0.7757
λ.BiogeochemicalFCI	3.4101	0.9166
p	-0.6170	0.3237
p.SoilT	-0.3951	0.1880
p.FlowAbove	-1.5548	0.5872
λ(Null), p(SoilT*FlowAbove)		
λ	0.1046	0.1940
p	-0.5397	0.2951
p.SoilT	-0.5230	0.1872
p.FlowAbove	-1.5270	0.5937

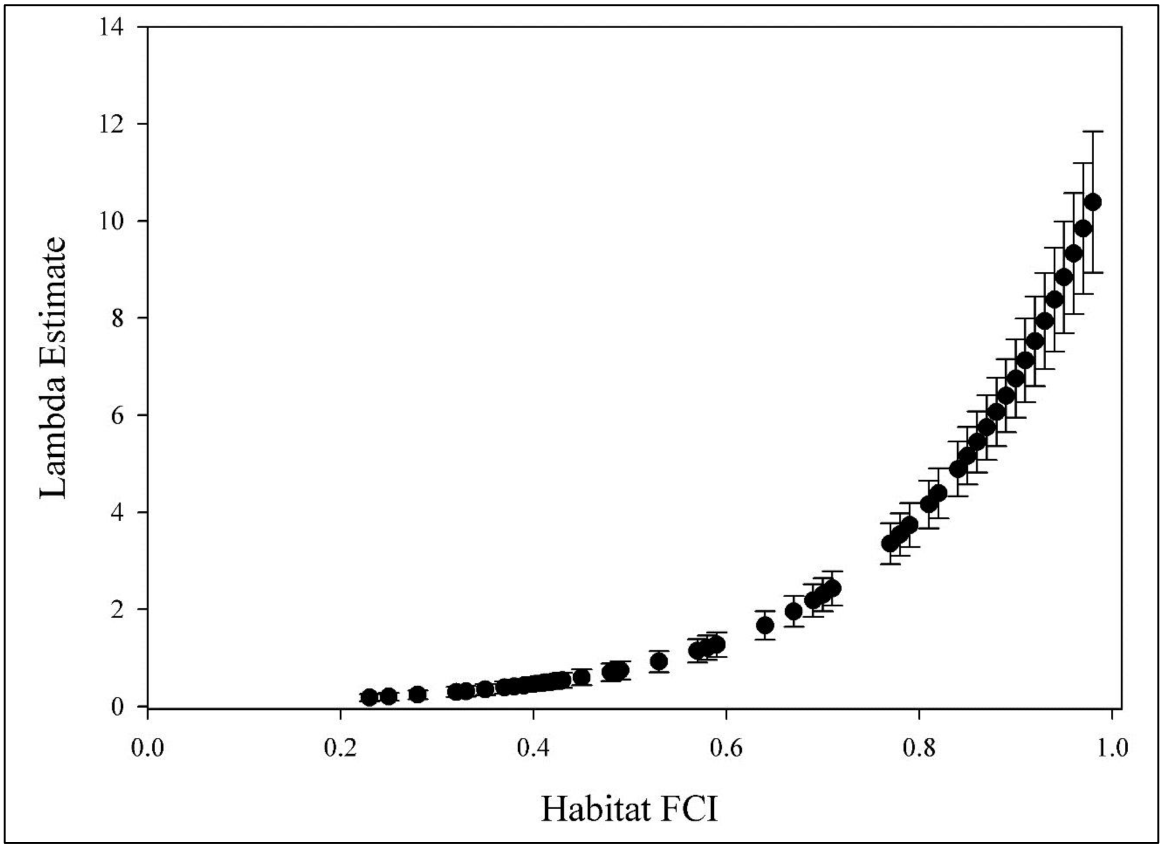

Figure C.11 Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimates of abundance (λ) per 25 m x 5m quadrat along with standard errors for *Desmognathus monticola*, southwest Virginia, summer 2013.

Table C.6 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) abundance models for *Desmognathus monticola*, southwest Virginia, summer 2013.

