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Validating the Operational Draft Regional Guidebook for the Functional Assessment of High-Gradient Ephemeral and Intermittent Headwater Streams in Western West Virginia and Eastern Kentucky

Chris V. Noble, Elizabeth A. Summers, and Jacob F. Berkowitz

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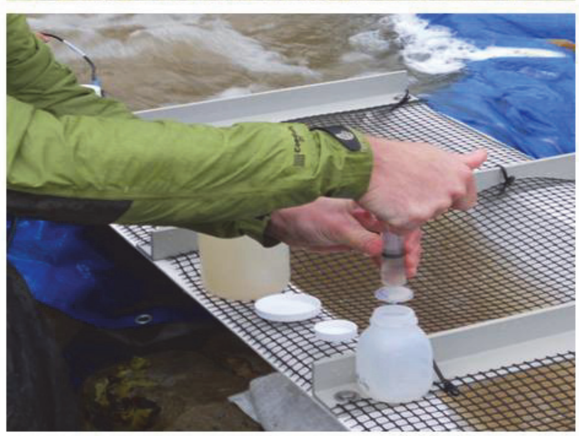


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Abstract

Headwater streams within the Appalachian region perform a number of ecosystem functions potentially impacted by human alterations. The current study sought to validate a Hydrogeomorphic (HGM) assessment, examining 1) Habitat, 2) Biogeochemical Cycling, and 3) Hydrology functions of high-gradient headwater streams in western West Virginia and eastern Kentucky. Validation of the Habitat function focused on: 1) abundance and richness of amphibians, 2) richness and composition of benthic macroinvertebrates, and 3) floristic quality. Validation of the Biogeochemical Cycling function examined 1) nutrient inputs, 2) processing, and 3) the stream loading of nutrients and materials. The Hydrology validation study examined: 1) surface hydrology, 2) sediment transport, and 3) stream channel geomorphology. Within all three functions, sites with high HGM scores performed Habitat, Biogeochemical Cycling, and Hydrology functions at increased levels compared to sites with lower scores. The HGM assessment effectively differentiated between high and low functioning headwater stream ecosystems, and validation results support the use of these functional assessments in headwater stream ecosystems.

Contents

Figures and Tables

Figures

Tables

Preface

This work was performed by the US Army Engineer Research and Development Center (ERDC) in cooperation with US Army Corps of Engineers (USACE) District offices in Huntington, West Virginia, and Louisville, Kentucky. Funding was provided through the Wetlands Regulatory Assistance Program (WRAP). This report was prepared by Chris V. Noble, Elizabeth A. Summers, and Jacob F. Berkowitz, Environmental Laboratory (EL), ERDC. This report was developed in cooperation with a Project Delivery Team of experts familiar with high-gradient headwater streams in the Appalachian region of eastern Kentucky and western West Virginia. The authors are grateful to the following individuals who participated in the Project Delivery Team meeting, contributed their time and expertise to planning the sampling design for the project, and provided comments about the report.

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This work was performed under the general supervision of Dr. Jacob Berkowitz, Acting Chief, Wetlands and Coastal Ecology Branch, EL; Dr. Edmond Russo, Chief, Ecosystem Evaluation and Engineering Division, EL; and Dr. Elizabeth C. Fleming, Director, EL.

COL Jeffrey R. Eckstein was Commander of ERDC; Dr. Jeffery P. Holland was Director.

Executive Summary

This study was designed to validate Hydrogeomorphic (HGM) assessment models for the 1) Habitat, 2) Biogeochemical Cycling, and 3) Hydrology functions of high-gradient headwater streams in western West Virginia and eastern Kentucky. The HGM functional models integrate multiple ecological components over spatial and temporal scales. As a result, no single measurement directly corresponds to HGM scores and determining validation success requires multiple lines of evidence. For the purposes of this report, validation success is defined as the ability of the HGM model to differentiate among levels of ecological functioning across a gradient of study site alteration. Validation includes tests of statistical significance; however, in cases where results were not statistically significant, validation was considered successful if model outcomes displayed the ability to discern between mature second growth forests with least-altered riparian and channel characteristics and altered locations. The models do not provide a predictive tool for individual ecological components (e.g., salamander abundance, carbon transport), but integrate multiple components of ecological functioning in order to remain robust, practicable, and rapid. Results demonstrate that the HGM models effectively differentiated study sites along an alteration gradient. Additionally, a sensitivity analysis of the HGM models identified key variables displaying the strongest effect on model outputs and demonstrated that all variables incorporated into the HGM models influence model outcomes as intended by model developers.

Validation of the Habitat function focused on three ecosystem attributes: 1) the abundance and species richness of stream and riparian salamanders, 2) taxa richness and community composition of benthic macroinvertebrates, and 3) the floristic quality of riparian vegetation. Each supported the current HGM model. Salamander species richness and total abundance were positively correlated with Habitat HGM scores. Sites with watershed alterations resulting from mining activities, silviculture, or agricultural practices had low HGM scores and contained few or no salamanders and differing aquatic invertebrate fauna compared to mature forested study sites. A Floristic Quality Index evaluating vegetation quality also positively correlated with HGM scores.

Validation of the Biogeochemical Cycling function examined 1) nutrient inputs, 2) processing, and 3) the stream loading of nutrients and materials. Each supported the current HGM model. Study sites containing few watershed alterations (e.g., impervious surfaces, agricultural inputs, mining operations) exhibited the highest HGM scores, and displayed high levels of nutrient inputs and material processing (e.g., decomposition, carbon release, nitrogen release). The stream loading of nutrients and materials demonstrated that additional biogeochemical cycling occurs within mature forested catchments with least-altered streams.

The Hydrology validation study examined three aspects of functionality: 1) surface hydrology, 2) sediment transport, and 3) stream channel geomorphology. Each supported the current HGM model. Surface hydrology (e.g., discharge frequency curves) and sediment transport maintained expected relationships at sites with high HGM scores. Sites with low HGM scores deviated from expected relationships, displaying discharge and sediment characteristics outside the range of values observed in least-altered locations. Study sites with least-altered channel and watershed attributes received high HGM scores and maintained a narrow range of channel geomorphology, while study sites containing altered watershed and channel properties had low HGM scores and channel morphologies outside of the expected range.

Sites with high HGM scores performed Habitat, Biogeochemical Cycling, and Hydrology functions at increased levels compared to sites with lower scores. The HGM models effectively differentiated between high and low functioning headwater stream ecosystems, and validation results support the use of these models in functional assessments of high-gradient stream ecosystems.

1 Introduction

1.1 Purpose

In 2010, the Corps of Engineers published a draft hydrogeomorphic (HGM) regional guidebook (Noble et al. 2010) establishing protocols for evaluating potential impacts to high-gradient headwater streams in western West Virginia and eastern Kentucky. This report documents the findings and conclusions of a study investigating the efficacy and applicability of the Corps ecosystem assessment method.

1.2 The HGM assessment method

The Hydrogeomorphic (HGM) approach is a system for developing functional indices and is used to assess the capacity of an ecosystem to perform functions. The HGM Approach was initially designed to be used in the context of the Clean Water Act, Section 404 Regulatory Program permit review process to consider alternatives, minimize impacts, assess unavoidable project impacts, determine mitigation requirements, and monitor the success of mitigation projects. However, a variety of other applications for the approach have been identified, including determining minimal effects under the Food Security Act, designing restoration projects, and managing streams. The HGM Approach was originally designed for use in wetlands; however, the HGM models examined in this report represent a modification of the HGM Approach for use in headwater streams (Noble et al. 2010).

The HGM Approach is a rapid, repeatable, science-based approach for assessing ecosystem functions. Ecosystem functions are the normal or characteristic activities that take place in an ecosystem (Smith et al. 1995). The approach focuses on measuring structural components of the ecosystem using familiar ecological sampling procedures rather than direct measures of ecosystem function. As a result, HGM assessments can be conducted during any time of the year. HGM guidebooks provide simple logic assessment models for measuring multiple ecosystem functions, and outline standard assessment protocols for collecting variables used in the assessment models. Assessment models are developed through the collection of data on sites with impacts representing the alteration gradient observed within a region. Comparing data from a variety of altered sites to locations exhibiting a minimum level of alteration (i.e., reference standard sites) allows for scaling of the assessment models and the generation of functional scores. Functional Capacity Index (FCI) scores range from 0 (no function) to 1 (functioning at a level characteristic of reference standard ecosystems within the same subclass).

US Army Corps of Engineers (USACE) has developed and maintains HGM assessment models for over 25 regional wetland and stream subclasses. In the headwaters of eastern Kentucky and western West Virginia, three regional HGM stream types were identified: high-gradient ephemeral streams, high-gradient intermittent streams, and high-gradient perennial streams. The first two subclasses are addressed in the guidebook *Operational Draft Regional Guidebook for the Functional Assessment of High-gradient Ephemeral and Intermittent Headwater Streams in Western West Virginia and Eastern Kentucky* (USACE 2010). The HGM guidebook and this report do not distinguish between these two subclasses based on data collected during the development of the guidebook and the practical difficulties field personnel experienced differentiating between ephemeral and intermittent stream reaches. The guidebook provides assessment models for three ecosystem functions: Hydrology, Biogeochemical Cycling, and Habitat.

1.3 Validation

The accuracy and efficacy of rapid assessment methods, including HGM, can be strengthened through what Smith et al. 2013 defined as "validation" or testing rapid assessment outcome correctness or reliability using comparisons with independent measures of ecosystem function. Goodall (1972) emphasizes that validation studies determine the degree of agreement between a model and the real system. Validation studies represent an important step in the model-building and ecosystem evaluation process; however, due to the expense required and competing priorities, validation is rarely conducted.

Validation studies examine relationships between rapid assessment results and a series of independent measures of ecosystem function. The independent measures selected often remain well beyond the scope of rapid assessment methodologies. For example, a rapid assessment procedure examining the quality of habitat based on forest composition and structure could incorporate intensive animal population studies into the validation process. Rapid assessment procedures are considered validated and appropriate if the findings of the animal population study (an independent measure) corresponds to rapid assessment results. Validation studies do not represent hypothesis testing, they instead examine the level of agreement observed between rapid assessment outcomes and a number of normal or characteristic activities taking place in an ecosystem (Smith et al. 1995).

Successful validation efforts demonstrate agreement between rapid assessment outputs and multiple independent measures of ecosystem function. By examining multiple independent measures of ecosystem function, model developers gain increased confidence in the validity of assessment results. However, rapid assessments address a wide range of factors that often cannot be captured by a single independent measure. As a result, rapid assessment results will not show agreement with the entire suite of potential measurements. If all relationships between the rapid assessment and validation measures remain unclear, modification of rapid assessment approaches may be required to improve ecosystem assessment procedures (Smith et al. 2013). For the purposes of this report, validation success is defined as the ability of the HGM model to differentiate among components of ecological functioning across a gradient of study site alteration. This includes tests of statistical significance; however, in cases where validation results fail statistical significance testing, the ability of HGM models to differentiate between mature second growth forests with least-altered stream channels and altered sites promotes confidence in the appropriateness of model configuration. This applies a weight of evidence approach in which multiple lines of evidence are combined to reach a conclusion concerning a complex environmental system (Stein et al. 2009). The models do not provide a predictive tool for individual ecological components (e.g., salamander abundance, carbon transport), but integrate multiple components of ecological functioning in order to remain robust, practicable, and rapid.

Testing model accuracy is of particular interest for assessment models created for headwater streams in western West Virginia and eastern Kentucky. In these locations, HGM assessments are being used as part of the permit review process for a variety of activities, including urban/suburban development, surface mining, road construction, silvicultural activities and agricultural expansion. Additionally, existing ecological data are scarce for ephemeral and intermittent streams in the Appalachian Region, which — due to their small size and the transient nature of their hydrologic regime — have received limited attention in scientific literature. Nonetheless, headwater streams constitute > 80% of stream network length and are the primary interface between upland and aquatic habitats (Wipfli et al. 2007).

Several studies have examined the validation of assessment models. To validate functional assessments developed for California vernal pools, Bauder et al. (2009) created "direct" measures using a small number of variables that related to ecological function and compared results to rapid assessment Functional Capacity Index (FCI) scores. Pohll and Tracy (1999) created a hydrologic model for a prairie pothole wetland to test assessment models for water storage functions, and modified assessment models based on study results. Similarly, Hill et al. (2006) applied a hydrologic model for a sinkhole depression on the Tennessee Highland Rim, evaluating the HGM assessment model for hydrology. Finally, Stein et al. (2009) reported findings of a study applying indices of biological integrity and bird species richness to a rapid assessment of estuarine and riverine wetland condition in California, USA.

1.4 Sensitivity testing

Sensitivity analysis provides a method for appraising model performance by applying incremental modifications to model variables (Waide and Webster 1976; Overton 1977). The analysis ensures that model outputs behave as intended, and aids in the identification of key variables displaying the strongest effect on FCI model outputs (Schroeder and Haire 1993). Furthermore, sensitivity testing detects model variables that exhibit minimal impacts on FCI scores, allowing for the possible elimination of unnecessary or redundant variables.

Sensitivity analysis manipulates model variable outputs (i.e., FCI scores) by varying each variable independently while holding all other variable subindices constant (Smith et al. 2013). The effect of each variable on model outcomes (i.e., the "sensitivity" of the model to each variable) is compared by examining the range of observed values as well as minimum and maximum attainable outputs. In general, variables displaying a wider range of values exhibit stronger control over model outcomes. A graphical representation of results depicts the relative strength, or sensitivity, of the model to each variable. More dominant variables demonstrate high slopes and increased distance between static sensitivity testing levels.

1.5 Approach and study design

The current study included developing a Project Delivery Team (PDT), identifying independent validation measures, selecting field sites, installing monitoring equipment and implementing sampling strategy, monitoring study sites, data analysis and reporting. On January 11-12, 2011, the PDT of 31 individuals representing federal agencies (USACE, US Environmental Protection Agency (EPA), US Fish and Wildlife Service (FWS)), state organizations (Kentucky Division of Water, West Virginia Department of Transportation, West Virginia Department of Natural Resources, others), and academia convened and provided input on the validation procedure undertaken by USACE (Appendix C). Following the directives of the guidance provided by the HGM guidebook, and available literature sources, USACE staff identified independent measures of ecosystem function. Each assessment model addressed by the HGM guidebook underwent validation via comparison to a series of independent measures (Table 1.5.1).

Table 1.5.1. HGM assessment models validation, examples of independent measure, and technique utilized.

Ecosystem assessment model	Independent measure	Measurement technique
Habitat	Salamander abundance and species richness	Cover board and basket samplers
	Invertebrate taxa richness and community metrics	Basket samplers
	Vegetative composition	Floristic Quality Index
Biogeochemical Cycling	Nutrient and material input	Leaf litter fall traps
	Nutrient and material processing	Leaf litter decomposition
	Nutrient and material cycling and water quality	Nutrient and sediment loading, water quality parameters
Hydrology	Transport of water to downstream systems	Analysis of characteristic discharge and sediment relationships
	Transport of sediment downstream	Bed load and suspended sediment transport
	Stream geomorphology	Width: depth ratios

Following selection of independent measures, staff identified 10 validation study sites representing a gradient of alteration commonly observed within the area. Study sites included least-altered streams within mature, second-growth, forested watersheds as well as sample locations exhibiting impairments from agricultural, silvicultural, recreational, and mining activities. HGM assessment scores were calculated for the three functions addressed in the HGM guidebook (Habitat, Biogeochemical Cycling, and

Hydrology). Installation of monitoring equipment and implementation of a sampling schedule also followed recommendations of the PDT and included a variety of measurements designed to capture the observed variability within the study area and address the independent measures selected (Table 1.5.2). Figure 1.5.1 provides a schematic of the sample design utilized. All measurements and methods employed receive in-depth descriptions in the following sections addressing each HGM function.

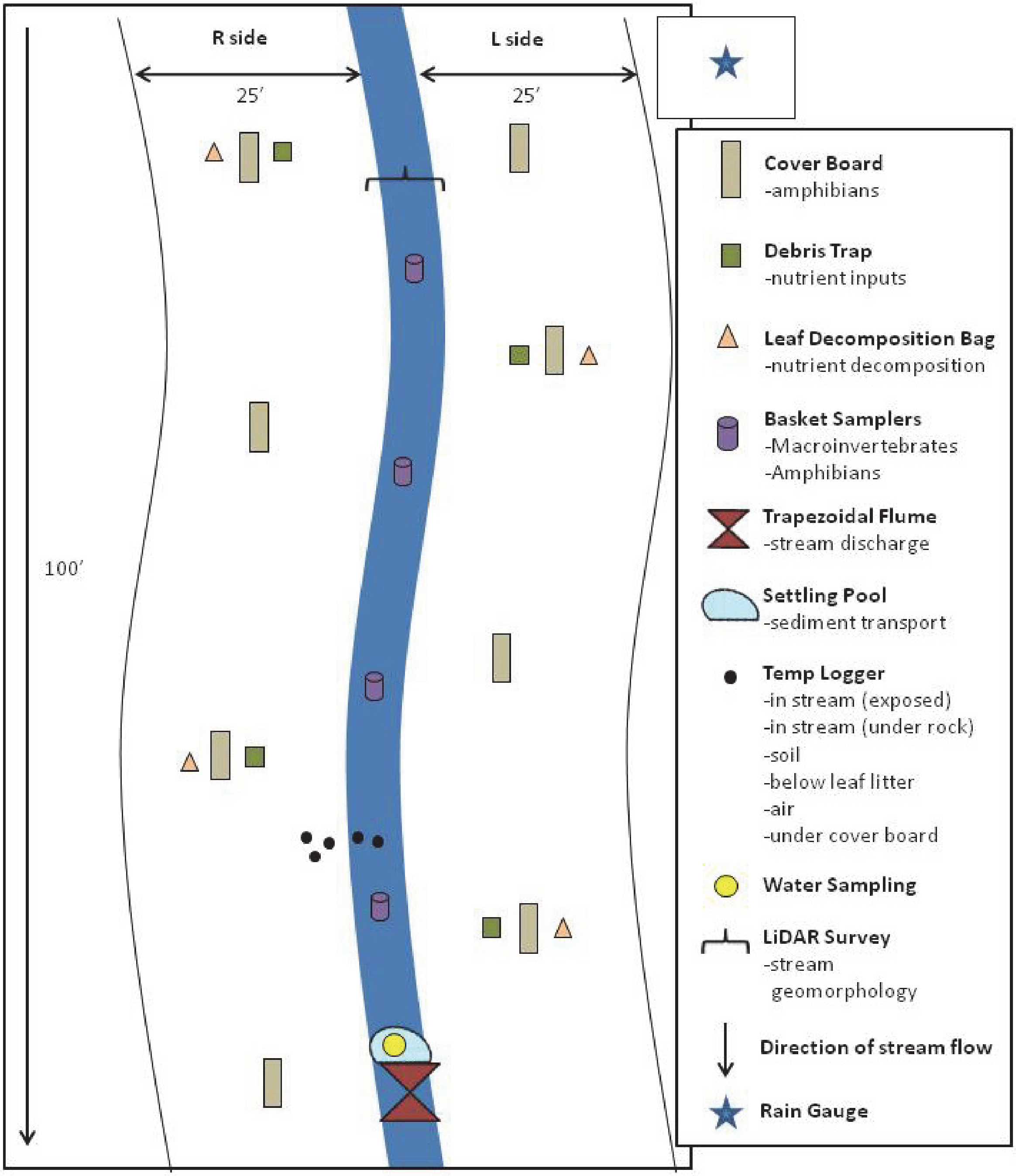
Following implementation of the sampling plan, data collection at each study site occurred during 2011 and 2012. Independent measures were compared with rapid assessment outcomes (i.e., FCI scores) via correlation analysis, and examination of departures of characteristics observed at the least-altered sample locations.

Each HGM function was independently validated, including the selection of the applied measures and the identification of least-altered study sites. For example, the Habitat function is validated by comparing results from the four least-altered study sites (selected based on stream and watershed characteristics affecting habitat) and sites exhibiting alteration due to surface mining, agriculture, and recent forestry activities. The Hydrology function is validated by comparing results from the five least-altered study

Table 1.5.2. Examples of equipment and sampling techniques applied during the validation study.

