VALIDATION OF A STREAM AND RIPARIAN HABITAT ASSESSMENT PROTOCOL USING STREAM SALAMANDERS IN THE SOUTHWEST VIRGINIA COALFIELDS[1]

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Abstract: Within the central Appalachia Coalfields, the aquatic impacts of large-scale land uses, such as surface mining, are of particular ecological concern. Identification and quantification of land use impacts to aquatic ecosystems are a necessary first step to aid in mitigation of negative consequences to biota. However, quantifying physical environmental quality such as stream and riparian habitat often can be quite difficult, particularly when there is time or fiscal limitations. As such, standard protocols such as the U.S. EPA's Stream Habitat Rapid Bioassessment Protocol have been established to be cost- and time-effective. This protocol estimates ten different stream and riparian conditions on a scale of 0 to 20. Unfortunately, using estimations can be problematic because of 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 these biases 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 variables along with watershed land use to calculate three different scores, a hydrology score, biogeochemical score, and habitat score. In our study, we examined the correlation of stream salamander presence and abundance to the three USACE scores. In the summer of 2013, we visited 70 sites in the southwest Virginia Coalfields multiple times to collect salamanders and quantify stream and riparian microhabitat parameters. Using occupancy and abundance analyses, we found strong relationships among three *Desmognathus spp.* and the USACE Habitat FCI score. Accordingly, the Habitat FCI score provides a reasonable assessment of physical instream and riparian conditions that may serve as a surrogate for understanding the community composition and integrity of aquatic salamander in the region.

Additional Key Words: Central Appalachia; headwater streams; rapid assessment; coal mining

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Introduction

Much is still unknown about many of the terrestrial and aquatic ecosystem dynamics and interactions that are changed by dramatic landscape alterations from coal mining practices (Stout and Wallace, 2005; Simmons et al., 2008). High quality physical stream and riparian habitat 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 facilitate subsequent aquatic biota recovery. However, accurately assessing and measuring physical stream and riparian habitat can be difficult because of fiscal and personnel constraints. Consequently regulators and mangers are accordingly challenged by the 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 characteristics 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 could be biased because of inexperienced or improperly trained personnel. A more quantitative method that takes into account surrounding riparian quality may provide more accurate, reliable results.

In 2010 the U.S. Army Corps of Engineers 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 (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 watershed land use. 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 variables: 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 watershed land use. 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 watershed land use are the variables that comprise the Biogeochemical FCI. 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 immediately adjacent to the Virginia coalfields, the protocol does not contain variables specific only to the western West Virginia and eastern Kentucky region. The region defined by the FCI protocol is 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 (Ecoregion 69). 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, our previous occupancy and abundance analyses of stream salamanders had strong relationships with riparian and instream habitat variables that are covariates that are also used in the FCI scores including canopy cover, stream substrate embeddedness, and stream bank erosion (Sweeten, 2015). We therefore decided to conduct two *post hoc* analyses (one using occupancy, one using abundance) to determine if there were any relationships among the three FCI scores and stream salamander metrics.

Methods

Regional Description

Our study area was located in Wise, Russell, and Dickenson counties in the southwest Virginia coalfields. This area is part of the Cumberland Plateau (Region 69d), a sub-region of the Central Appalachian Mountains (Omernik, 1987). Topography is characterized by steep mountains with narrow valleys with an average peak elevation of 760 m (Woodward and Hoffman, 1991). Most soils in this region are Utisols, Alfisols, and Inceptisols (McNab and Avers, 1995). Average annual precipitation is about 1150 mm with an average temperature of 13°C (Woodward and Hoffman, 1991; McNab and Avers, 1995). Regionally, it is estimated that 93 % of the Cumberland Plateau is forested, 4 % of the region is agricultural/open area, 2 % is barren, and 1 % is developed. Much of the open or barren classification is a result of past or current surface mining (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*) (McNab and Avers, 1995). Because of the steep topography, this region tends to have a high density of small- to medium-sized streams (McNab and Avers, 1995). Streams in this region are characterized by boulder/cobble substrate, moderate to high gradient, and low conductivity. The Cumberland Plateau has high levels of aquatic biodiversity and species richness, with many endemic species (Morse et al., 1993).