FCI Abundance Models and Covariates for *D. monticola*	**Beta**	**Standard Error**
λ(HabitatFCI), p(SoilT*FlowBelow)		
λ	-2.9324	0.5470
λ.HabitatFCI	5.3807	0.6274
p	-0.7992	0.1677
p.SoilT	-0.4181	0.1143
p.FlowBelow	1.6380	0.4228
λ(BiogeochemicalFCI), p(SoilT*FlowBelow)		
λ	-2.5325	0.5006
λ.BiogeochemicalFCI	4.8490	0.5742
p	-0.7642	0.1621
p.SoilT	-0.4281	0.1176
p.FlowBelow	1.7377	0.4294
λ(HydrologyFCI), p(SoilT*FlowBelow)		
λ	-1.3056	0.4256
λ.HydrologyFCI	3.4408	0.5241
p	-0.6644	0.1502
p.SoilT	-0.5026	0.1165
p.FlowBelow	1.6184	0.4223
λ(Null), p(SoilT*FlowBelow)		
λ	1.2477	0.0961
p	-0.6171	0.1436
p.SoilT	-0.6096	0.1159
p.FlowBelow	1.6984	0.4349

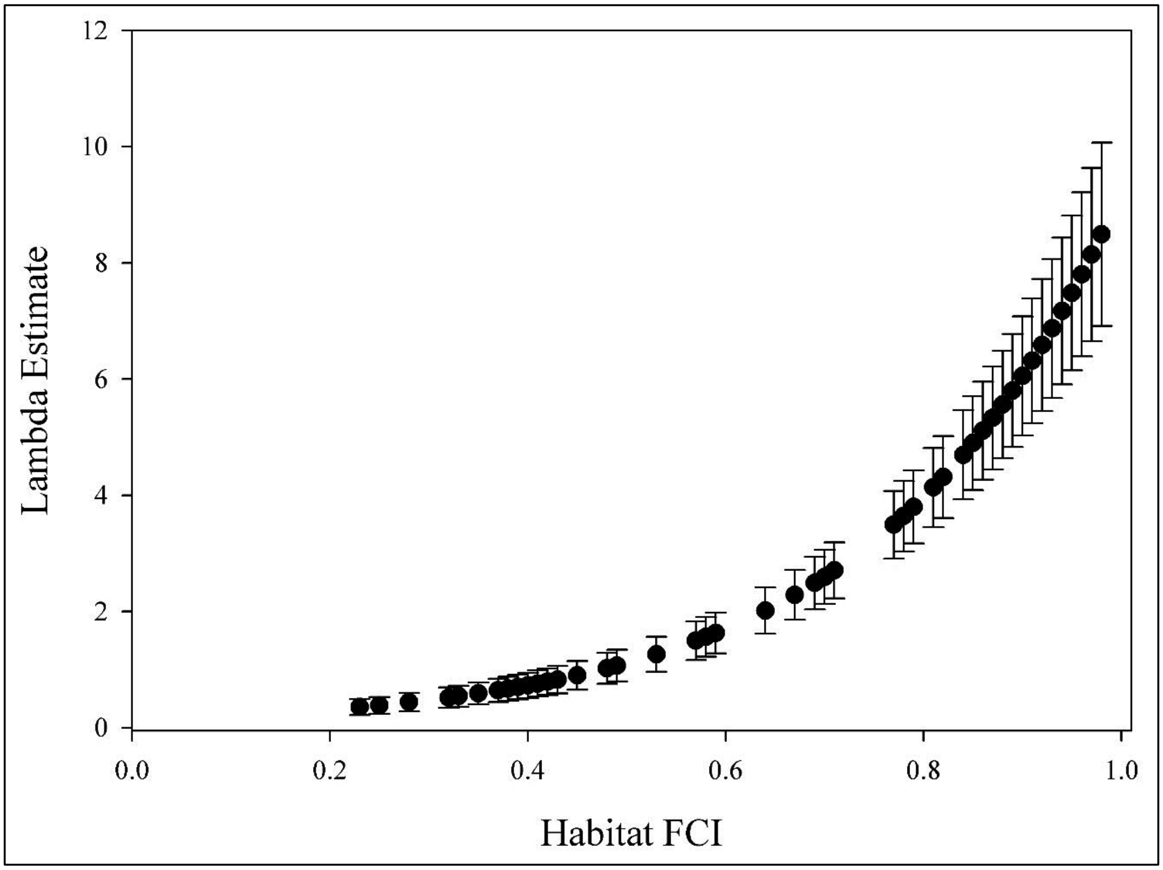

Figure C.12 Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimates of abundance (λ) per 25 m x 5m quadrat along with standard errors for *Desmognathus ochrophaeus*, southwest Virginia, summer 2013.