Equipment or sampling activity	Purpose	HGM function(s) applied
Trapezoidal flume with automated data logger	Monitor discharge and loading relationships	Hydrology, Biogeochemical Cycling, Habitat
Cover board	Salamander sampling	Habitat
Leaf litter fall trap	Quantify nutrient and material inputs	Biogeochemical Cycling
Leaf litter decomposition bag	Quantify nutrient cycling and organic matter processing	Biogeochemical Cycling
Basket sampler	Sample salamander and invertebrate populations	Habitat
Settling pool	Quantify sediment	Hydrology
Temperature data loggers in stream, soil, leaf litter, air, and cover board	Monitor trends and thresholds in temperature	Habitat, Biogeochemical Cycling
Channel morphology surveys	Determine stream channel width: depth ratios	Hydrology
Rain gauge	Determine rainfall normality	Hydrology, Biogeochemical Cycling, Habitat

Figure 1.5.1. Diagram of the location and number of variables collected within a 100-foot reach at validation study sites.



sites (selected based on characteristics affecting hydrology) and sites exhibiting alteration due to surface mining, agriculture, and recent forestry activities. Results represent the simplest and most straightforward analysis possible to validate the HGM assessment models examined. Each HGM assessment model was compared to a variety of independent measures of ecosystem function. HGM models are not designed as predictors of individual ecosystem characteristics (e.g., number of salamander species present, amount of nutrients transported); thus, the intention of this study is not to provide a comprehensive evaluation of the relationships between all potential ecosystem functions. Following data collection and analysis, the current report was generated and prepared for USACE HQ.

1.6 Report organization

The report is organized into sections addressing site characteristics, validation of the three ecosystem functions addressed in the HGM guidebook, and results of sensitivity analysis. Chapter 2 provides information for each study site, including stream characteristics, structure, and composition of vegetation, watershed size and condition, and a description of site conditions, alteration, and other observed impacts. Chapter 3 presents validation information concerning the Habitat assessment model. Chapter 4 contains validation results addressing the Biogeochemical Cycling model. Chapter 5 includes the results of validation efforts examining the Hydrology assessment model. Chapters 3-5 each summarize the results of a sensitivity analysis designed to determine the relative strength and appropriateness of each FCI model component. Appendix A is a detailed sensitivity analysis, Appendix B relates to rainfall normality, and Appendix C documents the Project Delivery Team summary of recommendations.

2 Study Area

The validation study was conducted in West Virginia, USA, from March 2011 – July 2012. Validation study sites were associated with the high-gradient headwater stream regional subclass and were located within the reference domain addressed by the regional guidebook (Figure 2.1; Noble et al. 2010). Study sites captured the various types of alterations commonly observed within the reference domain, including mature forests along channels with minimal alteration, constructed channels associated with valley fill activities, and impaired channels displaying silvicultural and agricultural impacts. Validation study sites exhibited a range of FCI scores for each of the three functions measured (Table 2.1). Study sites were located in areas with unlimited access to investigators, and were selected on the basis of aspect, catchment size, estimated hydroperiod, underlying geology, and characteristics needed to establish alteration gradient. However, because the regional guidebook is designed for use across a large physiographic area, site selection was not strictly limited by these factors.

Figure 2.1. Map of the reference domain for high-gradient headwater streams in western West Virginia and eastern Kentucky.

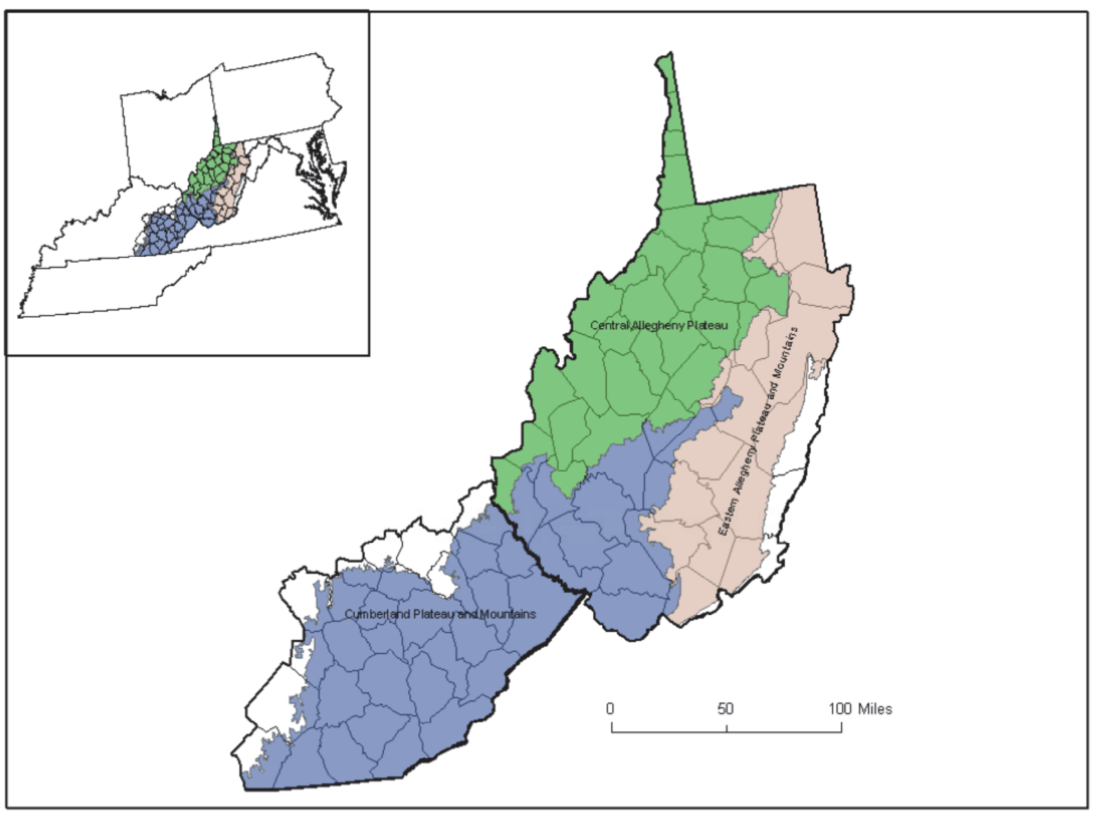


Table 2.1. FCI scores at validation study sites.

Site #	Habitat FCI Score	Biogeochemical Cycling FCI Score	Hydrology FCI Score
1	0.71	0.89	0.80
2	0.95	0.93	0.95
3	0.94	0.97	0.95
4	0.46	0.72	0.66
5	0.95	0.91	0.96
6	0.25	0.61	0.48
7	0.50	0.62	0.38
8	0.72	0.65	0.76
9	0.87	0.90	0.87
10	0.21	0.29	0.36

The validation study included a total of 10 headwater stream sites, located in the Upper and Lower Kanawha, Cheat, Upper Guyandotte and Twelvepole watersheds (WVDEP 2012). Watershed sizes ranged from 1.03 to 12.93 ha. Sites were located in the Central Allegheny Plateau, Eastern Allegheny Plateau, and Cumberland Plateau and Mountains Major Land Resource Areas (MLRAs). Sample size and site selection was driven by both financial and logistical restraints. Despite the small sample size, many significant relationships between HGM models and validation parameters were observed. In cases where statistical significance was not observed, a threshold or weight of evidence approach was applied when comparing HGM scores with validation study results.

Site 1

The study area encompasses the upper end of a small watershed supporting a hardwood forest (Figure 2.2). Stream channel slope is 9.9 percent. There is a small cleared utility right-of-way at the upper (east) end of the site. Site 1 is surrounded by upland forest. The watershed is 1.03 ha in area, and contains a cleared recreation area on the ridge-top a short distance north of the plot. The forest edge in this vicinity supports a number of non-native invasive plant species. Based on cores collected from the three largest trees present, the stand contains trees up to 82 years old (Figure 2.3). The study area exhibits impacts from silviculture and recreation.

The canopy of Site 1 is dominated by tuliptree (*Liriodendron tulipifera*), red maple (*Acer rubrum*), white oak (*Quercus alba*), mockernut hickory (*Carya alba*), and scarlet oak (*Quercus coccinea*). Other trees present,

Figure 2.2. Site 1.

Figure 2.3. Stand age was determined by taking tree cores, using an increment borer, from the three largest-diameter trees within the riparian zone at each study site.

but less common, include blackgum (*Nyssa sylvatica*), black oak (*Quercus velutina*), northern red oak (*Quercus rubra*), and sycamore (*Platanus* *occidentalis*). A single Virginia pine (*Pinus virginiana*) occurs on the upper edge of the site.

Site 1 has a subcanopy layer dominated by beech (*Fagus grandifolia*) and red maple with a few scattered sugar maples (*Acer saccharum*).

The shrub layer is dominated by pawpaw (*Asimina triloba*), mapleleaf viburnum (*Viburnum acerifolium*), and white ash (*Fraxinus americana*). There are also scattered individuals of northern spicebush (*Lindera benzoin*), beech, blackgum, wild black cherry (*Prunus serotina*), sweet cherry (*Prunus avium*), slippery elm (*Ulmus rubra*), sourwood (*Oxydendrum arboreum*), red maple, and sugar maple.

The most common species in the herbaceous layers include bearded shorthusk grass (*Brachyelytrum erectum*), Christmas fern (*Polystichum acrostichoides*), white snakeroot (*Ageratina altissima*), and poison ivy (*Toxicodendron radicans*). Two non-native invasive species, Japanese honeysuckle (*Lonicera japonica*) and Oriental bittersweet (*Celastrus orbiculatus*) occupy an area on the north side of the site where they are encroaching from the adjacent recreation area.

Site 2

Site 2 is characterized by a headwater stream in a steep-walled ravine consisting of hardwood forest (Figure 2.4). Stream channel slope is 22.9 percent. Upland hardwood forest surrounds the site, which encompasses a watershed of 6.53 ha. Based on tree core data, the stand contains trees up to 82 years old. There are no signs of recent alterations, except for a foot/bike trail that crosses the upper half of the site.

The canopy at Site 2 is dominated by white oak, tuliptree, sycamore, and beech. There are also scattered individuals of northern red oak, chestnut oak (*Quercus prinus*), yellow buckeye (*Aesculus flava*), sweet birch (*Betula lenta*), and bitternut hickory (*Carya cordiformis*).

This site has a subcanopy layer dominated by umbrella tree (*Magnolia tripetala*), sugar maple, beech, and sweet birch. There are also scattered individuals of eastern hemlock (*Tsuga canadensis*), blackgum, cucumber-tree (*Magnolia acuminata*), basswood (*Tilia americana*), black oak, and red maple.

Figure 2.4. Site 2.

The shrub layer is dominated by northern spicebush (*Lindera benzoin*), beech, and American witchhazel (*Hamamelis virginiana*). There are also scattered flowering dogwood (*Cornus florida*), red maple, alternateleaf dogwood (*Cornus alternifolia*), hophornbeam (*Ostrya virginiana*), American hornbeam (*Carpinus caroliniana*), and slippery elm. Buffalo nut (*Pyrularia pubera*), wild hydrangea (*Hydrangea arborescens*), mountain laurel (*Kalmia latifolia*), American holly (*Ilex opaca*), and a deciduous azalea (*Rhododendron* sp.) are present but rare.

The herbaceous layer contains several species with no clear visual dominants. Species present indicate a high quality site without significant recent alterations.

Site 3

Site 3 is characterized by a headwater stream in a steep ravine supporting a mature hardwood forest and a sparse understory (Figure 2.5). Stream channel slope is 16.9 percent. Based on tree core data, the stand contains trees up to 94 years old. The study watershed exhibits no signs of recent alteration and encompassed an area of 4.13 ha.

Figure 2.5. Site 3.

The canopy at Site 3 is dominated by beech with lesser amounts of sugar maple, red maple, black oak, sweet birch, and tuliptree.

The shrub layer is sparse and is dominated by beech and northern spicebush.

The herbaceous layer is dominated by Japanese stiltgrass (*Microstegium vimineum*), but herbaceous cover is sparse within the study area.

Site 4

The headwater stream at Site 4 is a constructed groin ditch running along the edge of a valley fill (Figure 2.6). Stream channel slope is 14.4 percent. The adjacent hillside exhibits natural topography on one side of the stream. The surrounding watershed (9.13 ha) contains a valley fill and a weedy regraded portion of the original hillside. Upland hardwood forest surrounds the valley fill area. Based on tree core data, the stand contains trees no older than 13 years old. The study site shows evidence of recent mining activity.

Figure 2.6. Site 4.

The canopy at Site 4 is young and sparse and is dominated by sycamore with a few scattered sweet birch, princesstree (*Paulownia tomentosa*), and tuliptree.

There is no shrub layer present.

The herbaceous layer is nearly continuous and is dominated by tall fescue (*Schedonorus arundinaceus*), broomsedge bluestem (*Andropogon virginicus*), sericea lespedeza (*Lespedeza cuneata*), and Japanese stiltgrass.

Site 5

Site 5 is characterized by a headwater stream in a steep-walled ravine supporting a hardwood forest (Figure 2.7). Stream channel slope is 10.2 percent. Upland hardwood forest surrounds the site, which encompasses a watershed of 12.93 ha. Based on tree core data, the stand contains trees as old as 109 years. The study area exhibits no signs of recent alterations.

Figure 2.7. Site 5.

The canopy at Site 5 is dominated by northern red oak, beech, sugar maple, and red maple. There are also scattered individuals of cucumber-tree, sourwood, white oak, sweet birch, eastern hemlock, blackgum, bitternut hickory, and basswood.

A subcanopy is present and includes umbrella tree, beech, sugar maple, basswood, and red maple.

The shrub layer is dominated by northern spicebush and also includes American witchhazel.

The herbaceous layer contains several species and is dominated by Christmas fern.

Site 6

Site 6 is a constructed groin ditch with valley fill on one side of the drainage and the original hillside on the other side (Figure 2.8). Stream channel slope is 4.3 percent. A portion of this original hillside is still forested in the downstream portion of the study area. The watershed area is 1.39 ha and contains trees up to 17 years old. The study site shows evidence of recent mining activity.

Figure 2.8. Site 6.

The canopy is young and sparse and is dominated by tree-of-heaven (*Ailanthus altissima*), and European alder (*Alnus glutinosa*) with scattered white pine (*Pinus strobus*), wild black cherry, boxelder (*Acer negundo*), sycamore, black locust (*Robinia pseudo-acacia*), slippery elm, and white ash.

The shrub layer is sparse on the valley fill side of the drain and dense on the natural hillside side of the drain. It is dominated by three species of blackberry/raspberry (*Rubus* sect. *Arguti*, *Rubus occidentalis*, and *Rubus phoenicolasius*), staghorn sumac (*Rhus typhina*), and autumn olive (*Elaeagnus umbellata*) with scattered tuliptree, redbud (*Cercis canadensis*), black locust, green ash (*Fraxinus pennsylvanica*), slippery elm, flowering dogwood, and gray dogwood (*Cornus racemosa*). Wild hydrangea is restricted to the forested portion of the original hillside in the southeast corner of the plot.

The herbaceous layer is dominated by tall fescue, (*Sericea lespedeza*), tall goldenrod (*Solidago altissima/canadensis*), hairy white oldfield aster (*Symphyotrichum pilosum*), and bird's-foot trefoil (*Lotus corniculatus*).

Site 7

Site 7 is a headwater stream in a grazed pasture, with steep slopes immediately adjacent to the study area (Figure 2.9). Stream channel slope is 17.9 percent. The watershed is 1.28 ha, and contains trees up to 67 years old. The site is surrounded by pasture, allowing direct cattle access to the stream. To prevent damage to equipment, a 50-ft x 100-ft portion of the sampling area was fenced off to exclude cattle during the entire study period. The installed fencing only excluded the cattle from the instrumented area; cattle had access to areas immediately above, adjacent to, and below the instrumented area. The fencing allowed unimpeded access to salamanders, invertebrates, and other small animals.

Figure 2.9. Site 7.

The canopy at Site 7 consists of scattered trees along the drainage and is dominated by black walnut (*Juglans nigra*) and white oak. Other trees include scattered tuliptree, sycamore, white ash, American elm (*Ulmus americana*), black cherry, black oak, redbud, red mulberry (*Morus rubra*), and pignut hickory (*Carya glabra*).

The shrub layer is very sparse and includes autumn olive and hawthorn (*Crataegus* sp.).

The herbaceous layer is dominated by non-native species, including tall fescue in the open pasture areas and Japanese stiltgrass in the shaded ravine along the creek. Most native flora occurs on the steep slopes above the creek where grazing is least intense.

Site 8

Site 8 is a headwater stream surrounded by a mature riparian buffer zone with large canopy trees of a variety of species (Figure 2.10). Stream channel slope is 6.1 percent. Watershed size is 8.9 ha and contains trees as old as 77 years. The study area is surrounded by a young, even-aged hardwood stand and includes a planted Japanese larch (*Larix kaempferi*) monoculture which contains many dead trees. Although the stream channel appears stable and unaltered, the surrounding watershed shows signs of silvicultural impacts.

The canopy at Site 8 is dominated by northern red oak, black oak, red maple, and white ash with scattered sweet birch, black cherry, blackgum, sugar maple, and white ash.

The shrub layer is dominated by American witchhazel and northern spicebush with scattered striped maple (*Acer pensylvanicum*), beech, sweet birch, and lowbush blueberry (*Vaccinium pallidum*).

The herbaceous layer is dominated by New York fern (*Thelypteris noveboracensis*), fan clubmoss (*Lycopodium digitatum*), and common greenbrier (*Smilax rotundifolia*). Lady fern (*Athyrium filix-femina* var. *asplenioides*) and Christmas fern are also conspicuous.

Figure 2.10. Site 8.

Site 9

Site 9 is characterized by a headwater stream in a steep-walled ravine supporting a hardwood forest (Figure 2.11). Stream channel slope is 11.0 percent. Upland hardwood forest surrounds the site, which encompasses a watershed of 4.37 ha. The stand contains many large trees up to 103 years old. There are no signs of recent alterations within the study area.

The canopy at Site 9 is tall and is dominated by tuliptree, sugar maple, and basswood with scattered beech, sweet birch, black cherry, and white ash.

The shrub layer is dominated by striped maple and beech with scattered sugar maple, American witchhazel, sweet birch, elderberry (*Sambucus* sp.), and cucumber-tree. Also present are black cherry, hophornbeam, and alternateleaf dogwood. A few scattered small individuals of Japanese barberry (*Berberis thunbergii*) were observed.

The herbaceous layer is dominated by spinulose woodfern (*Dryopteris carthusiana*), New York fern, Christmas fern, Canadian woodnettle (*Laportea canadensis*), and a highbush blackberry. Species present indicate no recent alterations.

Figure 2.11. Site 9.

Site 10

Site 10 is a constructed, boulder-lined groin ditch in a large valley fill (Figure 2.12). Stream channel slope is 20.9 percent. The site is part of a large, active mine complex containing many valley fills. The watershed is 3.12 ha and contains a series of settling ponds which control on-site hydrology. The site contains trees up to 12 years old; all trees were planted at the time of groin ditch construction. The surrounding site is a large valley fill dominated by non-native grasses and forbs with scattered young trees. The study site shows evidence of recent mining activity, resulting in altered hydrology.

The canopy at Site 10 is sparse and is dominated by young planted white pine with a few scattered black cherry, boxelder, sycamore, slippery elm, and tuliptree.

The shrub layer is dominated by shrub lespedeza (*Lespedeza bicolor*) with scattered but dense patches of highbush blackberry and black raspberry and scattered clumps of autumn olive and multiflora rose (*Rosa multiflora*).

There are also a few scattered individuals of slippery elm, wild black cherry, sugar maple, red maple, redbud, and flowering dogwood.

The herbaceous layer is dominated by a dense cover of tall fescue with dense areas of *Sericea lespedeza* and scattered patches of tall goldenrod and hairy white oldfield aster.

Figure 2.12. Site 10.

3 Validation of the Habitat function

3.1 Introduction

3.1.1 Functional definition and independent measures selected

The following chapter presents the results of a validation study relating selected independent measures of habitat functioning to Habitat FCI scores. The purpose of this section focuses on testing the ability of the Habitat HGM rapid assessment model to differentiate between site conditions occurring along a gradient of alteration. The study is not intended to provide a comprehensive evaluation of all potential habitat functions, interactions, or relationships as they apply to investigations evaluating stream or landscape ecological research.

The *Operational Draft Regional Guidebook for the Functional Assessment of High-gradient Ephemeral and Intermittent Headwater Streams in Western West Virginia and Eastern Kentucky* (Noble et al. 2010) defines the HGM Habitat function as:"...the capacity of a high-gradient headwater stream ecosystem to provide critical life requisites to selected components of the vertebrate and invertebrate wildlife community."