Site Selection

We selected five 12 digit Hydrologic Unit Codes (HUC-12) watersheds in southwest Virginia as study sites (Table 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. We divided streams within these watersheds into segments by their stream order, and gave each stream segment in each watershed a unique identification number. We selected 70 first- or second-order stream segments using best professional judgment for sites that had both allowed landowner access and that we considered safe to sample. Because of a lack of accessible sites, we sampled 10 sites in the Pigeon Creek watershed and 10 sites in the Dumps Creek watershed. Roaring Fork and Rocky Fork watersheds each had 15 salamander sampling locations, and 20 sites were sampled in the Callahan Creek watershed.

To accurately estimate detection rates, we visited 67 of the 70 locations three times each in 2013 (Bailey et al., 2004; MacKenzie and Royle, 2005). Because of access issues, we were only able to sample three of the sites twice in 2013. At each sampling location a 25 m long by 5 m wide quadrat was placed parallel to the stream channel with the stream center as the right or left edge of the quadrat (Hairston, 1986; Jung et al., 2000). Right or left quadrat placement was determined randomly using a coin flip. We hand-captured adult salamanders (all by overturning all rocks, detritus, and logs within the 25 m x 5 m quadrat at each sampling site). We identified all adult salamanders to species in the field and immediately released them to within 2 m of each capture location. All transformed salamanders (sexually mature and immature) were considered to be adults. 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. Because of the difficulty of identifying larval salamanders to the species level, we identified larval salamanders to genus. Salamander surveys were approved by Virginia Tech Institutional Animal Use and Care Committee protocol 13-053-FIW.

Table 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 (square kilometers) 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

FCI Assessment Protocol

A full description of the original field protocols for the FCI protocol was defined by Noble et al. (2010). Although we largely followed the FCI protocol, some modifications were used in this study. First, because of site conditions habitat measurements were recorded from a 25 m x 10 m quadrat centered on the salamander quadrat rather than the 30.5 m x 15.2 m quadrat as suggested by the FCI protocol. Secondly, we took three canopy cover measurements, six detritus measurements, and six herbaceous cover measurements, rather than the 10 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. 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 equation used for calculating the Hydrology FCI is:

$$FCI = \left\{ \frac{V_{WLUSE} + \frac{[V_{LWD} + min(V_{SUBSTRATE}, V_{EMBED}, V_{BERO})]}{2}}{2} \right\} \quad (1)$$

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; www.esri.com).

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

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

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.

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

$$FCI = \langle \min(V_{EMBED,} V_{SUBSTRATE}) X \frac{\left\{\left(\frac{V_{LWD}+V_{DETRITUS}}{2}\right)+\left[\frac{\left(\frac{V_{SNAG}+V_{SSD}+V_{HERB}+V_{SRICH}}{6}\right)+V_{WLUSE}}{4}\right]\right\}}{2} \rangle \qquad (3)$$

where V_{EMBED} = the mean embeddedness of the stream channel (n = 30); $V_{SUBSTRATE}$ = the median stream substrate particle size (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_{SNAG} = the number of standing, dead snags per 25 m of stream; 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); V_{SRICH} = riparian vegetation speci**e**s richness score per 25 m of stream reach; and V_{WLUSE} was calculated using land cover data from Maxwell et al. (2014), in ArcMap.

Biogeochemical function is defined as "the ability of a 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). One of two equations was used to calculate the Biogeochemical FCI score. If there was a mean channel canopy cover of ≥ 20 percent, then the equation is:

$$FCI = \left\{ V_{EMBED}\ X \sqrt{\left[\frac{\left(\frac{V_{LWD}+V_{DETRITUS}+V_{TBDH}}{3}\right)+V_{WLUSE}}{2}\right]} \right\} \qquad (4)$$

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_{TDBH} = the mean diameter at breast height (DBH) of trees with DBH ≥ 10 cm; and V_{WLUSE} was calculated using land cover data from Maxwell et al. (2014), in ArcMap.