Table C.7 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) abundance models for *Desmognathus ochrophaeus*, southwest Virginia, summer 2013.

FCI Abundance Models and Covariates for *D. ochrophaeus*	**Beta**	**Standard Error**
λ(HabitatFCI), p(AirT*Rain)		
λ	-2.0075	0.5082
λ.HabitatFCI	4.2315	0.5802
p	-0.8477	0.2368
p.AirT	-0.3570	0.1258
p.Rain	-1.1594	0.3738
λ(BiogeochemicalFCI), p(AirT*Rain)		
λ	-1.7804	0.4919
λ.BiogeochemicalFCI	3.8607	0.5647
p	-0.7564	0.2192
p.AirT	-0.3703	0.1282
p.Rain	-1.2670	0.3741
λ(HydrologyFCI), p(AirT*Rain)		
λ	-0.0929	0.3963
λ.HydrologyFCI	1.7426	0.5000
p	-0.6313	0.1952
p.AirT	-0.4919	0.1274
p.Rain	-1.4358	0.3754
λ(Null), p(AirT*Rain)		
λ	1.1552	0.1250
p	-0.5952	0.1873
p.AirT	-0.5431	0.1276
p.Rain	-1.4975	0.3763

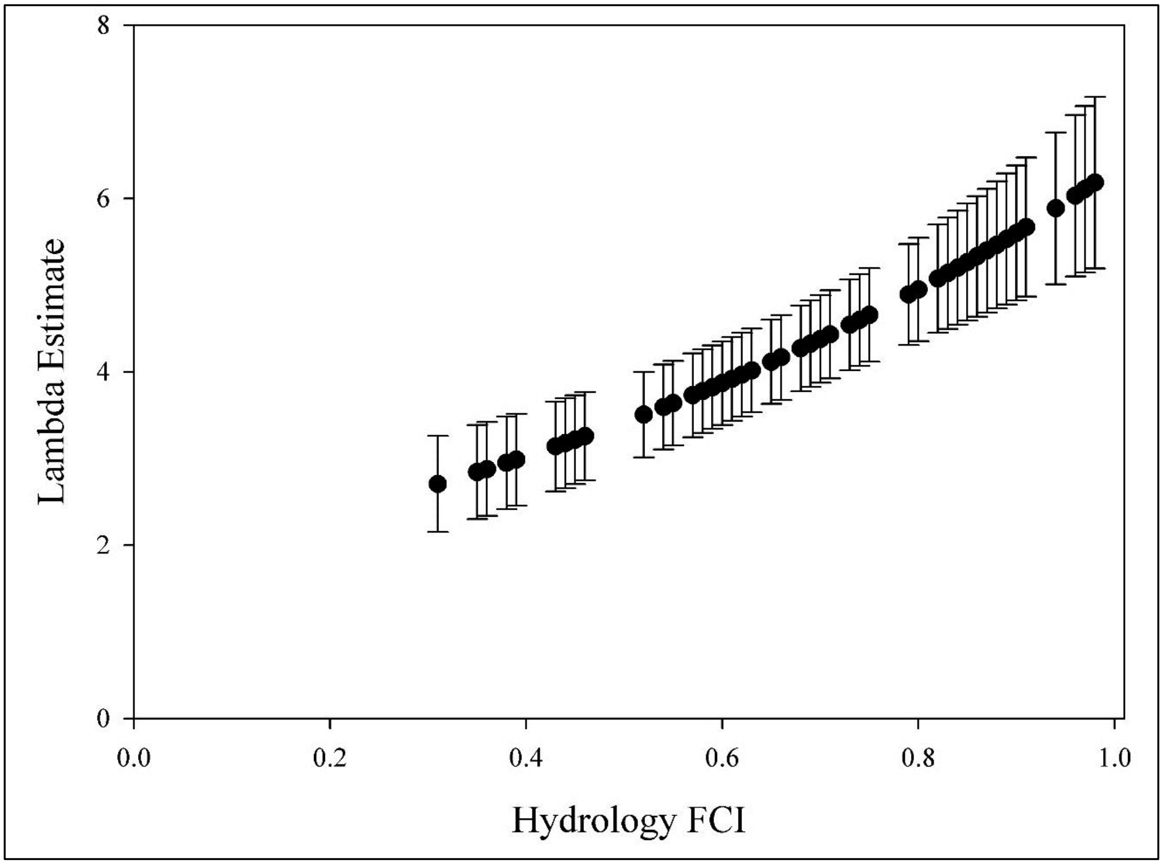

Figure C.12 Hydrology Functional Capacity Index (FCI; see Noble et al. 2010) scores and individual site estimates of abundance (λ) per 25 m x 5m quadrat along with standard errors for *Eurycea* spp., southwest Virginia, summer 2013.