Validation of the assessment model followed the recommendations of (Smith et al. 2013) by comparing assessment results against several selected independent measures of ecosystem function identified by a panel of experts (Project Delivery Team). The independent measures selected address the ability of study sites to provide habitat for three floral and faunal communities which utilize headwater stream habitat: 1) salamanders, 2) benthic macroinvertebrates, and 3) riparian vegetation. The selection of independent measures incorporated input from the Project Delivery Team (PDT), approaches suggested in the HGM guidebook, and a review of available literature sources. For example, Noble et al. (2010) recommended that species richness of amphibians and macroinvertebrates be applied as independent measures of habitat functioning and outlined the importance of vegetative structure and composition to habitat quality.

3.1.2 Summary of findings

Each of the three communities examined as part of the Habitat model validation effort support the current HGM model configuration.

Salamander species richness and total abundance was positively correlated with Habitat FCI values. The greatest number of individual salamanders and the greatest number of salamander species were detected at sites displaying mature forested watersheds, high percent of forest canopy cover and large amounts of woody debris. Few or no salamanders were detected at study sites displaying low canopy cover and watershed alterations resulting from former mining activities or agricultural practices (e.g., grazing by cattle).

The greatest number of aquatic invertebrate taxa was detected at study sites displaying watershed and channel characteristics at or near the reference standard range, including mature forest canopy, unaltered channel characteristics, and few watershed alterations. Conversely, the lowest number of aquatic invertebrate taxa was detected at sites that lacked mature forest structure or that had been subjected to watershed alterations such as tree harvesting and mining activities (e.g., valley fill, engineered stream channels).

Floristic Quality Assessments used vegetative inventories obtained from each study site (Section 3.4). Floristic Quality Index (FQI) values correlated with Habitat FCI values. Study sites with watershed and riparian characteristics at or near the reference standard range contained the greatest number of sensitive plant taxa. Study sites exhibiting degradation from channel or watershed alteration displayed lower FQI values.

Findings demonstrate that study sites with the highest Habitat FCI values supported the greatest richness of salamanders and benthic macroinvertebrates, and higher quality vegetation. Study sites with low Habitat FCI values displayed the greatest levels of watershed and channel alteration, and supported fewer taxonomic groups within the three communities examined. Sensitivity analysis results demonstrate that all variables incorporated into the Habitat model influence FCI scores as intended by model developers (Section 3.5).

3.2 Salamanders

3.2.1 Rationale for selecting the independent measure

Salamanders are major contributors to energy flow and nutrient cycling in eastern forests, often acting as the dominant predator in headwater streams (Burton and Likens 1975; Ohio EPA 2002; Spight 1967). Headwater streams provide important habitat for salamanders due to the absence of fish, and several species (e.g., *Plethodon glutinosus* and *Eurycea cirrigera*) have been observed at higher frequencies in headwater streams than in perennial streams (Barr and Babbit 2002; Schneider 2010). Amphibians represent excellent indicators of ecosystem health due to their susceptibility to environmental stressors (Welsh and Ollivier 1998). Characteristics such as highly permeable skin, unshelled eggs, limited dispersal capability, and a biphasic life history that requires both aquatic and terrestrial habitats cause amphibians to be sensitive to habitat degradation (USEPA 2002, Wells 2007). Population studies have shown significantly lower salamander abundances in headwater streams affected by watershed alterations such as silviculture and residential developments (Petranka et al. 1993; Wilson and Dorcas 2003).

3.2.2 Methods

Terrestrial salamander sampling within the riparian zone utilized eight plywood cover boards installed at each site (four on each side of the channel; Figure 3.2.1; Wilson and Gibbons 2009). Each board measured 24 in. x 48 in. x 0.5 in. (61 cm x 122 cm x 1.3 cm). Boards underwent aging outside for a minimum of three months before placement at sites. Cover boards were overturned and checked for the presence of salamanders once per month from August – November 2011 and March – June 2012, and species and abundance of all salamanders detected was recorded. Sampling was not performed during December – February because many salamanders hibernate underground during freezing temperatures. At Site 6, two *Plethodon cinereus* were detected in March 2011; however, this occurred outside of the defined study period. As a result, these individuals were excluded from analysis.

Salamanders present in the channel were quantified using basket samplers (Figure 3.2.2). Basket samplers were created using 1.8 x 2.5-cm hexagonal Polyvinyl Chloride (PVC) mesh folded into rectangular, closed baskets (19.5 x 28.5 x 9.5 cm). Each basket was filled with 4.5 kg of purchased cobble (average diameter 2.8 cm, average mass 38.5 g), and the remaining basket space was filled with leaves collected on-site. Four replicate baskets were installed at each site during October 2011, January 2012, and April 2012. Basket samplers were placed at approximately equal distances throughout the study reach and were staked to stream substrate to maintain moisture and hold the sampler in place. Basket samplers remained in place for approximately one month to allow colonization by salamanders. Samplers were only collected when both the streambed and samplers were wet from recent rainfall; however, surface water was not always present at

Figure 3.2.1. (clockwise from top left) 1) Cover boards distributed throughout the riparian zone, 2) cover boards were checked once per month for the presence of salamanders, 3) northern slimy salamander (*Plethodon glutinosus*) found under a cover board, and 4) northern two-lined salamander (*Eurycea bislineata*) found under a cover board.

the time of collection. During collection, samplers were lifted from the streambed and placed in a large tub of water, where cobble and large leaves were washed thoroughly and removed. Adult salamanders were identified and released onsite. All larval salamanders detected in cobble traps were preserved in ethanol for later identification in the lab.

Salamander metrics calculated at each site included species richness (total number of species detected among all sampling dates) and abundance (total number of individuals detected among all sampling dates; Heyer et al. 1994). Calculated metrics combined both cover board and basket sampler data. Statistical procedures included normality testing applying the Shapiro-Wilk test ($p>0.05$ indicates a normal distribution) followed by Pearson Product Moment Correlation analysis ($\alpha = 0.05$; JMP, SAS Institute 2012). Abundance data were square-root transformed to satisfy normality assumptions, a common procedure for estimates of abundance (Sokal and Rohlf 1995).

Figure 3.2.2. Basket samplers were placed within the stream channel for one month to sample salamanders.

Vegetative cover may facilitate salamander activity by buffering against temperature extremes and maintaining litter moisture (Fraser 1976, Jaeger 1980, Johnson et al. 1985). Six temperature loggers (Onset, Stowaway TidBiT, TBI32-20+50, Bourne, MA) recorded temperature (Celsius) every 8 hours (0400, 1200 and 2000 hrs). At each site, loggers recorded temperature in ambient air, below riparian/buffer zone detritus, in the stream channel (exposed and under substrate), 4.0 in. (10 cm) below the soil surface, and under a cover board. Average daily temperature range for each logger was calculated by subtracting the daily minimum temperature at each site from the daily maximum temperature. Statistical tests compared average daily temperature ranges for each sample location to Habitat FCI scores using Pearson Product Moment Correlation analysis (JMP, SAS Institute 2012).

3.2.3 Results

Of the nine salamander species observed across all study sites, seven species have known distributions encompassing all counties in which study sites were located (Table 3.2.1, Amphibiaweb 2013). Salamander species richness ranged from 0 – 7 species per site across all sampling dates (Table 3.2.1; Figure 3.2.3). Species richness data displayed normal distribution (Table 3.2.2). Comparison of species richness with the HGM Habitat FCI score yielded a significant positive correlation (r = 0.664, n = 10, P = 0.036, Table 3.2.3, Figure 3.2.4).

Total salamander abundance at each site ranged from 0 – 36 individuals (Figure 3.2.5). Abundance (square-root transformed) was normally distributed and positively correlated (r = 0.709, n = 10, P = 0.022) with the Habitat FCI score (Tables 3.2.2, 3.2.3; Figure 3.2.6). These results demonstrate that the HGM Habitat model was correlated with metrics of the amphibian community associated with headwater streams and adjacent riparian zones within the study area.

Temperature logger data shows that the daily temperature range under leaf litter was greatest at altered reaches (Figure 3.2.7). Mean leaf litter temperature range at sites varied from 0.58 – 3.15°C per day. Leaf litter temperature range data were normally distributed (Table 3.2.2) and negatively correlated with Habitat FCI scores (r = -0.85, n = 10, P = 0.002; Table 3.2.3; Figure 3.2.8). Similar to leaf litter temperature data, the other five temperature loggers (ambient air, below the soil surface, in stream channel, in stream under substrate, and under a cover board) also displayed temperature ranges which were negatively correlated with Habitat FCI scores (data not shown).

Table 3.2.1. Amphibian species detected under cover boards and in basket samplers at HGM Study sites, West Virginia, 2011 – 2012. “Not within range” indicates a study area outside the geographic range of the species.

Species	SITE 1	SITE 2	SITE 3	SITE 4	SITE 5	SITE 6	SITE 7	SITE 8	SITE 9	SITE 10
Desmognathus fuscus		X						X	X	
Desmognathus monticola		X			X				X	
Desmognathus ochrophaeus					Not in range			X	X	Not in range
Eurycea bislineata	X			X			X		X	
Gyrinophilus porphyriticus			X						X	
Plethodon cinereus								X		
Plethodon glutinosus	X	X	X		X		X	X	X	
Plethodon richmondi	X		X	X				Not in range	Not in range	
Pseudotriton ruber								X	X	
Habitat FCI Score	0.71	0.95	0.94	0.46	0.95	0.25	0.50	0.72	0.87	0.21

Figure 3.2.3. Salamander species richness at each sample location.

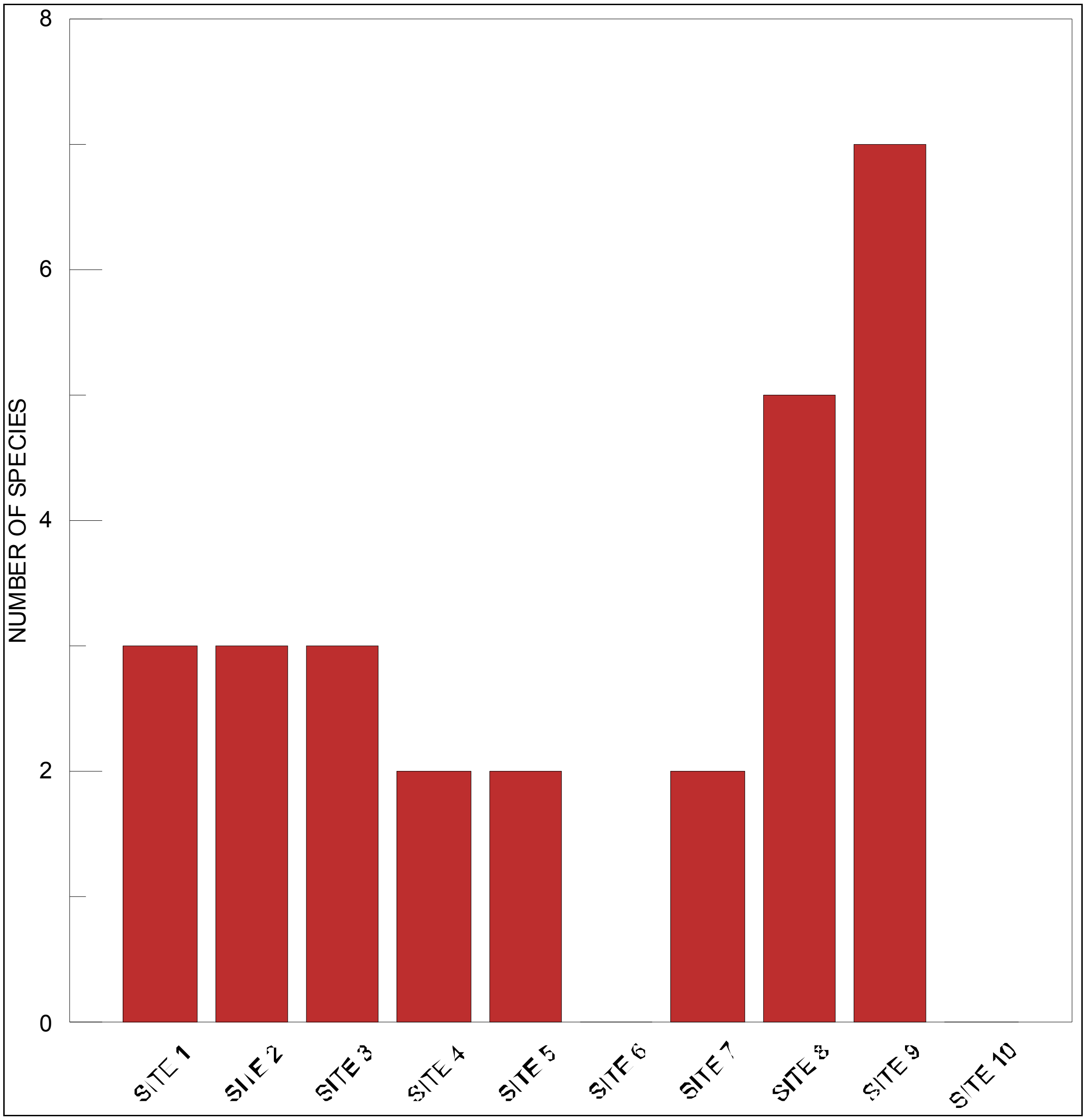


Table 3.2.2. Tests of Normality – Species Richness, sqrt[Abundance] and Leaf Litter Temperature Range

Parameter	Shapiro-Wilk		
	W	df	p
Species Richness	0.909	10	0.275
sqrt(Abundance)	0.940	10	0.550
Litter Temp Range	0.922	10	0.375

Figure 3.2.4. Salamander species richness compared with HGM habitat FCI score.

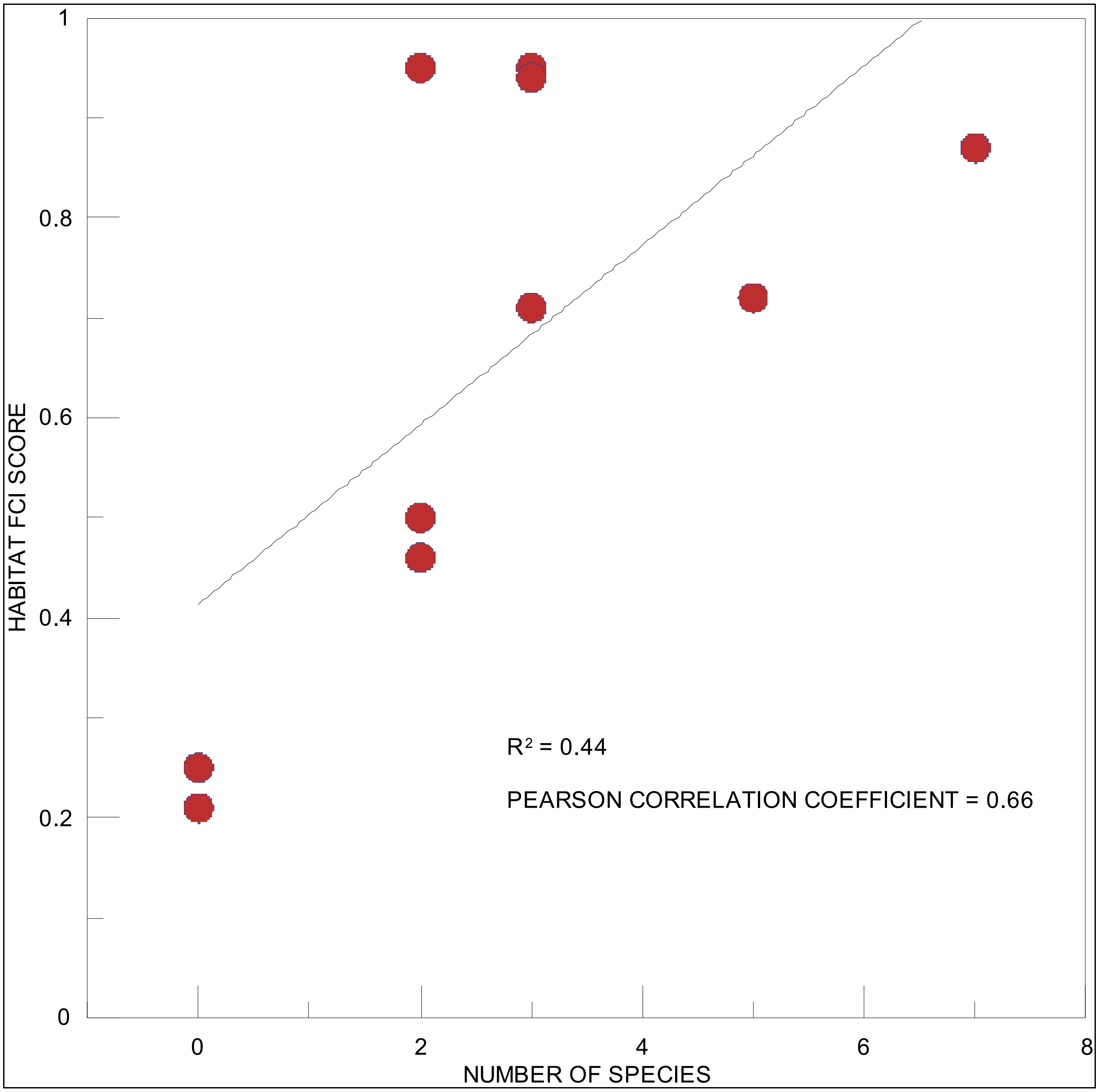


Figure 3.2.5. Total salamander abundance at each sample location.

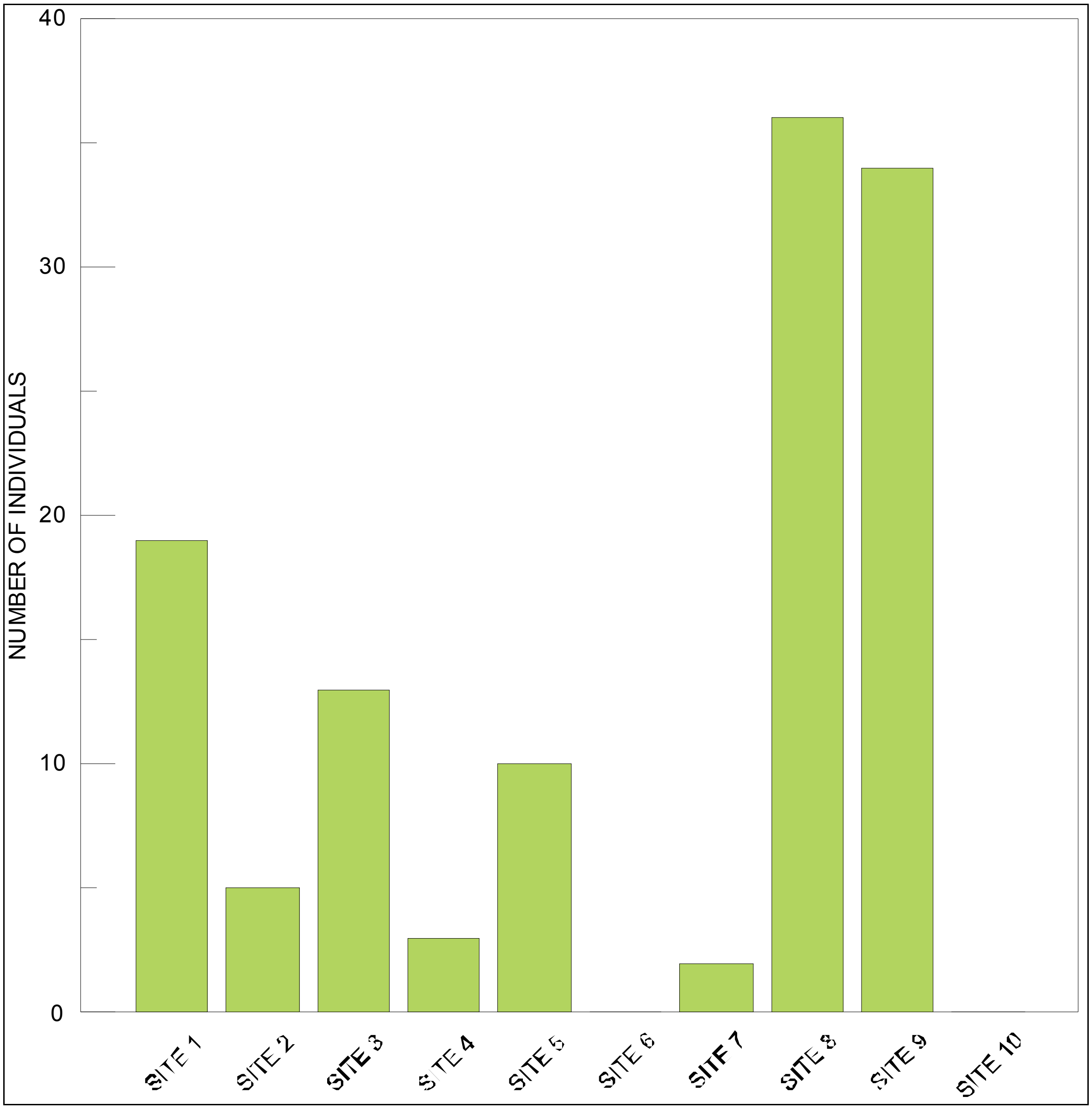


Table 3.2.3. Pearson Correlation Coefficients. Correlation is significant at the 0.05 level*.