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

$$FCI = \left\{ V_{EMBED}\ X \sqrt{\left[\frac{\left(\frac{V_{LWD}+V_{DETRITUS}+V_{SSD}+V_{HERB}}{4}\right)+V_{WLUSE}}{4}\right]} \right\} \qquad (5)$$

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.

Occupancy and Abundance Modeling

To assess salamander occupancy and detection probabilities and abundance estimates, we used the Program PRESENCE software (USGS, Laurel, MD; www.mbr-pwrc.usgs.gov/software). Program PRESENCE was developed to examine and rank multiple hypotheses using an information-theoretic approach (AIC) and maximum likelihood to determine the best-fit model for the data (Bailey et al., 2007; Kroll et al., 2010). Within Program PRESENCE, we 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, 2004).

Additionally, we estimated detection probabilities for each species to determine which environmental conditions most influenced detection. Detection is important in AIC analyses in order to produce the most reliable estimates of 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). We used a two-step method to determine which detection covariates to include for each species group in the occupancy analysis (Burnham and Anderson, 2002). We ran seven *a priori* detection covariates for each species group 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, we normalized continuous detection covariates as well as FCI scores in order to compare beta (effect size) values among models.

Results

We captured 1,145 stream salamanders consisting of nine species during the 207 surveys over the 2013 collection. Because of the large number of *Eurycea* spp. larval salamanders and the small number of adult *Eurycea*, we combined larval and adult *Eurycea longicauda* (Long-tailed Salamander) and *Eurycea cirrigera* (Southern Two-lined Salamander) to the genus-level for the *Eurycea* spp. group. Based on results from Sweeten (2015), the four salamander groups that had sufficient data for occupancy analysis were also used for analysis of abundance: *Desmognathus* *fuscus* (Northern Dusky Salamander), *Desmognathus monticola* (Seal Salamander), *Desmognathus ochrophaeus* (Mountain Dusky Salamander), and *Eurycea* spp.

Occupancy

The Habitat FCI Model was the best occupancy model for *Desmognathus fuscus* with an AIC weight of 0.85 and strong empirical support with a Δ ACI of 0.00 (Table 2). 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 3). Individual site occupancy probabilities for *D. fuscus* ranged from 0.01 to 0.89 (Fig. 1A). The Hydrology FCI Model also had moderate empirical support with a Δ AIC of 3.58 and an AIC weight of 0.14 (Table 2; Fig. 1C). A model is considered to have strong empirical support if it has a Δ AIC < 2.0. Whereas a model with a Δ AIC of 2.0-4.0 is considered to have moderate empirical support.

The only occupancy model with empirical support for *Desmognathus monticola* was the Biogeochemical FCI Model with an AIC weight of 0.97 (Table 2). *D. monticola* showed a strong positive correlation to the Biogeochemical FCI score with a beta estimate of 10.89 (SE = 2.95; Table 2). Individual site occupancy probabilities ranged from 0.04 to 0.94 (Fig. 1B).

The best occupancy model for *Desmognathus ochrophaeus* was the Habitat FCI Model which had an AIC weight of 0.63 (Table 2). Occupancy of *D. ochrophaeus* was positively correlated to the Habitat FCI scores (Beta estimate = 4.43; SE = 1.40; Table 3), and individual site occupancy estimates ranged from 0.14 to 0.82 (Fig. 1A). There was also empirical support for the Biogeochemical FCI Model with a Δ AIC of 1.11 and an AIC weight of 0.36 (Table 2; Fig. 1B).

The best occupancy model for *Eurycea* spp. was the Hydrology FCI Model with an AIC weight of 0.47 (Table 2). The beta estimate of 2.90 (SE = 1.77) shows a positive correlation between *Eurycea* spp. occupancy and Hydrology FCI scores (Table 3). Individual site occupancy estimations ranged from 0.50 to 0.87 (Fig. 1C). The Biogeochemical FCI Model also had substantial empirical support for *Eurycea* spp. with a Δ AIC of 0.91 (Table 2). 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. ranged from 0.70 to 0.76 (Fig. 1B).