Table C.8 Beta estimates and standard errors for all covariates in the Functional Capacity Index (FCI; see Noble et al. 2010) abundance models for *Eurycea* spp., southwest Virginia, summer 2013.

FCI Abundance Models and Covariates for *Eurycea* spp.	**Beta**	**Standard Error**
λ(HydrologyFCI), p(SoilT*Rain)		
λ	0.6131	0.3206
λ.HydrologyFCI	1.2330	0.4193
p	-0.7359	0.1668
p.SoilT	0.3820	0.0858
p.Rain	0.7597	0.2997
λ(BiogeochemicalFCI), p(SoilT*Rain)		
λ	1.9538	0.2554
λ.BiogeochemicalFCI	-0.7134	0.3439
p	-0.6764	0.1631
p.SoilT	0.3189	0.0865
p.Rain	0.8093	0.3084
λ(Null), p(SoilT*Rain)		
λ	1.4642	0.1113
p	-0.6891	0.1629
p.SoilT	0.3482	0.0856
p.Rain	0.7854	0.3057
λ(HabitatFCI), p(SoilT*Rain)		
λ	1.4882	0.2415
λ.HabitatFCI	-0.0359	0.3219
p	-0.6874	0.1635
p.SoilT	0.3465	0.0869
p.Rain	0.7851	0.3060

Chapter 5

Conclusions and Management Recommendations

Overall, there was little difference in results from the occupancy analysis (Sweeten 2015a) as compared to the results of the abundance analysis (Sweeten 2015b) for aquatic salamanders in the southwest Virginia coalfields. The top four models for *Desmognathus fuscus* were variable in ranking order between the occupancy and abundance analyses, however, each remained in the top four for both analyses. Abundance results ranked the **Shading Model** (containing canopy cover, herbaceous cover, sapling/shrub density, and tree DBH as covariates) and the **Trees Model** (containing canopy cover, tree DBH, and riparian tree species richness covariates) as the only models with empirical support for *Desmognathus monticola*. While these two models were ranked sixth and seventh in the occupancy analysis, they still had empirical support (Δ AIC = 3.98 and 2.83 respectively). The **Shading Model** was the only abundance model with empirical support for *Desmognathus ochrophaeus*. In the occupancy analysis this model was ranked second with a Δ AIC of 2.01. Additionally, occupancy and abundance model averages for the three *Desmognathus* species showed few differences, confirming the similarity of the two statistical approaches.

However, *Eurycea* spp. did have significantly different results between the occupancy and abundance analyses. The **Sediment Model** (consisting of turbidity, stream embeddedness, and stream bank erosion covariates) being the only abundance model with empirical support whereas the **HUC12-Recent Mining Model** (percent recent mining in the HUC-12 watershed) and the **HUC12-Forested Model** (percent forested in the HUC-12 watershed) were top occupancy models. Although the **Sediment Model** is a microhabitat model, both turbidity and stream substrate embeddedness are often a result of erosion from upstream watershed land uses such as agriculture, urbanization, and surface coal mining (Orser and Shure 1972: Price et al. 2011). Surface coal mining, at least in the short-term, creates vast areas of bare, relatively impervious surfaces that leads to increased sedimentation of streams (Minear and Tschantz 1976; Mitsch et al. 1983). This may explain differences in results between the abundance estimates and the occupancy analysis where the **HUC-12 Recent Mining Model** had the most empirical support (Sweeten 2015a). Past research has documented decreases in *Eurycea* abundances with increased impervious surface from urbanization similar to the results with the occupancy **HUC-12 Recent Mining Model** (Sweeten 2015b) as none of the other potential mining-related parameters, such as the **Chemistry Model** (consisting of conductivity and water temperature), the **Valley Fill Model** (consisting of the area of valley fill in the subwatershed), the **Recent Mining Model** (percent recent mining in the subwatershed), and the **Surface Mining Model** (consisting of percent recent mining, area of valley fills and number of ponds in the subwatershed as well as the number of ponds downstream), had empirical support in either the occupancy or abundance analyses (Miller et al. 2007; Minton 1968; Orser & Shure 1972; Willson and Dorcas 2003).