Statistic	Species Richness	Abundance	Litter Temp Range
Pearson Correlation	0.664	0.709	-0.855
p-value	0.036*	0.022*	0.001*
N	10	10	10

Figure 3.2.6. Salamander abundance compared with HGM Habitat FCI score. Correlation coefficient is based on square root abundance.

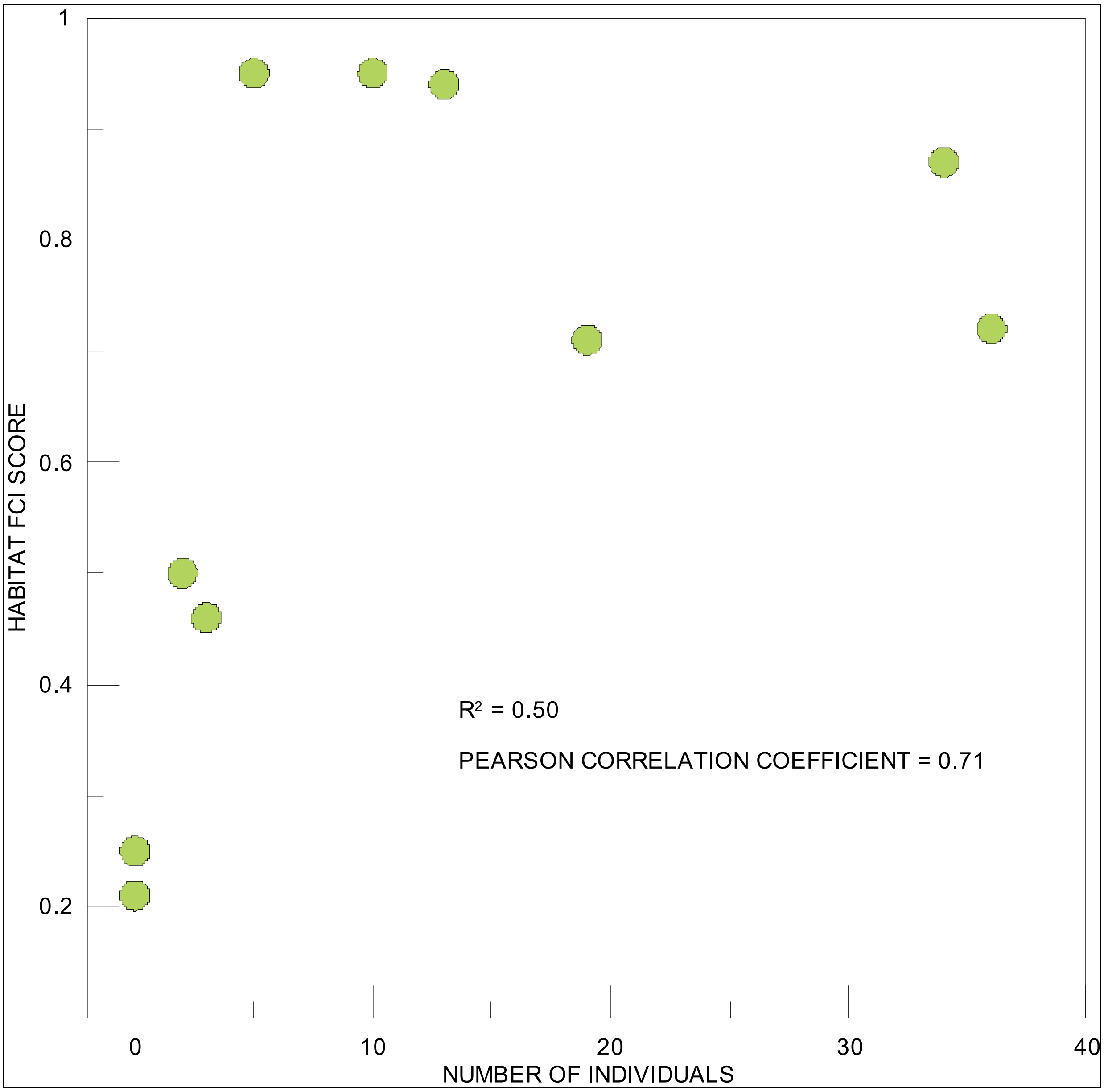


Figure 3.2.7. Mean daily temperature range (°C) under leaf litter at each sample location. Error bars represent one standard deviation.

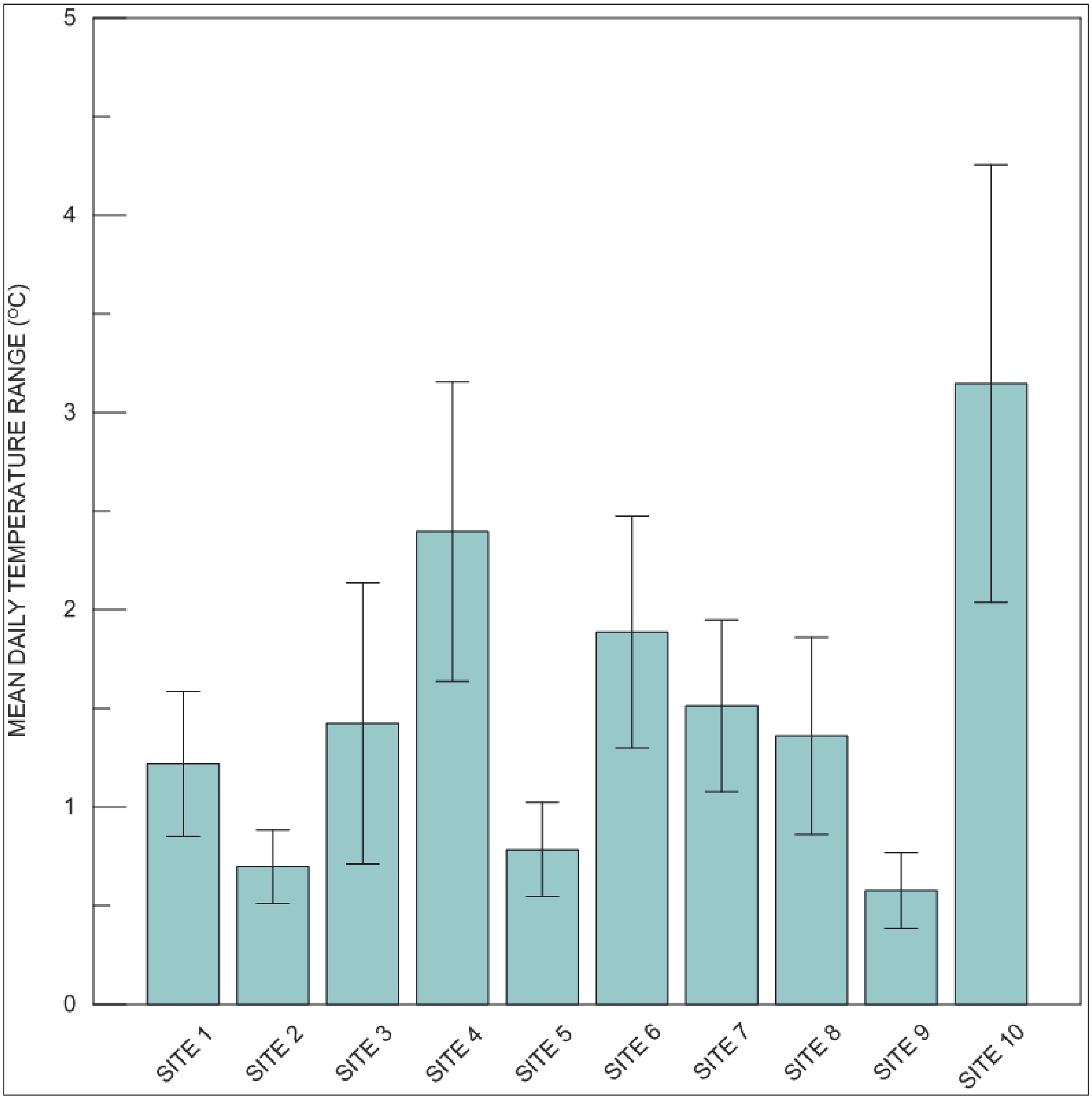
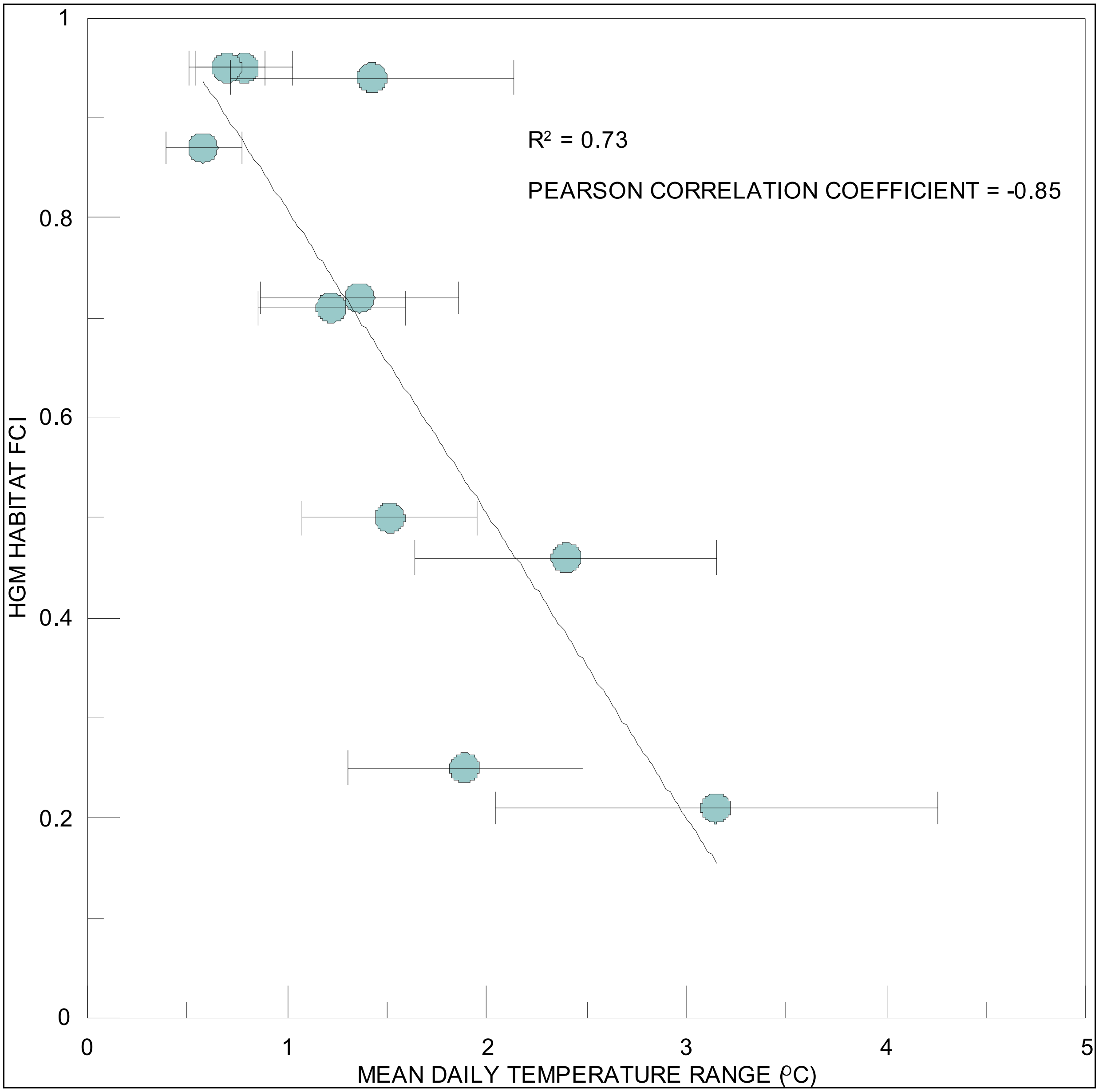


Figure 3.2.8. Mean daily temperature range (°C) under leaf litter compared with Habitat FCI score. Error bars represent one standard deviation.



Sites with the lowest daily temperature range were characterized by mature second growth forested watersheds displaying high canopy cover and leaf litter cover. These results are consistent with previous findings, which have shown an increase in the amplitude of temperature fluctuations following a decrease in forest cover (Brown and Krygier 1970; Childs et al. 1985; Spittlehouse and Stathers 1990).

3.2.4 Summary

The collected amphibian community data provides evidence that the HGM Habitat model discriminated between locations displaying a range of watershed, channel, and riparian/buffer zone characteristics. Salamander species richness and abundance estimates were positively correlated with Habitat FCI score. Study sites with high species richness and abundance exhibited unaltered channel characteristics and forested watersheds with mature second growth forest structure and high percent canopy cover. The lowest species richness and abundance was observed at study sites exhibiting alterations. Few or no salamanders occurred at sites displaying watershed alterations due to surface mining or agricultural grazing.

3.3 Invertebrates

3.3.1 Rationale for selecting the independent measure

Aquatic macroinvertebrate communities are used commonly as indicators of ecological condition in perennial streams (e.g., Cummins and Klug 1979; Kerans and Karr 1994; Lenat 1993, Plafkin et al. 1989; Pond et al. 2003; Pond et al. 2012, Gerritsen et al. 2000). Similar to other biological indicators of ecosystem health, invertebrate communities reflect the cumulative effects of multiple stressors; thus, sampling invertebrates has potential advantages over measuring physical or chemical parameters (Hodkinson and Jackson 2005). Invertebrates are relatively easy to sample, and possess characteristics such as gills and limited dispersal ability which make many taxa highly susceptible to stressors, including changes in water quality and land use (Delong and Brusven 1998; Kerans and Karr 1994; Lenat and Crawford 1994; Shieh et al. 1999; Snyder et al. 2003). Aquatic macroinvertebrate communities are composed of multifarious species possessing varying life history traits and differing tolerances to habitat conditions (Figure 3.3.1); thus, metrics relating to community composition can be derived from the species assemblage present at a particular site. Community metrics (e.g., abundance, taxa richness, and relative proportions of functional feeding groups) that vary along a gradient of human alteration provide insight into relative ecological function. For example, a number of researchers have observed a decline in invertebrate taxa richness with increasing alteration (Ohio EPA 1988; Kerans and Karr 1994; DeShon 1995).

Figure 3.3.1. Examples of benthic macroinvertebrate genera found at sites (clockwise from top left): 1) mayfly in the family Ameletidae (photo by David Funk), 2) stonefly in the family Nemouridae (photo by Tom Murray), 3) caddisfly in the family Lepidostomatidae (photo by Tom Murray), and 4) caddisfly in the family Phryganeidae. Photos are for example only.

3.3.2 Methods

Invertebrate sampling occurred both above-ground and in the stream bed utilizing basket samplers. Commonly used sampling methodologies requiring flowing water, such as kicknets or Surber samplers, cannot be used in ephemeral streams due to shallow water depths and because invertebrates often retreat to the benthic zone during dry periods. The use of basket samplers containing artificial substrate is an approach used to sample benthic invertebrates within a standardized area, and is adaptable to use in ephemeral streams because it can be used in extremely shallow water. Basket sampling is also not inhibited by problematic substrates, such as those found in boulder-stabilized stream channels, and areas subjected to valley fill (Dickson et al. 1971; Hilsenhoff 1969; Reice 1980; Weisburg et al. 1990).

The same basket samplers used for salamander sampling and described in Section 3.2.2 were used for invertebrate sampling, with four replicate baskets installed at each site during October 2011, January 2012, and April 2012. Wash water was filtered through a #30 (0.6-mm mesh) sieve. All aquatic invertebrates found were preserved in 95 percent ethyl alcohol, and were later picked from the debris (Figure 3.3.2). Specimens were sent to the EPA Region III Freshwater Biology Lab and identified to genus or the lowest taxonomic level possible.

Figure 3.3.2: Basket samplers were placed within the stream channel for one month to sample benthic macroinvertebrates (clockwise from top left): 1) basket sampler in stream, 2) emptying basket sampler into bucket for rinsing and sorting, 3) transferring invertebrates and small material into a jar containing ethanol after removal of cobble and large leaves, and 4) sorting invertebrates and preserving in ethanol for identification.

Invertebrate community metrics selected for analysis were based upon existing scientific literature and include four common community composition variables which reflect changes in stream and watershed condition (Barbour et al. 1999, Kerans and Karr 1994, Pond et al. 2003, Gerritsen et al. 2000, Wilton 2004). Taxa richness is the total number of aquatic macroinvertebrate taxa, based on the lowest identifiable taxonomic level (usually genus). EPT richness is the total number of taxa detected that belong to the orders Ephemeroptera, Plecoptera or Tricoptera. Intolerant richness is the number of taxa, based on the lowest identifiable taxonomic level, known to be disturbance intolerant. Percent Chironomids is the percentage of all individuals detected that belong to the family Chironomidae. Taxa richness, EPT richness, and intolerant taxa richness tend to decrease as stream alteration increases, while percent Chironomids tends to increase as stream alteration intensifies (Kerans and Karr 1994, Pond et al. 2003). Invertebrate data from the four basket samplers at a site were combined during each sampling period, and metric values were averaged across all sampling periods. Statistical procedures included normality testing using the Shapiro-Wilk test ($p>0.05$ indicates a normal distribution), followed by comparison of HGM Habitat FCI values with community metrics for each site using Pearson Product Moment Correlation analysis ($\alpha = 0.05$; JMP, SAS Institute 2012).

3.3.3 Results

Of the four community metrics tested, no single invertebrate community metric significantly correlated with the Habitat FCI score (results presented below). As a result, an analysis was conducted examining the departure of community metrics from the range observed at the four least-altered sites, which included study sites containing mature forest structure (e.g., high percent canopy cover, large tree diameter, large amounts of woody debris, high percent cover of detritus) and unaltered, stable channel characteristics. Although the least-altered study sites were not selected on the basis of FCI scores, the four least-altered study sites selected all exhibited Habitat FCI scores over 0.8 (Table 2.1). All five of the most highly altered sites exhibited one or more invertebrate community metrics outside of the least-altered range (Table 3.3.1). Site 8, which exhibited mature second-growth forest despite silvicultural alterations, displayed invertebrate community metrics within the range observed at least-altered study sites. These results affirm the ability of the HGM Habitat model to differentiate between a range of alteration intensities, which supports the current assessment model.

Mean taxa richness at sites ranged from 10.0 – 44.7 taxa (Figure 3.3.3). Data were normally distributed (Table 3.3.2). Taxa richness was not significantly correlated with HGM Habitat FCI values ($r = 0.45$, $n = 10$, $P = 0.19$; Figure 3.3.4, Table 3.3.3). However, a subset of altered study sites deviated from the range of values observed within the least-altered sites

(>20 taxa), providing evidence that the HGM model differentiates between least-altered stream channels with forested watersheds and altered sites (Table 3.3.1).

Table 3.3.1. Invertebrate community metrics which fall outside the range of values observed at least-altered study sites (least-altered sites indicated by asterisk).

Site	Habitat FCI Score	Impact	Invertebrate Community Metric			
			Taxa Richness	EPT Richness	Intolerant richness	% Chironomids
1	0.71	Silviculture,Recreation	X	X	X	
2*	0.95	Least-altered				
3*	0.94	Least-altered				
4	0.46	Surface mining	X	X	X	
5*	0.95	Least-altered				
6	0.25	Surface mining	X			
7	0.5	Grazing		X		
8	0.72	Silviculture				
9*	0.87	Least-altered				
10	0.21	Surface mining		X	X	X

Mean EPT richness at sites ranged from 0.7 – 12.3 taxa (Figure 3.3.5). Data were normally distributed (Table 3.3.2). EPT richness was not significantly correlated with HGM Habitat FCI values (r = 0.53, n = 10, P = 0.12; Figure 3.3.6, Table 3.3.3). However, a subset of altered study sites deviated from the range of values observed within the least-altered sites (>3 EPT taxa), providing evidence that the HGM model differentiates between least-altered stream channels with forested watersheds and altered study sites (Table 3.3.1).