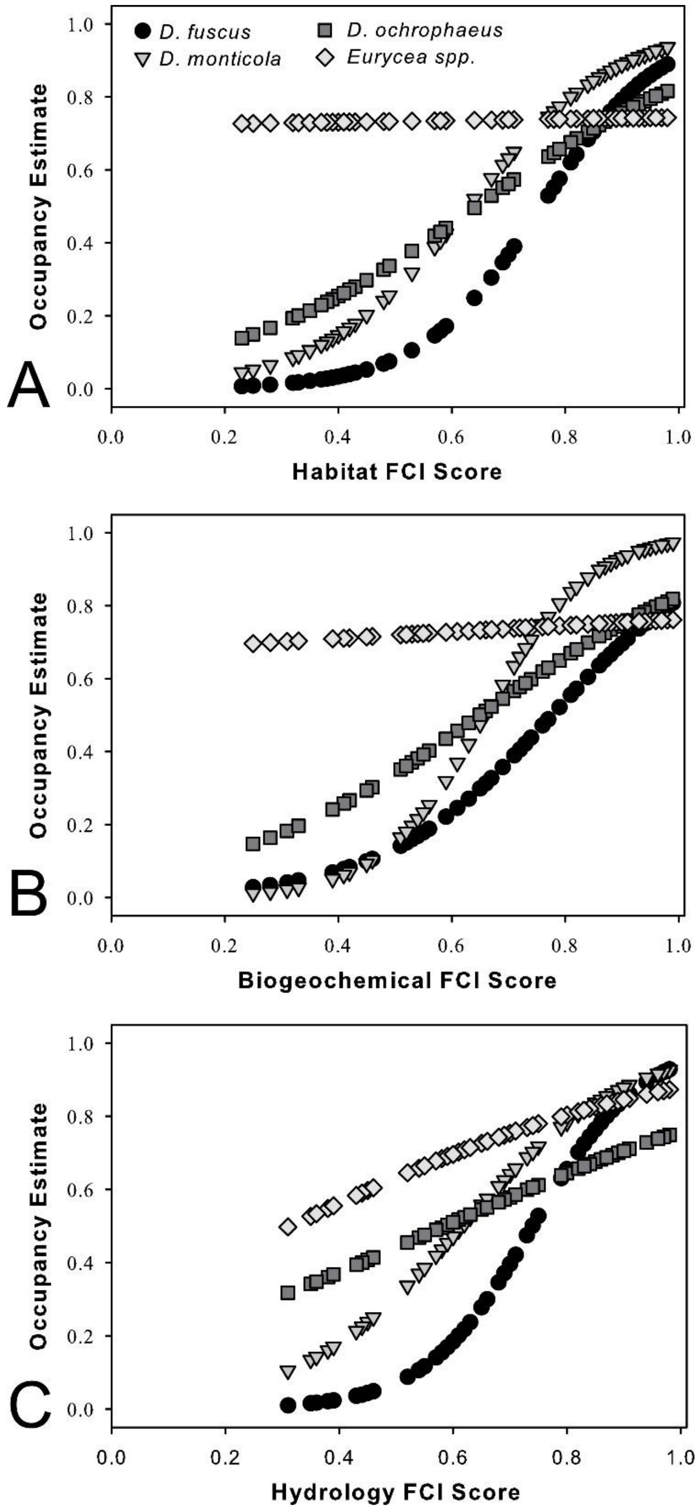

Figure 1. Individual site estimations of occupancy (Ψ) and the Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores (A), the Biogeochemical FCI scores (B), and the Hydrology FCI scores (C) for *Desmognathus fuscus* (circles), *Desmognathus monticola* (triangles), *Desmognathus ochrophaeus* (squares), and *Eurycea* spp. (diamonds), southwest Virginia, summer 2013.

Additionally, there was moderate empirical support for the Habitat FCI Model which had a Δ AIC of 2.91 and an AIC weight of 0.11 (Table 2). 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 3; Fig. 1A).