Occupancy and abundance analyses of the FCI habitat assessment protocol (Sweeten 2015c) found that overall the Habitat FCI score was strongly correlated to *Desmognathus* spp. For example, the **Habitat FCI model** was the top model for *D. fuscus* and *D. ochrophaeus* in both the occupancy and abundance analyses. However, results between the occupancy and abundance analyses were slightly different in some cases. Occupancy results showed the **Biogeochemical FCI model** was the only model with empirical support for *D. monticola*, whereas, the **Habitat FCI Model** was the only model with empirical support in the abundance analysis. In the occupancy analysis for *Eurycea* spp. all three FCI models (Hydrology, Biogeochemical and

Habitat) had empirical support. However, in the abundance analysis, the **Hydrology FCI Model** was the only model with empirical support. The high densities of *Eurycea* may explain differences between the occupancy and abundance analyses. When there are low densities of salamanders there is little difference between the binomial of the presence/absence and the count data for abundance. However, when abundances are high, that difference between the presence/absence data and the count data becomes much larger and more influential on the results.

My results indicate that occupancy sampling and analysis techniques may provide a quicker and cheaper alternative to traditional abundance sampling, particularly if the goal of the study is to examine general trends and relationships across a large number of sites (i.e., stream and riparian habitat condition assessment using the Habitat FCI). Occupancy sampling designs would allow field biologists to sample a site only long enough to find one individual of each focal species. That, in turn, would allow for many more sites to be sampled. However, it should be noted that if the scope of the project is narrowly focused, then an occupancy sampling design may not be optimal.

My results also show the importance of riparian habitat quality, particularly for *Desmognathus* salamanders. Mature forest-like conditions such as canopy cover, native tree species richness, and detritus cover, were found to greatly influence the presence/absence of *Desmognathus* species as well as abundance. Currently, most mine land reclamation in central Appalachia has two terminal land uses: reforestation or grass-shrubland. Reforestation techniques such as those recommended by Appalachian Regional Reforestation Initiative (ARRI) are becoming increasingly common since the early 2000s (personal communication, L. Tankersley, DMLR Ecologist). The main goal of ARRI is to plant high-value hardwood trees on reclaimed mine land in order to produce commercially valuable crop trees along with quick-growing early successional trees for soil stability and wildlife value.

Currently, the ARRI protocol does not have different management plans for riparian areas and upland areas. Additionally, current regulatory reclamation standards for grassland/shrubland land uses do not require reforestation of riparian areas. It is my recommendation that specific guidelines be developed for the reclamation of riparian areas regardless of target land use. These riparian guidelines should include the planting of quick-growing early successional tree species such as black birch (*Betula lenta*), yellow poplar (*Liriodendron tulipifera*), black willow (*Salix nigra*), American sycamore (*Platanus occidentalis*), eastern cottonwood (*Populus deltoids*) and white pine (*Pinus strobus*) in order to quickly establish a canopy cover, stabilize the soil, and impede invasive plant species (See Davis et al. 2012 for more extensive list of suitable tree species adapted for moist/wet sites). Moreover riparian tree species should be selected based on wildlife value instead of crop tree value. Timber harvest within a riparian zone has been shown to cause changes in benthic macroinvertebrate, stream salamander, and fish densities and community structure (Rios and Bailey 2006; Jones et al. 2011; Maigret et al. 2014). Additionally, in the Appalachians riparian and upland tree species compositions naturally differ. For example, black birch (*Betula lenta*), yellow poplar (*Liriodendron tulipifera*), rhododendron (*Rhododendron maximum*), and eastern hemlock (*Tsuga canadensis*) are more associated with riparian areas and near streams, whereas, oaks (*Quercus* spp.) and red maple (*Acer rubrum*) are more suited to medium to xeric upland sites (Clinton et al. 2010; Hedman and van Lear 1995). In order to return mine reclamation land to the closest approximation of a native Appalachian forest, it is imperative to have differential reforestation management plans for upland and riparian areas.

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