Mean intolerant taxa richness at sites ranged from 0.7 – 14.3 (Figure 3.3.7). Data were normally distributed (Table 3.3.2). Percent intolerant taxa was not significantly correlated with HGM Habitat FCI values (r = 0.47, n = 10, P = 0.17; Figure 3.3.8, Table 3.3.3). However, a subset of altered study sites deviated from the range of values observed within the least-altered sites ($\geq$3 intolerant taxa), providing evidence that the HGM model differentiates between least-altered stream channels with forested watersheds and altered study sites (Table 3.3.1).

Mean percent Chironomids at sites ranged from 8.5 percent – 82.9 percent (Figure 3.3.9). Data were normally distributed following square-root transformation (Table 3.3.2). Chironomid taxa richness was not significantly correlated with HGM Habitat FCI values ($r = 0.24$, n = 10, $P = 0.50$; Figure 3.3.10, Table 3.3.3). However, one altered study site deviated from the range of values observed within the least-altered sites (>57 percent Chironomids), providing evidence that the HGM model differentiates between least-altered stream channels with forested watersheds and altered study sites (Table 3.3.1).

Figure 3.3.3. Mean aquatic invertebrate taxa richness at each sample location. Error bars represent one standard deviation.

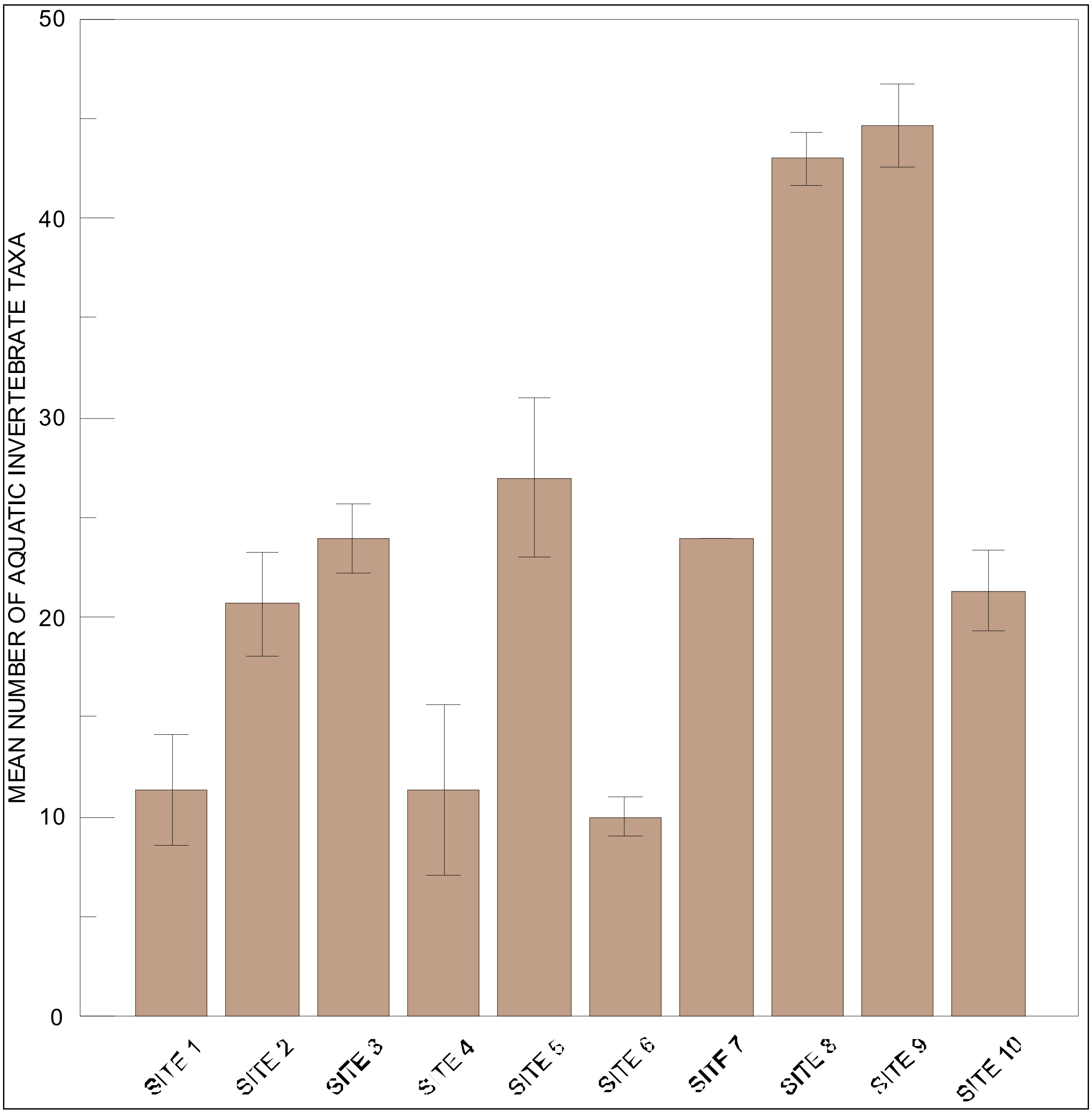


Table 3.3.2. Tests of Normality – Invertebrate Community Metrics

Parameter	Shapiro-Wilk		
	W	Df	*P*-value
Taxa Richness	0.87	10	0.12
EPT Richness	0.87	10	0.09
Intolerant Richness	0.85	10	0.06
sqrt % Chironomids	0.92	10	0.40

Figure 3.3.4. Mean aquatic invertebrate taxa richness compared with HGM Habitat FCI score. The dashed line represents the lowest value observed at least-altered sites. Error bars represent one standard deviation.

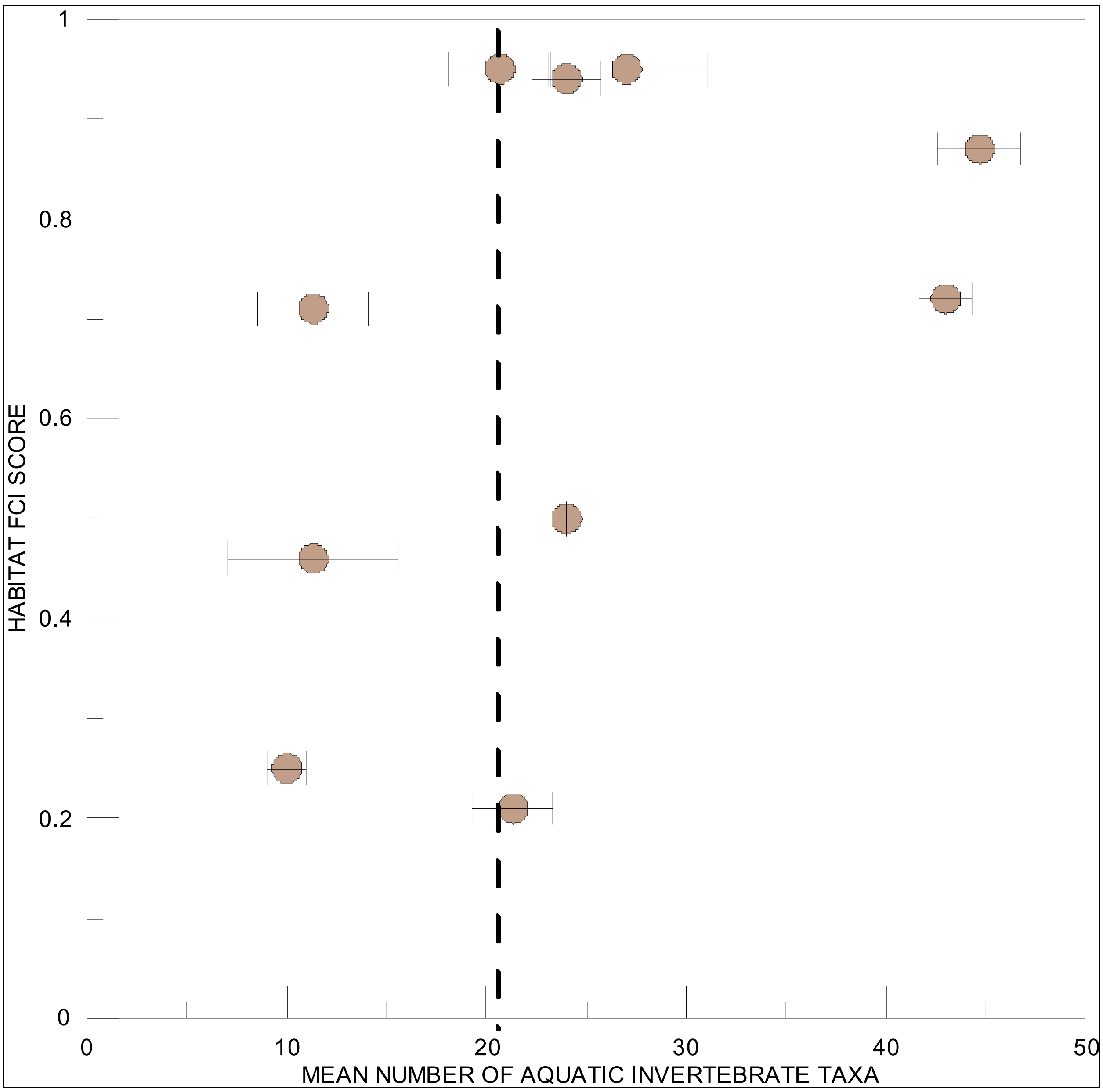


Table 3.3.3. Pearson Correlation Coefficients.

Predictor	Pearson Correlation	*P*-value	N
Taxa Richness	0.45	0.19	10
EPT Richness	0.53	0.11	10
Intolerant Richness	0.47	0.17	10
Sqrt % Chironomids	0.24	0.5	10

Figure 3.3.5. Mean Ephemeroptera, Plecoptera and Tricoptera taxa richness at each sample location. Error bars represent one standard deviation.

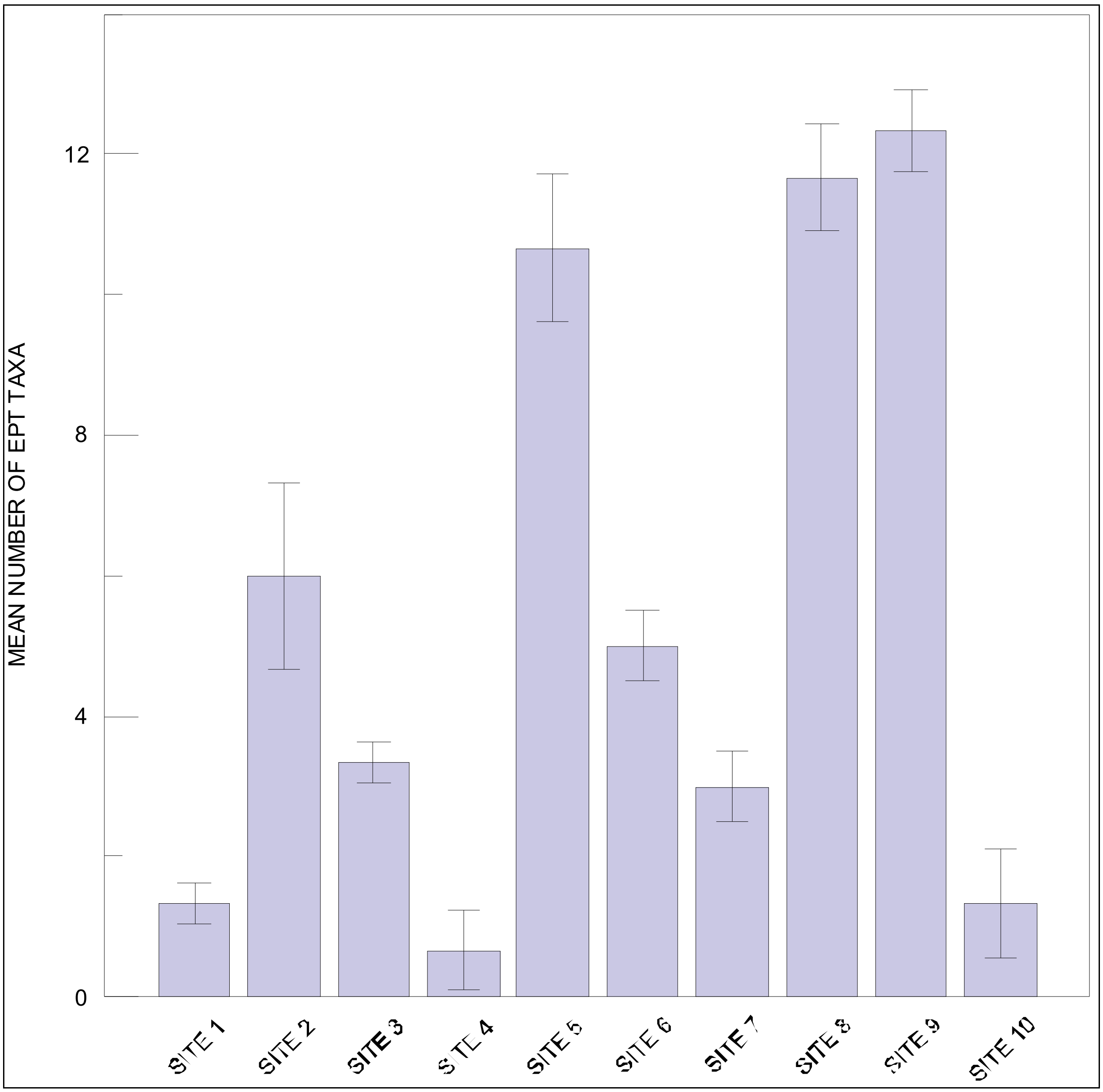


Figure 3.3.6. Mean Ephemeroptera, Plecoptera and Tricoptera taxa richness compared with Habitat FCI scores. The dashed line represents the lowest value observed at least-altered sites. Error bars represent one standard deviation.

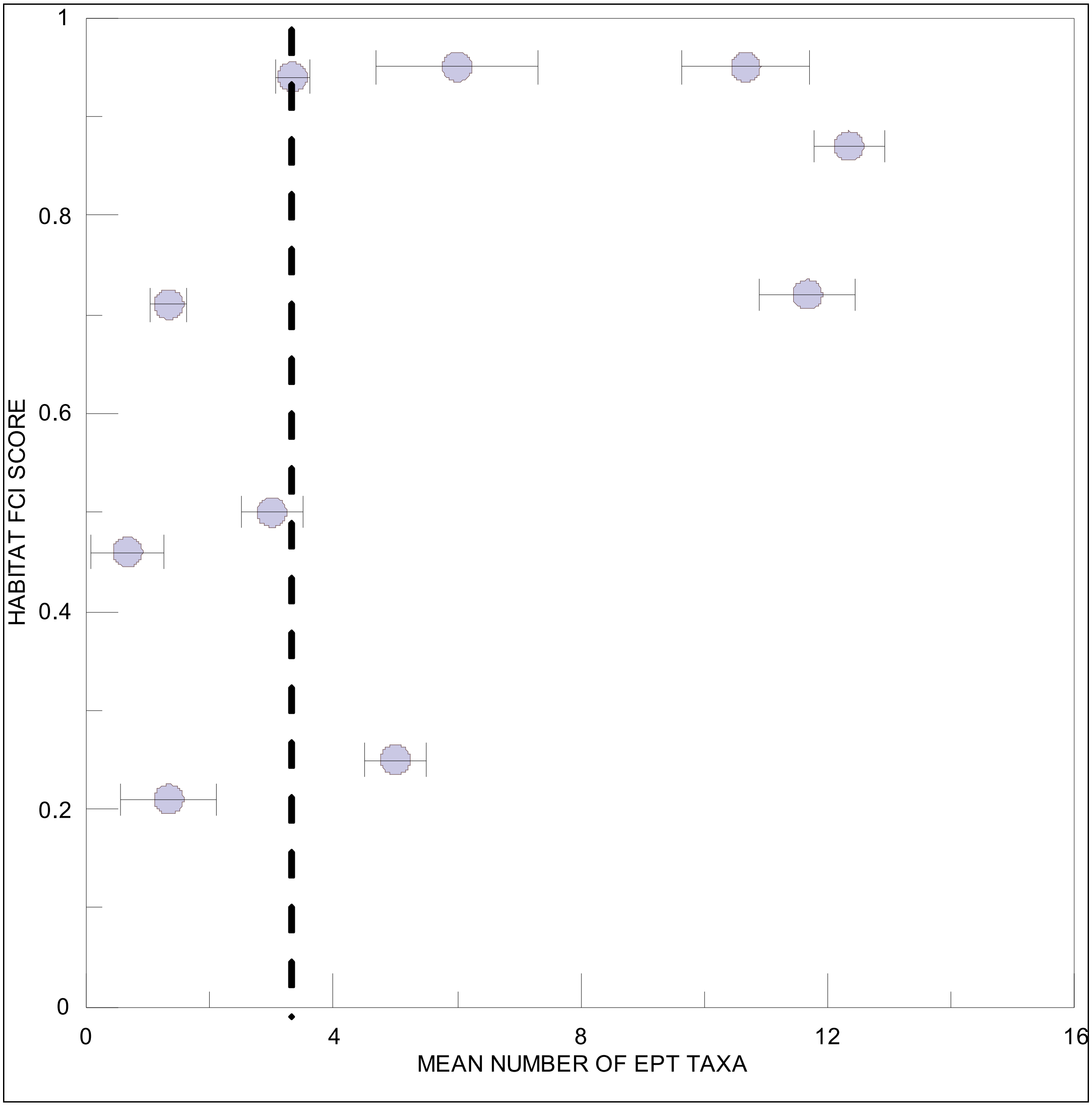


Figure 3.3.7. Mean number of disturbance intolerant taxa at each sample location. Error bars represent one standard deviation.

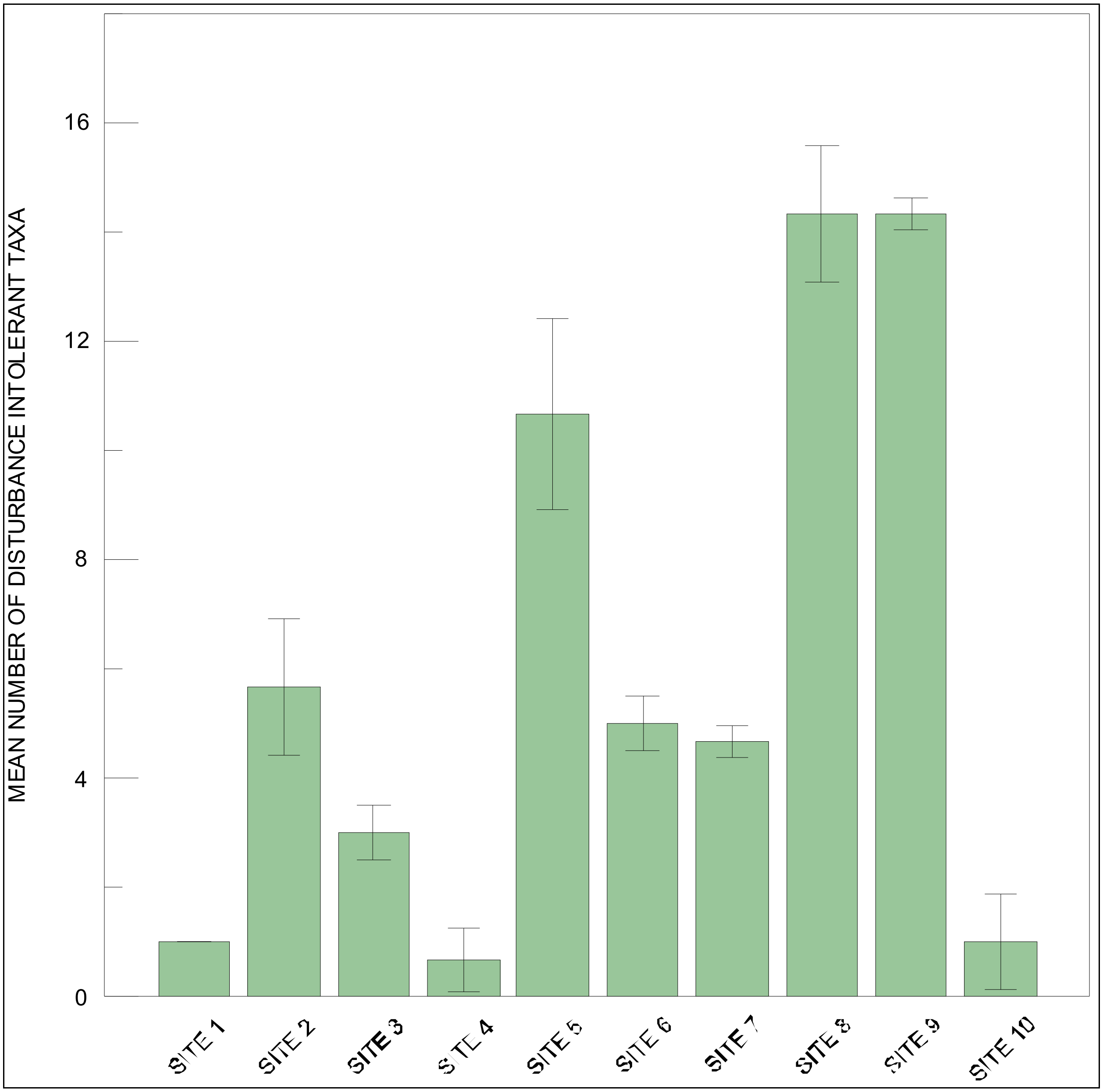


Figure 3.3.8. Mean number of disturbance intolerant taxa compared with Habitat FCI scores. The dashed line represents the lowest value observed at least-altered sites. Error bars represent one standard deviation.