Table 2. Model results for occupancy probabilities of FCI 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 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 while 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 3. Beta estimates and standard errors for each 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

Abundance

The best abundance model for *Desmognathus fuscus* was the Habitat FCI Model with an AIC weight of 0.60 (Table 4), and a beta estimate of 5.04 (SE = 1.11; Table 5). Individual site abundance estimates for *D. fuscus* range from 0.1 to 3.1 salamanders per site (Fig. 2A). 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).

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). The beta estimate was 5.4 with a standard error of 0.63 (Table 5). Abundance estimates by site ranged from 0.2 to 10.4 for *D. monticola* (Fig. 2A).

Desmognathus ochrophaeus had one abundance model with empirical support, the Habitat FCI Model (AIC weight = 0.9994; Table 4). The beta estimate for the Habitat FCI score was 4.23 (SE = 0.58) and showed a positive correlation to *D. ochrophaeus* abundances (Table 5). Individual site estimates of abundance ranged from 0.4 to 8.5 (Fig. 2A).

The Hydrology FCI Model was the only abundance model with empirical support for *Eurycea* spp. with an AIC weight of 0.88 (Table 4). 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 5; Fig. 2B).

Table 4. Model results for abundance estimates (lambda) of FCI 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 5. Beta estimates and standard errors for each 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

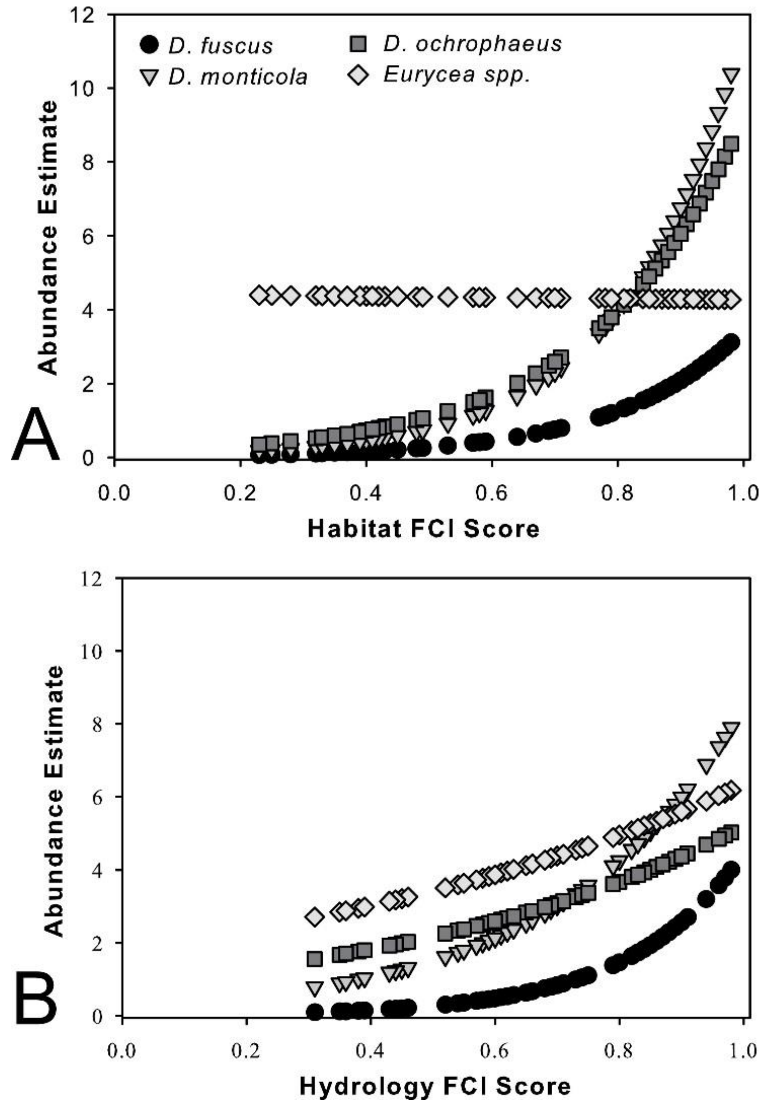

Figure 2. Individual site estimations of abundance (λ) per 25 m x 5 m quadrat and the Habitat Functional Capacity Index (FCI; see Noble et al. 2010) scores (A) and the Hydrology FCI scores (B) for *Desmognathus fuscus* (circles), *Desmognathus monticola* (triangles), *Desmognathus ochrophaeus* (squares), and *Eurycea* spp. (diamonds), southwest Virginia, summer 2013.