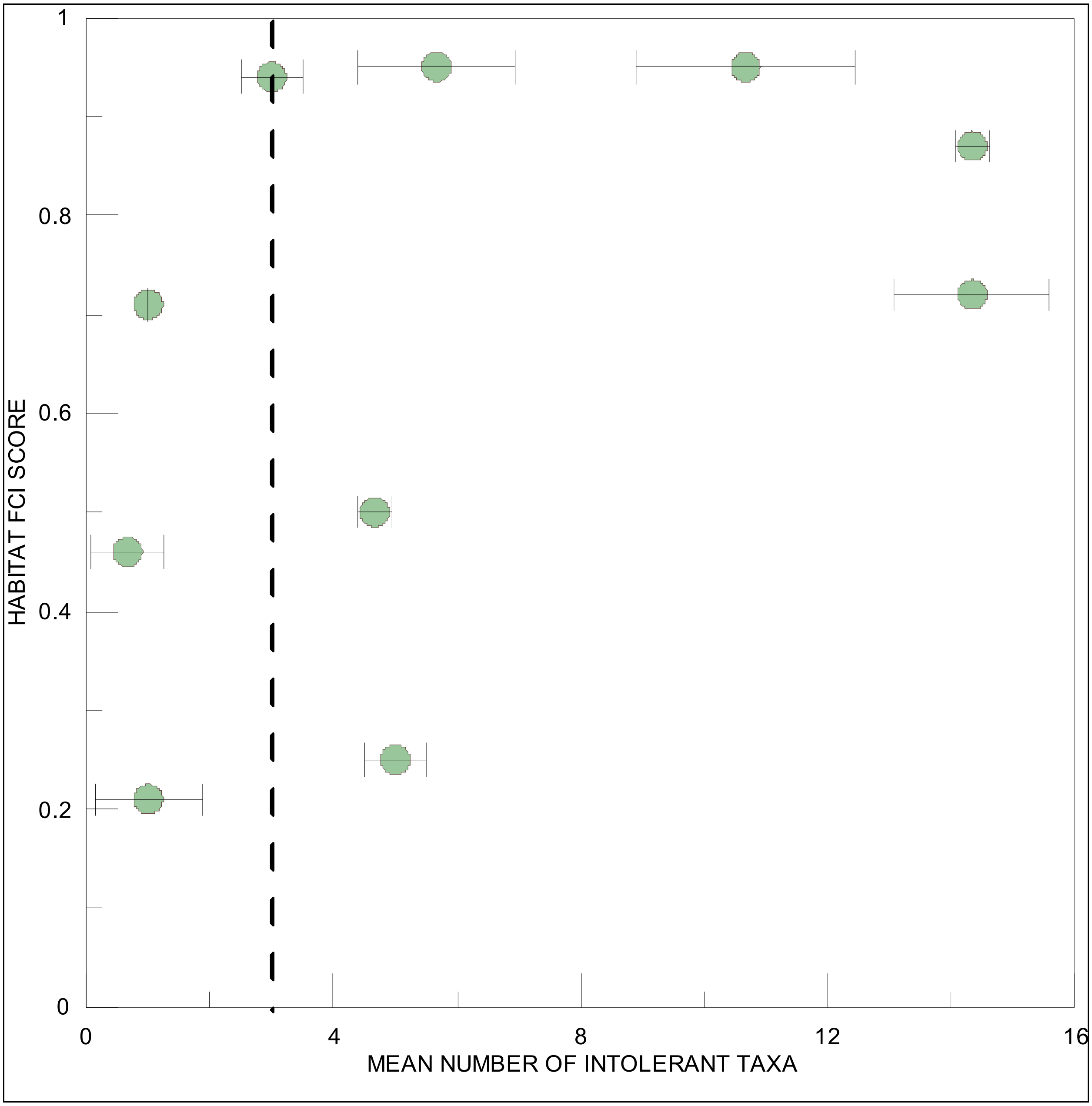


Figure 3.3.9. Mean percent Chironomids at each sample location. Error bars represent one standard deviation.

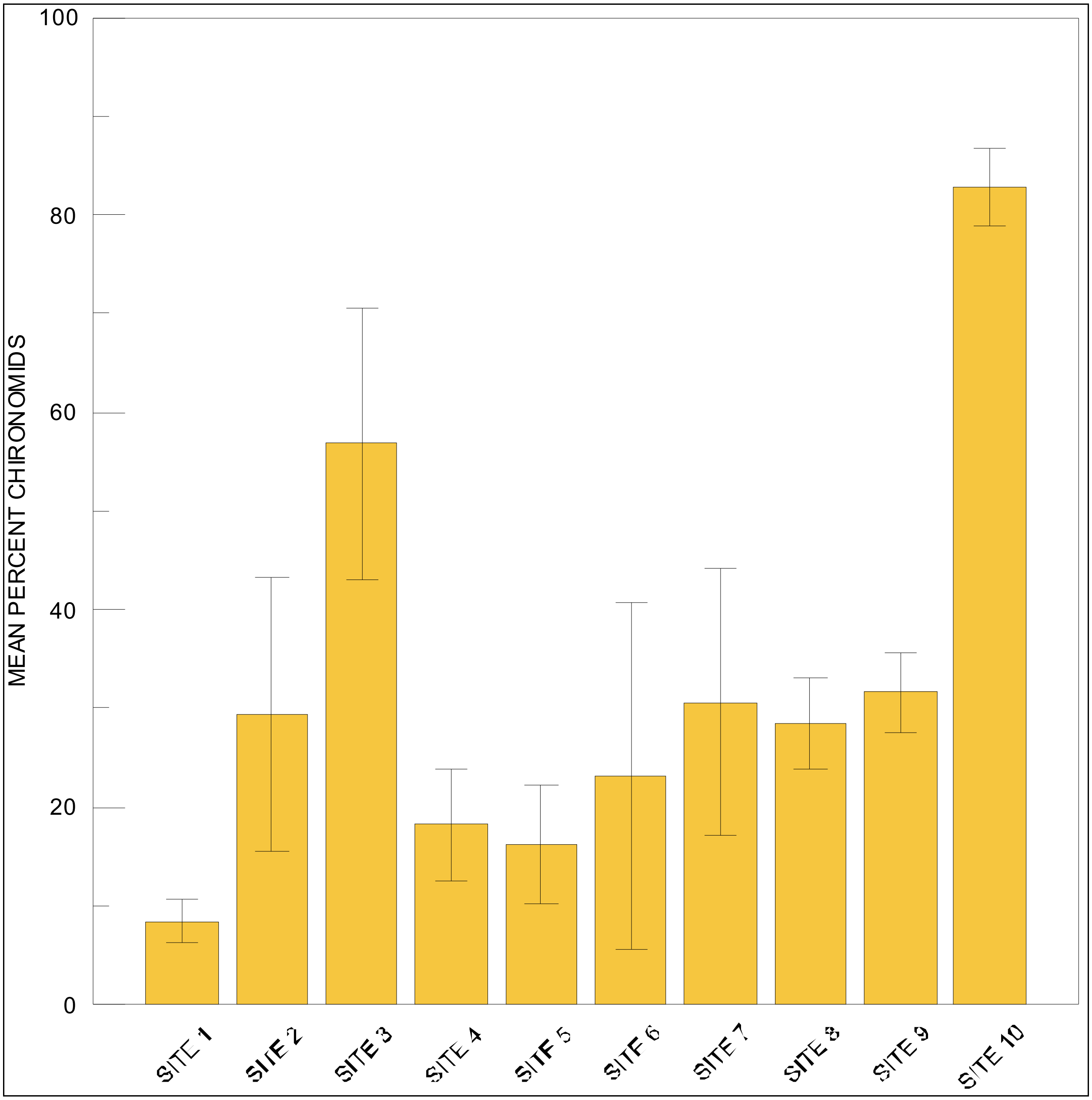
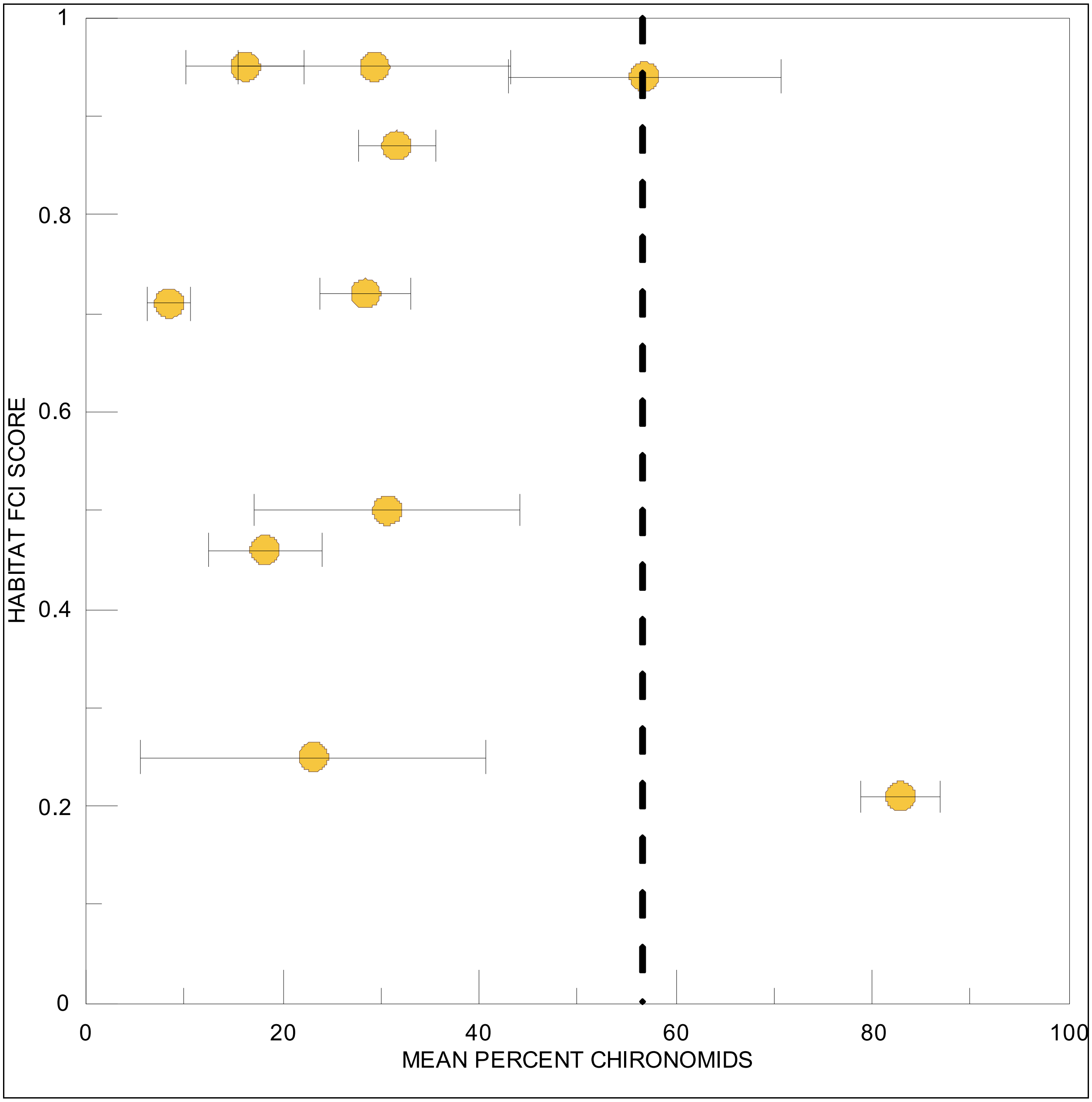


Figure 3.3.10. Mean percent Chironomids compared with Habitat FCI score. The dashed line represents the highest value observed at least-altered sites. Error bars represent one standard deviation.



3.3.4 Summary

At the five study sites exhibiting the lowest HGM Habitat FCI scores, at least one of the four community metrics examined departed from the range observed at least-altered sites; thus, lack of significant correlation between metrics and HGM outputs does not negate model validity. However, results demonstrate the challenge of utilizing invertebrate metrics as indicators of headwater stream condition. Invertebrate assessments, while widely employed in perennial stream assessments, are seldom used in ephemeral and intermittent streams for a variety of reasons. The invertebrate community in streams with temporary hydrologic regimes is generally limited to those taxa that can complete their life cycles rapidly (Dieterich 1992; Ohio EPA 2002; Bonada et al. 2007), thus, invertebrate community metrics may provide less information than in perennial streams with more rich invertebrate taxa assemblages. Additionally, comparisons of invertebrate metrics across headwater streams may provide limited utility due to the effect of variable water levels on invertebrate community composition. Several studies show higher invertebrate taxa diversity and/or richness in perennial streams than in intermittent streams (Feminella 1996; Fritz and Dodds 2005; Meyer and Meyer 2000; Pond et al. 2003; del Rosario and Resh 2000; Williams 1996; Wood et al. 2005; Wright et al. 1984). Lower invertebrate diversity and abundance estimates have been documented in headwater streams compared to streams with longer hydroperiods (Dieterich 1992; Halwas et al. 2005; Price et al. 2003). The lack of a significant relationship between invertebrate community composition and a gradient of alteration at study sites may indicate that commonly used invertebrate metrics are not ideal measures of ecosystem functioning for ephemeral and intermittent streams.

3.4 Riparian vegetation

3.4.1 Rationale for selecting the independent measure

Riparian vegetation composition is a key factor in determining the ability of a stream ecosystem to provide wildlife habitat. Vegetation species composition is a reflection of disturbance regime, and changes in vegetation composition often provide the most easily measurable evidence of degradation (White 1979). Vegetation is intricately tied to hydrology and soil processes, and changes to plant community composition, such as the introduction of exotic plant species, directly affect ecosystem functions by altering soil microbial community structure (Kourtev et al. 2002), nutrient cycling (Belnap and Phillips 2001; Ehrenfeld et al. 2001; Evans et al. 2001; Mack et al. 2001; Scott et al. 2001), hydrology (Dyer and Rice 1999; Zavaleta 2000) and fire regimes (D'Antonio and Vitousek 1992). Headwater streams are influenced by the processes occurring within the riparian zone, and near-stream vegetation directly influences functioning within the stream channel. For example, leaf litter fall represents the primary input of coarse particulate matter and nutrients to streams, influencing the availability of food and refugia for in-stream organisms (Webster and Benfield 1986, Allan 1995, Gessner et al. 1999).

Vegetation inventories are used commonly as measures of ecological function, because they can be performed rapidly and provide insight into multiple ecological functions. Floristic Quality Assessment (FQA) is a method of assessing plant community composition based on the tolerance of each plant taxon to alteration and habitat degradation, as well as its level of fidelity to a particular habitat type (Rentch and Anderson 2006). In practice, an FQA is performed by conducting a complete vegetative inventory of a site, then comparing each species detected with a published Coefficient of Conservatism (C) value ranging from 0 to 10. Species with high floristic quality receive a score of 10, while non-native species automatically acquire a score of 0. The final score for each site is the Floristic Quality Index (FQI), which is the average of all C values for a site. FQIs have been shown to accurately distinguish between levels of alteration and correlate with soil chemistry features such as soil total organic carbon, phosphorus and calcium (Lopez and Fennessy 2002). While the HGM model requires documentation of the presence or absence of a limited number of species, an FQI represents a survey of all plant species resulting from a comprehensive inventory of all plants detected at a study site. As a result, FQIs were selected as appropriate independent measures of the HGM Habitat function.

3.4.2 Methods

Vegetation inventories were conducted twice at each site, during October 10 – 14, 2011 and April 23 – 27, 2012. At each site, a plot was delineated, extending the entire length of the validation study reach (100 feet) and 25 feet from the stream on each side. Each plot was traversed on foot several times. All vascular plant species observed within the plots were identified to species to the extent possible at the time of the site visit. Using the West Virginia working FQA, an FQI score was calculated for each site (Table 3.4.1; West Virginia Natural Heritage Program 2012). Normality was tested using the Shapiro-Wilk test ($p>0.05$ indicates a normal distribution) and HGM Habitat FCI scores were compared with FQI values for each site using Pearson Product Moment Correlation analysis ($\alpha = 0.05$; JMP, SAS Institute 2012).

3.4.3 Results

A total of 405 plant species were identified across all sites, with a total of 67 – 125 species occurring within an individual site. FQI scores at study sites ranged from 1.9 – 5.8, and displayed a normal distribution (Figure 3.4.1, Table 3.4.2).

Table 3.4.1. Example C values for the most commonly observed species at study sites.

Scientific Name	Common Name	C value
Liriodendron tulipifera	tuliptree	5
Polystichum acrostichoides	Christmas fern	5
Carex laxiflora	broad looseflower sedge	5
Galium triflorum	fragrant bedstraw	5
Acer rubrum	red maple	3
Prunus serotina	black cherry	3
Rosa multiflora	multiflora rose	0
Solidago caesia	wreath goldenrod	6
Carex digitalis var. digitalis	slender woodland sedge	6
Dioscorea villosa	wild yam	5
Eurybia divaricata	white wood aster	5
Fagus grandifolia	beech	6
Lindera benzoin	northern spicebush	5
Platanus occidentalis	American sycamore	5
Polygonum cespitosum	Oriental lady's thumb	0
Stellaria media	common chickweed	0
Symphyotrichum lateriflorum	calico aster	3
Toxicodendron radicans	poison ivy	1
Ageratina altissima	white snakeroot	3
Betula lenta	sweet birch	5
Circaea lutetiana subsp. Canadensis	broadleaf enchanter's nightshade	5
Cornus florida	flowering dogwood	5
Fraxinus Americana	white ash	5
Galium aparine	stickywilly	2
Hydrangea arborescens	wild hydrangea	6
Maianthemum racemosum	false Solomon's seal	5
Oxydendrum arboretum	sourwood	5
Parthenocissus quinquefolia	Virginia creeper	3
Podophyllum peltatum	mayapple	5
Polygonum virginianum	jumpseed	5
Prenanthes altissima	tall rattlesnakeroot	6
Quercus rubra	northern red oak	6
Ranunculus recurvatus	blisterwort	4
Smilax rotundifolia	roundleaf greenbrier	2
Verbesina alternifolia	yellow crownbeard	2

Figure 3.4.1. Floristic Quality Assessment values at each sample location.