Discussion

The three *Desmognathus* species were most correlated with 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 best model for all three *Desmognathus* species. The relationship between *Desmognathus* spp. and the Habitat FCI score may be a function of direct correlation to the variables in the Habitat FCI. Many of the Habitat FCI variables measure riparian conditions such as canopy cover, riparian vegetation species richness, detritus cover, herbaceous cover, and LWD counts. Past research has indicated strong associations between canopy cover and *Desmognathus* abundance (Crawford and Semlitsch, 2008; Ward et al., 2008; Sweeten, 2015). *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 maintaining soil, stream, and air temperatures suitable for salamanders, it also increases other microhabitat characteristics associated with maintaining a cool, moist surface environment such as detritus cover. Crawford and Semlitsch (2008) reported a positive correlation between detritus depth and *D. monticola* and *E. b. cirrigera* abundance. 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 mature, less-disturbed conditions. *Desmognathus* spp. have been shown to be positively correlated with characteristics that often occur 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 Allegheny Mountains of West Virginia, Moseley et al. (2008) reported a positive relationship between *Desmognathus* spp. abundance and time since forest harvest. Ford et al. (2002) demonstrated that *Desmognathus* spp. abundance was most correlated with stand basal area in the 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 increased numbers of salamanders in older forests 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) reported 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 corresponded well with *Eurycea* spp. in either the occupancy or abundance analysis. The occupancy analysis of *Eurycea* spp. indicated empirical support for all three FCI models. However, the beta values for the FCI scores were low with large standard errors. The differences we found between *Desmognathus* and *Eurycea* may be explained by the hypothesis that stream salamanders can be grouped as either disturbance avoiders or disturbance tolerant (Surasinghe and Baldwin, 2015). Disturbance avoiders are generally long-lived salamanders that depend on forests and are sensitive to riparian disturbances. Disturbance tolerant species often can be characterized as short-lived, microhabitat generalists that can withstand riparian land uses (Surasinghe and Baldwin, 2015). Based on results from this study, *Desmognathus* spp. appeared to be disturbance avoiders whereas *Eurycea* spp. were disturbance tolerant. Several studies throughout Appalachia proper have reported that in undisturbed areas *Desmognathus* spp. were the dominant stream salamanders, whereas in disturbed areas *Eurycea* spp. were the dominant stream salamanders (Resetarits, 1997; Hyde and Simons, 2001; Hamilton, 2002). Ward et al. (2008) also reported similar findings in central Appalachia. Abundances of the disturbance tolerant *E. b. cirrigera,* were higher at roadside sites as compared to forested control sites while the inverse was true for *Desmognathus* spp. (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 studies 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 (Burton, 1976; Petranka, 1984; Muenz et al., 2008; Barrett et al., 2012). We posit that one reason *Eurycea* is disturbance tolerant is that this genus is better able to tolerate poor water quality and riparian habitat conditions, as even in degraded conditions prey items are often readily available. However, disturbance tolerant species may not be adapted to nor depend on disturbances, and as a result these species may not be immune to localized extirpations if certain environmental thresholds are exceeded.

The implications of our 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. Our results indicate that the Habitat FCI score is a good indicator of *Desmognathus* spp. presence and abundance. These results support the use of a presence-absence sampling design of *Desmognathus* to indicate physical stream and riparian conditions. For example, if *D. fuscus*, *D. monticola*, and *D. ochrophaeus* are all found at a site then it can be presumed that the Habitat FCI score is high. Our findings also show that *Eurycea* spp. should not be used as an indicator taxa for stream and riparian habitat conditions. As disturbance tolerators *Eurycea* spp. lack strong correlations in occupancy and abundance to riparian conditions.

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