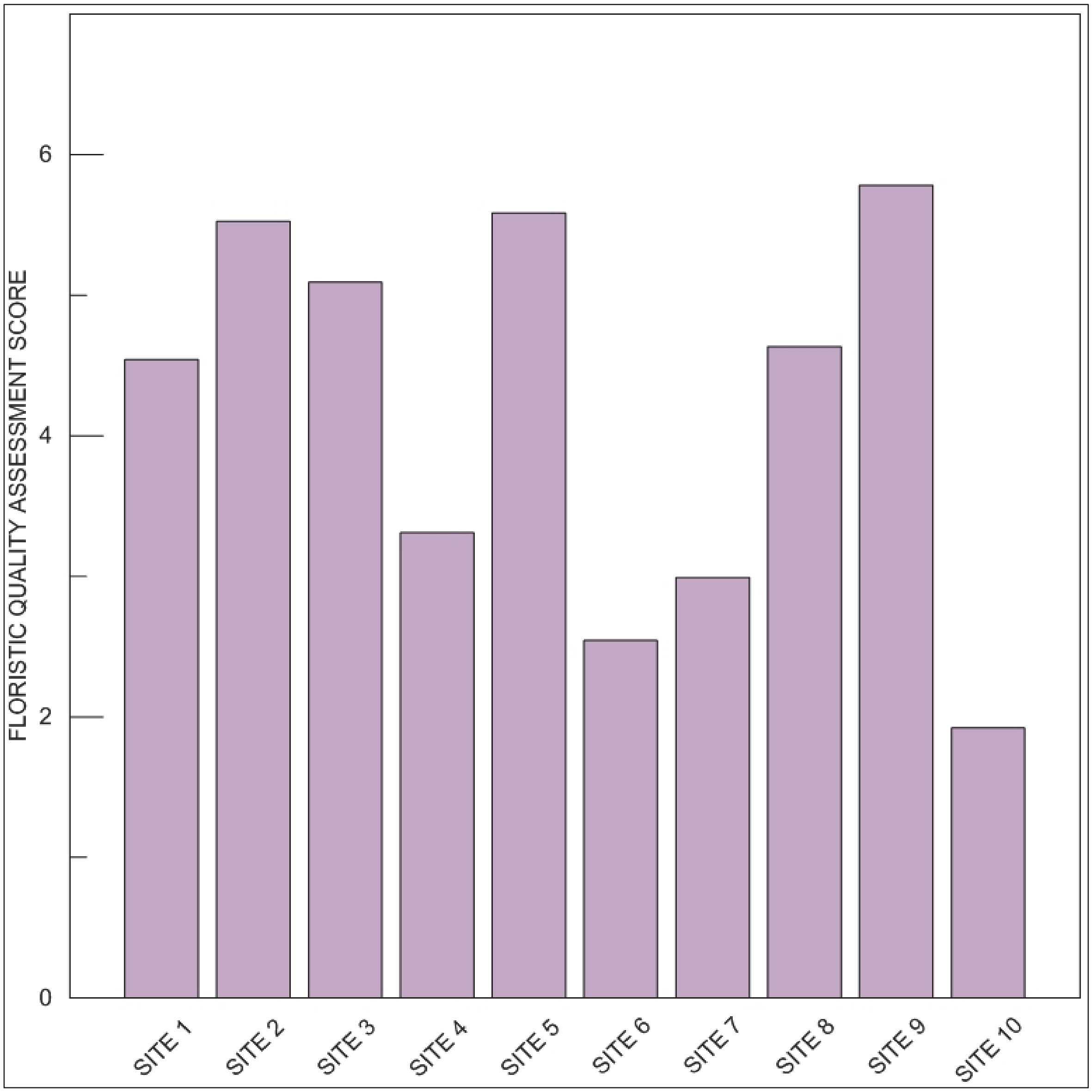
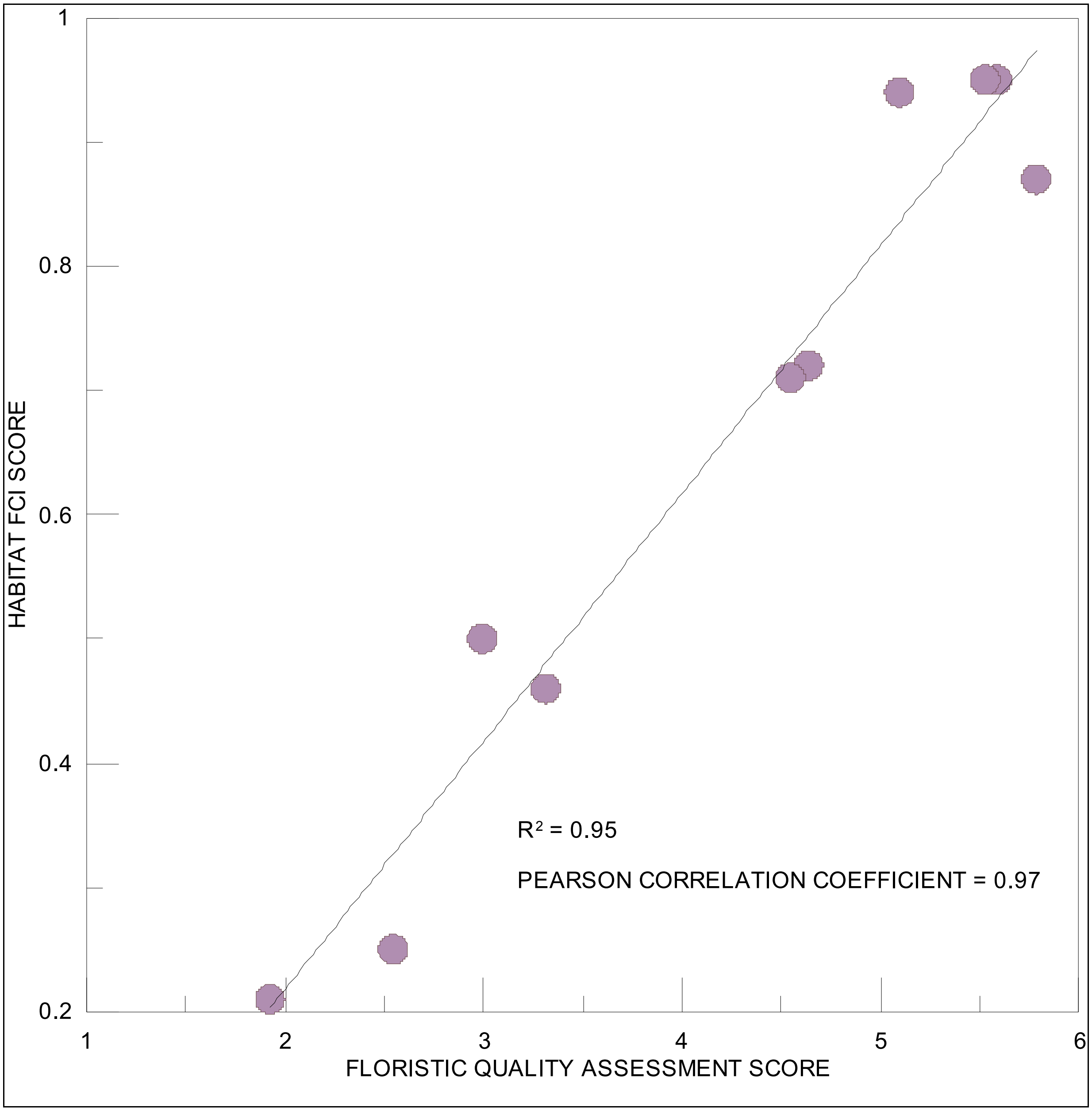


Table 3.4.2. Statistical Tests – Floristic Quality Index.

Statistical Procedure	W	N	*P*-value
Shapiro-Wilk Test	0.91	10	0.26
Pearson Correlation	0.97	10	<0.0001

Comparison of Floristic Quality Assessment values with the HGM Habitat FCI scores yielded a strong positive correlation ($r = 0.975$, $n = 10$, $P < 0.0001$, Figure 3.4.2), supporting the Habitat FCI model. All six study sites displaying HGM Habitat FCI scores <0.75 obtained FQI scores <5.0. In contrast, the four least-altered sites all exhibited FQI scores >5.0.

Figure 3.4.2. Floristic Quality Assessment values compared with HGM Habitat FCI score.



3.4.4 Summary

Floristic Quality Assessment data provides evidence that the HGM Habitat model differentiated between sample locations displaying a range of watershed, channel, and riparian/buffer zone characteristics. Study sites which obtained high FQI values displayed watershed conditions at or near the reference standard range, such as mature forest structure and lack of significant watershed alterations. Conversely, sites exhibiting low FQI values contained sites affected by ongoing agricultural impacts, past surface mining effects, and timber harvesting.

3.5 Habitat function sensitivity analysis

Sensitivity analysis results demonstrate that all variables incorporated into the Habitat model influence model outcomes as intended by model developers. Appendix A presents sensitivity testing results for all variables, and a representative subset is presented in this section. Examples and data analysis are provided for one variable displaying a strong influence over FCI scores and a variable displaying a weaker impact on FCI scores. For the Habitat function with canopy cover ≥20 percent (Equation 3.5.1), sensitivity analysis results demonstrate that percent canopy cover ($V_{CCANOPY}$), stream particle embeddedness (V_{EMBED}), and channel substrate size ($V_{SUBSTRATE}$) maintain the strongest effect of model FCI outputs (Figure 3.5.1, Table 3.5.1). For example, when $V_{CCANOPY}$ remains at the lowest allowable subindex score (0.1), the maximum attainable model FCI output equals 0.74 (Figure 3.5.2). When the $V_{CCANOPY}$ value reaches the maximum of 1.0, the range of potential FCI scores equals 0.09 – 1.0. Increasing the $V_{CCANOPY}$ score incrementally from 0.1 to 1.0 while holding all other variables constant results in a 35 – 216% increase in FCI scores. The selection and scaling of tree canopy cover as a controlling factor in the model is based on the importance of mature forest and vegetation structure for providing shade, refuge, and food to stream and riparian organisms. Substrate size and particle embeddedness were selected as controlling factors in the model due to their importance for determining the amount of in-stream habitat structure available to benthic macroinvertebrates and aquatic amphibians.

The remaining variables have a smaller effect on FCI values; however, each variable reacts appropriately across the range of potential model outcomes and in combination displays an appreciable impact on model results. For example, when V_{TDBH} remains at the minimum acceptable value of 0.1, potential FCI values range from 0.03 – 0.96 (Figure 3.5.3). When V_{TDBH} is maximized (subindex = 1.0), the observed range of values remains between 0.07 and 1.0. Increasing V_{TDBH} incrementally from 0.1 to 1.0 while holding all other variables constant results in a 4 – 135% increase in FCI scores. Achieving an FCI score of 1.0 requires all nine of the model variables to exhibit values equaling 1.0.

$$FCI = \left\langle \left[\frac{V_{CCANOPY} + \min\left(V_{EMBED}, V_{SUBSTRATE}\right)}{2} \right] \times \left\{ \frac{\left(\frac{V_{LWD} + V_{DETRITUS}}{2}\right) + \left[\frac{\left(\frac{V_{SNAG} + V_{TDBH} + V_{SRICH}}{3}\right) + V_{WLUSE}}{2} \right]}{2} \right\} \right\rangle^{1/2} \quad (3.5.1)$$

Figure 3.5.1. Conceptual distribution of variable influence on Habitat FCI values at sites with >20 percent canopy cover.

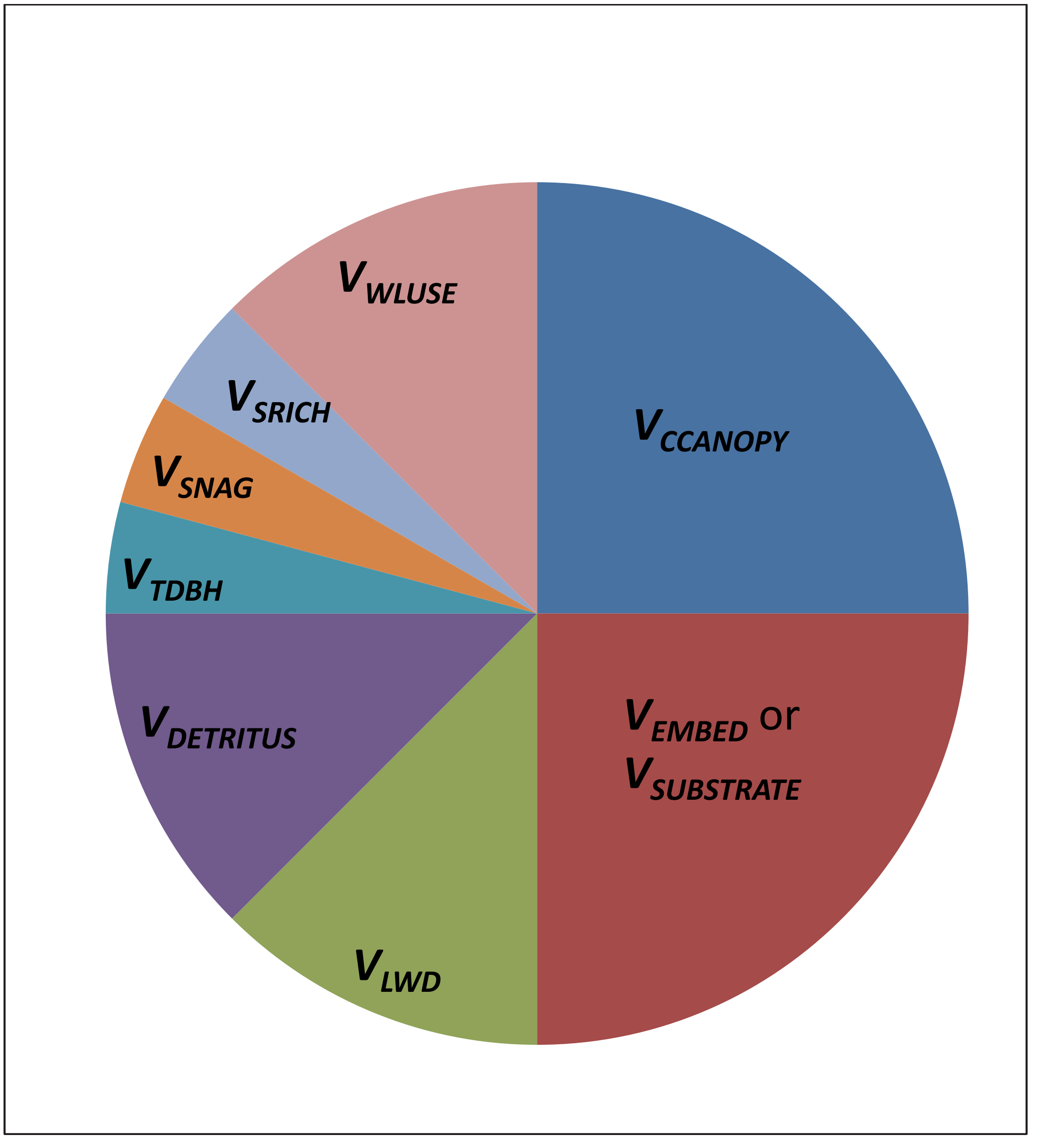


Table 3.5.1. Range of FCI scores attainable based on sensitivity analysis (≥20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.10, 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
$V_{CCANOPY}$	0.06	0.03	0.09	0.13	0.10	0.23	0.19	0.21	0.40
V_{EMBED}	0.00	0.03	0.03	0.00	0.10	0.10	0.04	0.21	0.25
$V_{SUBSTRATE}$	0.01	0.03	0.04	0.03	0.07	0.10	0.07	0.18	0.25
V_{LWD} / $V_{DETRITUS}$ / V_{WLUSE}	0.09	0.03	0.12	0.09	0.09	0.18	0.11	0.22	0.33
V_{TDBH} / V_{SNAG}	0.04	0.03	0.07	0.03	0.10	0.13	0.04	0.24	0.28
V_{SRICH}	0.04	0.03	0.07	0.04	0.10	0.13	0.04	0.24	0.28
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
$V_{CCANOPY}$	0.23	0.39	0.61	0.25	0.56	0.81	0.26	0.74	1.00
V_{EMBED}	0.11	0.39	0.50	0.19	0.56	0.75	0.26	0.74	1.00
$V_{SUBSTRATE}$	0.15	0.35	0.50	0.22	0.53	0.75	0.29	0.71	1.00
V_{LWD} / $V_{DETRITUS}$ / V_{WLUSE}	0.13	0.43	0.56	0.13	0.65	0.78	0.13	0.87	1.00
V_{TDBH} / V_{SNAG}	0.04	0.48	0.52	0.04	0.72	0.76	0.04	0.96	1.00
V_{SRICH}	0.04	0.48	0.52	0.04	0.72	0.76	0.04	0.96	1.00

When canopy cover is < 20 percent, the maximum attainable Habitat FCI score is restricted to 0.84. For the Habitat function with canopy cover <20 percent (Equation 3.5.2), sensitivity analysis results demonstrate that channel substrate size ($V_{SUBSTRATE}$) has the strongest effect on model FCI outputs (Figure 3.5.4, Table 3.5.2). When $V_{SUBSTRATE}$ remains at the lowest allowable subindex score (0.0), the maximum attainable model FCI output equals 0.0 (Figure 3.5.5). When $V_{SUBSTRATE}$ reaches a maximum value of 1.0, the range of potential FCI values equals 0.02 – 0.84. Increasing $V_{SUBSTRATE}$ incrementally from 0.1 to 1.0 while holding all other variables constant results in a 0 – 216% increase in FCI scores.

The remaining variables exhibit a smaller effect on FCI values; however, each variable reacts appropriately across the range of potential model outcomes and in combination maintain an appreciable effect on model outcomes. For example, when V_{SNAG} remains at the minimum allowable value of 0.1, possible FCI outputs range from 0.0 – 0.83 (Figure 3.5.6). When maximized (subindex = 1.0), the observed range of values equals 0.0 – 0.84. Increasing V_{SNAG} incrementally from 0.1 to 1.0 while holding all other variables constant results in a 0 – 12% increase in FCI scores. Achieving the maximum attainable score of 0.84 requires that all nine model variables equal 1.0.

Figure 3.5.2. Results of sensitivity test of $V_{CCANOPY}$ (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

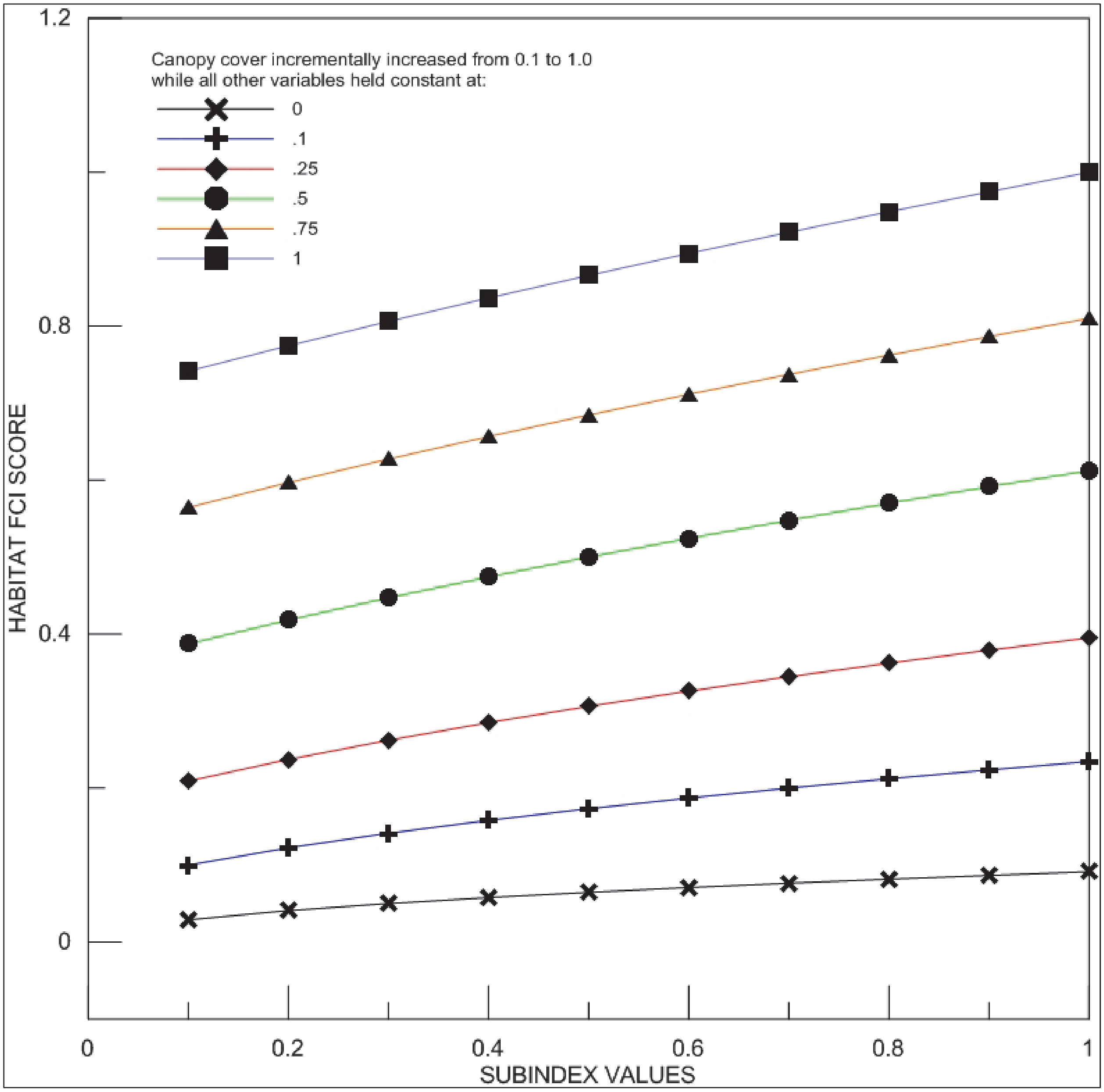
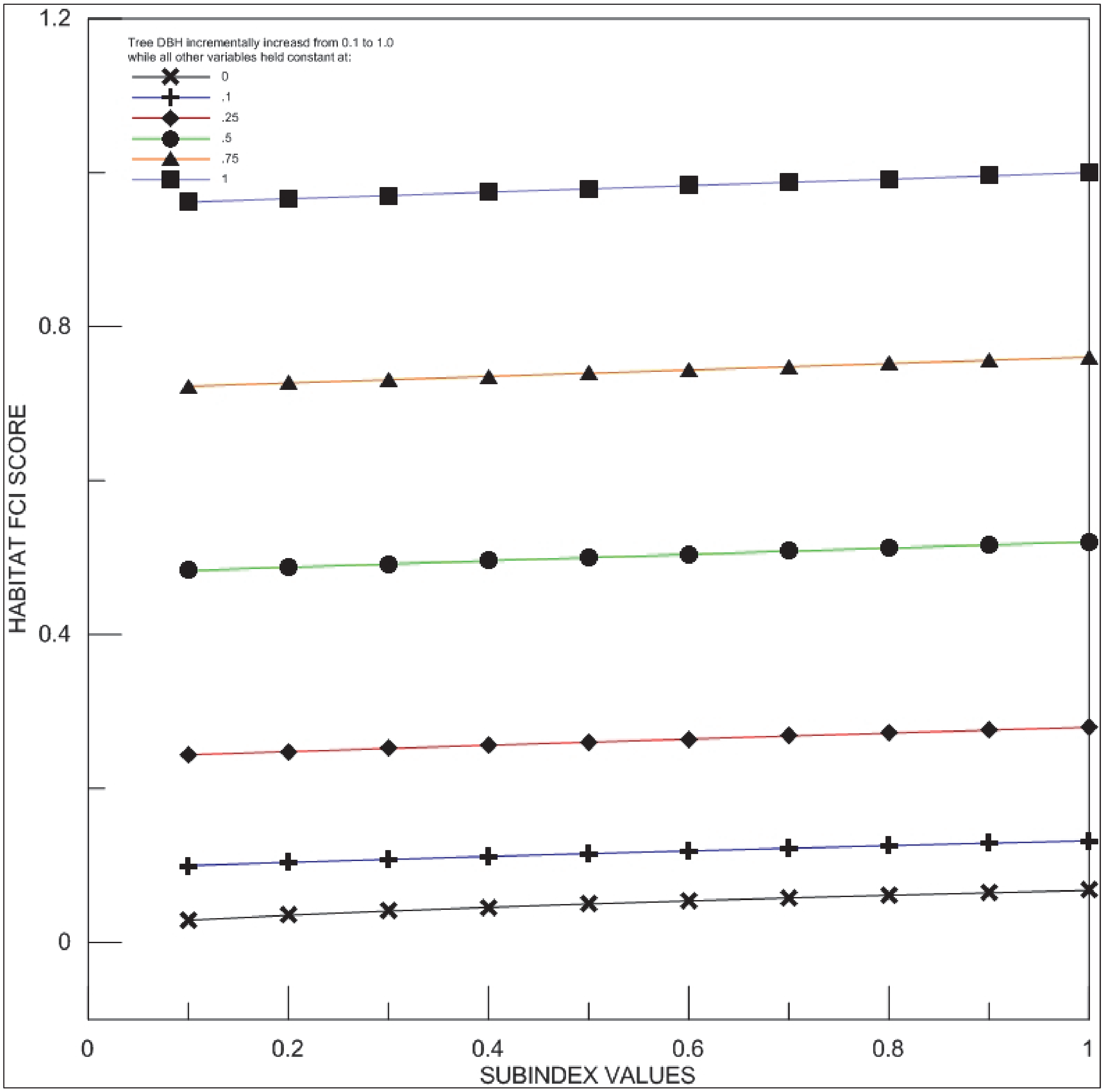


Figure 3.5.3. Results of sensitivity test of V_{TDBH} (canopy cover $\geq$20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0. Results for V_{SNAG} are identical.



$$FCI = \left\langle \min\left(V_{EMBED}, V_{SUBSTRATE}\right) \times \left\{ \frac{\left(\frac{V_{LWD} + V_{DETRITUS}}{2}\right) + \left[\frac{\left(\frac{V_{SNAG} + V_{SSD} + V_{HERB} + V_{SRICH}}{6}\right) + V_{WLUSE}}{4}\right]}{2} \right\} \right\rangle^{1/2} \quad (3.5.2)$$

Figure 3.5.4. Conceptual distribution of variable influence on Habitat FCI values in areas displaying <20 percent canopy cover.

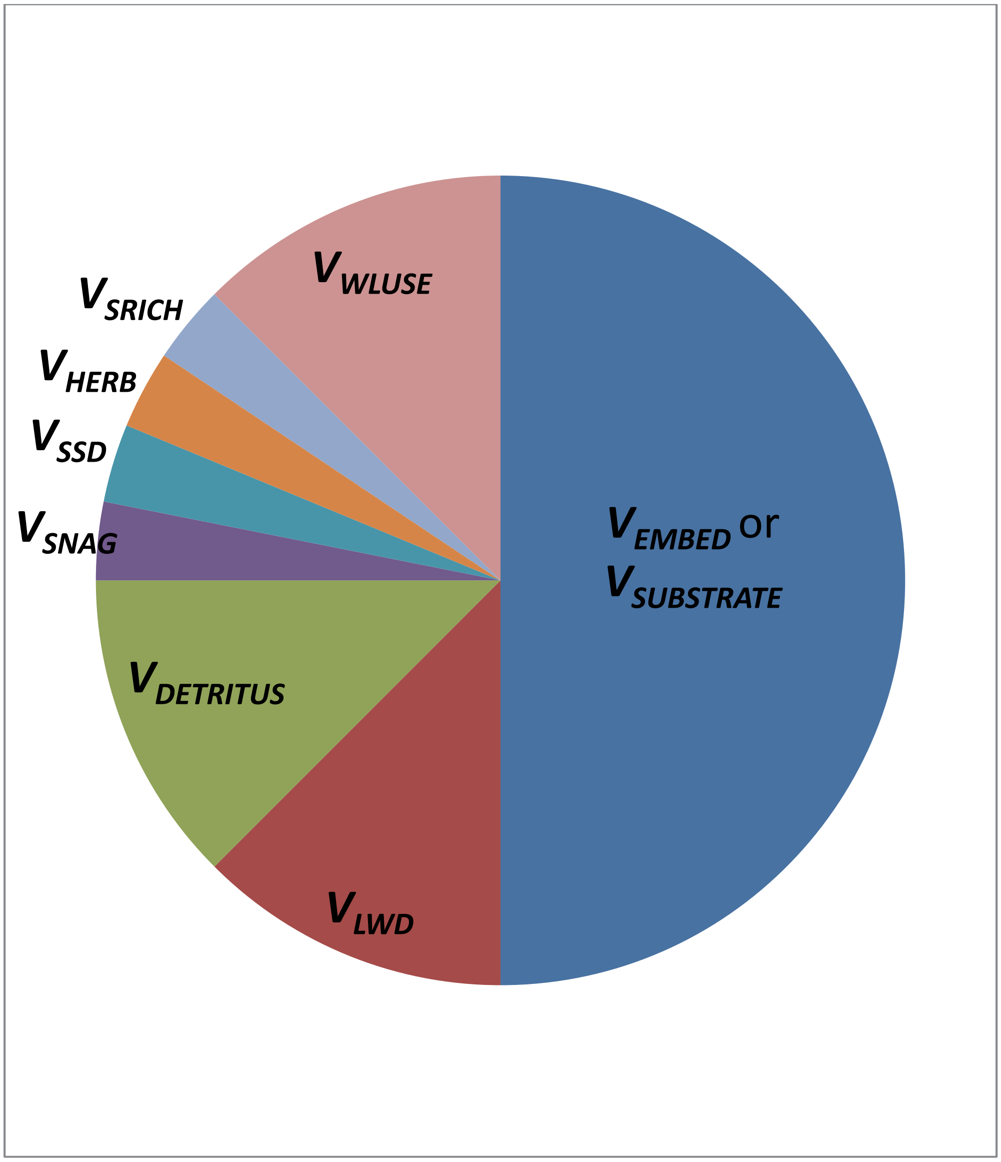


Table 3.5.2. Range of FCI scores attainable based on sensitivity analysis (<20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.10, 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.00	0.00	0.00	0.00	0.08	0.08	0.08	0.13	0.21
$V_{SUBSTRATE}$	0.01	0.00	0.01	0.08	0.00	0.08	0.21	0.00	0.21
V_{LWD} / $V_{DETRITUS}$	0.00	0.00	0.00	0.10	0.07	0.17	0.13	0.17	0.30
V_{WLUSE}	0.00	0.00	0.00	0.06	0.08	0.14	0.07	0.19	0.26
V_{SNAG} / V_{SSD} / V_{HERB} / V_{SRICH}	0.00	0.00	0.00	0.01	0.08	0.09	0.01	0.21	0.22
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.23	0.19	0.42	0.40	0.23	0.63	0.58	0.27	0.84
$V_{SUBSTRATE}$	0.42	0.00	0.42	0.63	0.00	0.63	0.84	0.00	0.84
V_{LWD} / $V_{DETRITUS}$	0.15	0.34	0.49	0.16	0.51	0.67	0.16	0.68	0.84
V_{WLUSE}	0.07	0.38	0.46	0.08	0.57	0.65	0.08	0.76	0.84
V_{SNAG} / V_{SSD} / V_{HERB} / V_{SRICH}	0.01	0.42	0.43	0.01	0.62	0.63	0.01	0.83	0.84

Figure 3.5.5. Results of sensitivity test of $V_{SUBSTRATE}$ (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

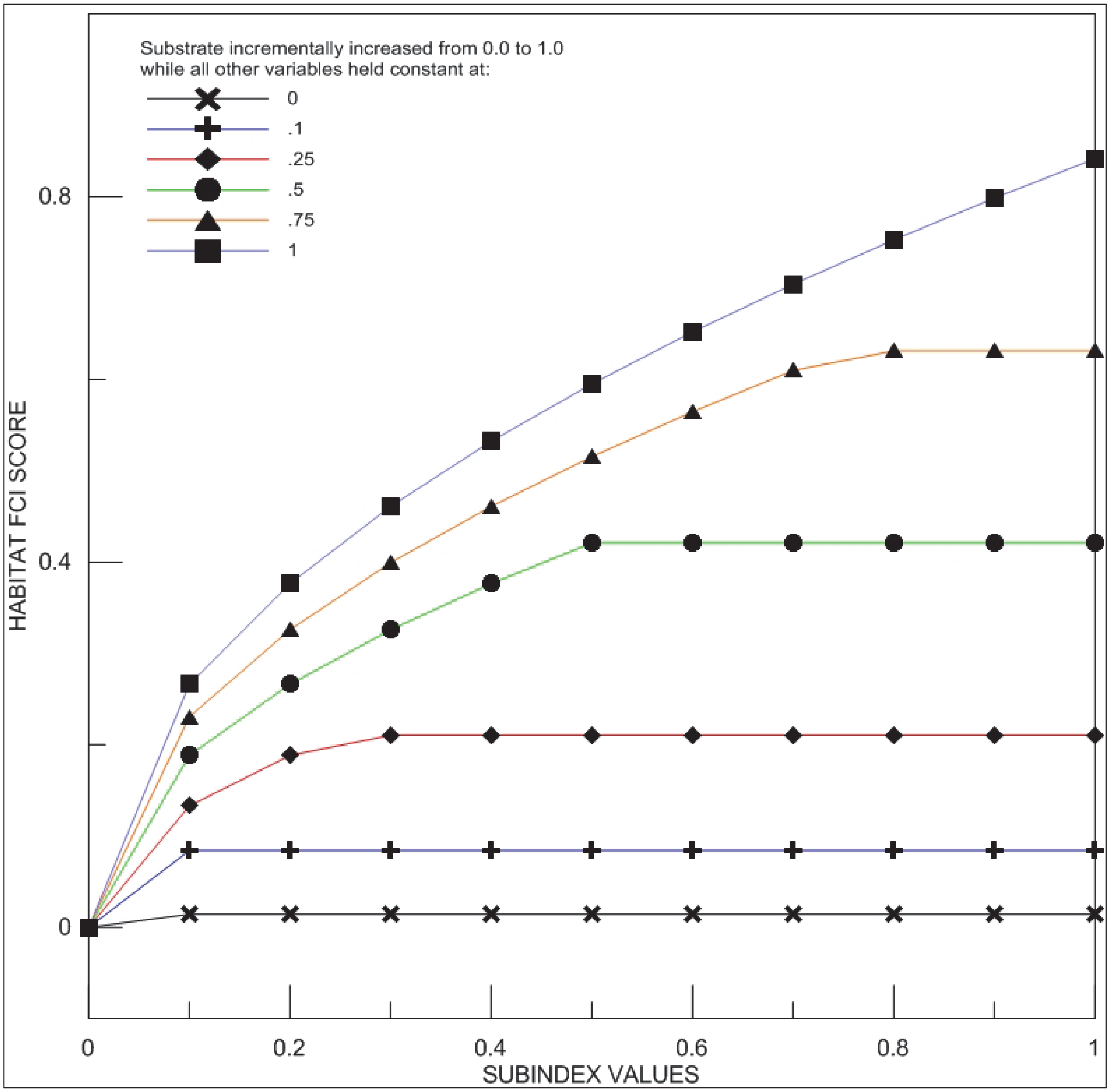
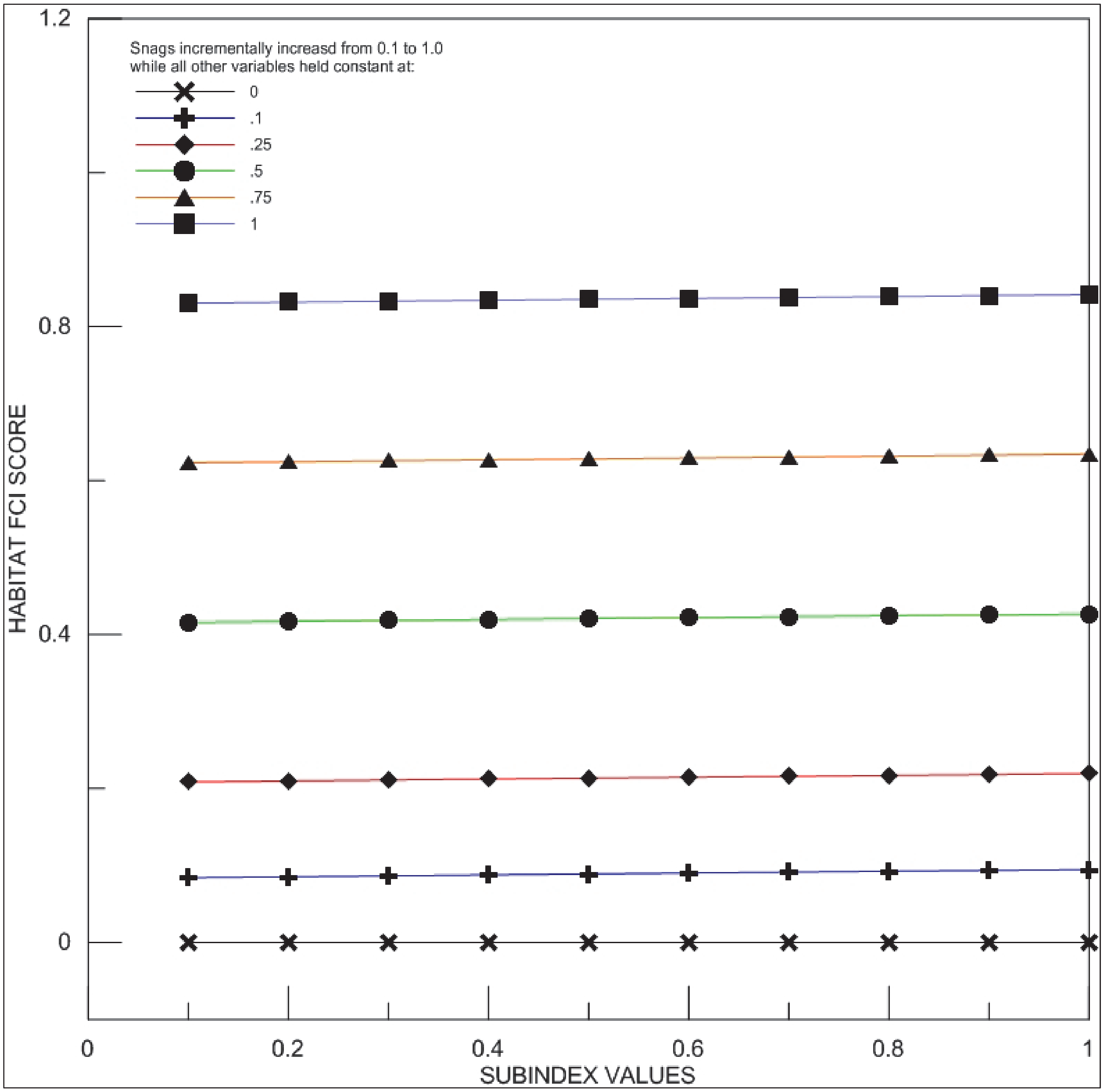


Figure 3.5.6. Results of sensitivity test of V_{SNAG} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



4 Validation of Biogeochemical Cycling function

4.1 Introduction

4.1.1 Functional definition and independent measures selected

The following chapter presents validation study results relating to the Biogeochemical Cycling function. The purpose of this section focuses on testing the ability of the Biogeochemical Cycling HGM rapid assessment model to differentiate between site conditions occurring along a gradient of alteration when compared against independent measures of ecological function. The study is not intended to provide a comprehensive evaluation of all potential biogeochemical functions, interactions, and relationships as they apply to general investigations evaluating stream or landscape ecological research.

The *Operational Draft Regional Guidebook for the Functional Assessment of High-gradient Ephemeral and Intermittent Headwater Streams in Western West Virginia and Eastern Kentucky* (Noble et al. 2010) defines the Biogeochemical Cycling function as:

> "The ability of the high gradient headwater stream ecosystem to retain and transform inorganic materials needed for biological processes into organic forms and to oxidize those organic molecules back into elemental forms through respiration and decomposition. Thus, biogeochemical cycling includes the activities of producers, consumers, and decomposers."

The validation model is presented in Section 4.5 and Equations 4.5.1 and 4.5.2. Validation of the assessment model followed the recommendations of Smith et al. (2013) by comparing HGM rapid assessment results against several selected independent measures of ecosystem function. The independent measures selected address three aspects of biogeochemical cycling as defined above: 1) input of nutrients and other materials into the ecosystem, 2) processing of nutrients and other materials within the ecosystem, and 3) cycling of nutrients and other materials within the ecosystem and transport to adjacent areas. The selection of independent measures incorporated input from the PDT, approaches suggested in the HGM guidebook, and a review of available literature sources. Table 4.1.1

outlines the independent measures applied. Table 4.1.2 provides site conditions and HGM Biogeochemical Cycling FCI score for each site.

Table 4.1.1. Independent measures of the Biogeochemical Cycling function.

Independent measure	Description
Leaf litter fall carbon	Amount of carbon per unit area introduced into each study site
Leaf litter fall nitrogen	Amount of nitrogen per unit area introduced into each study site
Leaf litter decomposition	Amount of material processed (% weight lost) within leaf litter bags at each sample location
Leaf litter carbon release	Amount of material processed (% carbon released) within leaf litter bags at each sample location
Leaf litter nitrogen release	Amount of material processed (% nitrogen released) within leaf litter bags at each sample location
Inorganic nitrogen loading	Estimated total inorganic nitrogen output per unit area within each catchment over the study period
Kjeldahl nitrogen loading	Estimated total organic nitrogen output per unit area within each catchment over the study period
Dissolved organic carbon loading	Estimated total organic carbon output per unit area within each catchment over the study period
Total phosphorous loading	Estimated total phosphorous output per unit area within each catchment over the study period
Total suspended solids loading	Estimated total suspended solids output per unit area within each catchment over the study period
Maximum conductivity	Maximum conductivity value observed within each sample location throughout the study period
Average pH	Average pH value observed within each sample location throughout the study period
Daily temperature range	Average daily temperature range observed within each sample location throughout the study period

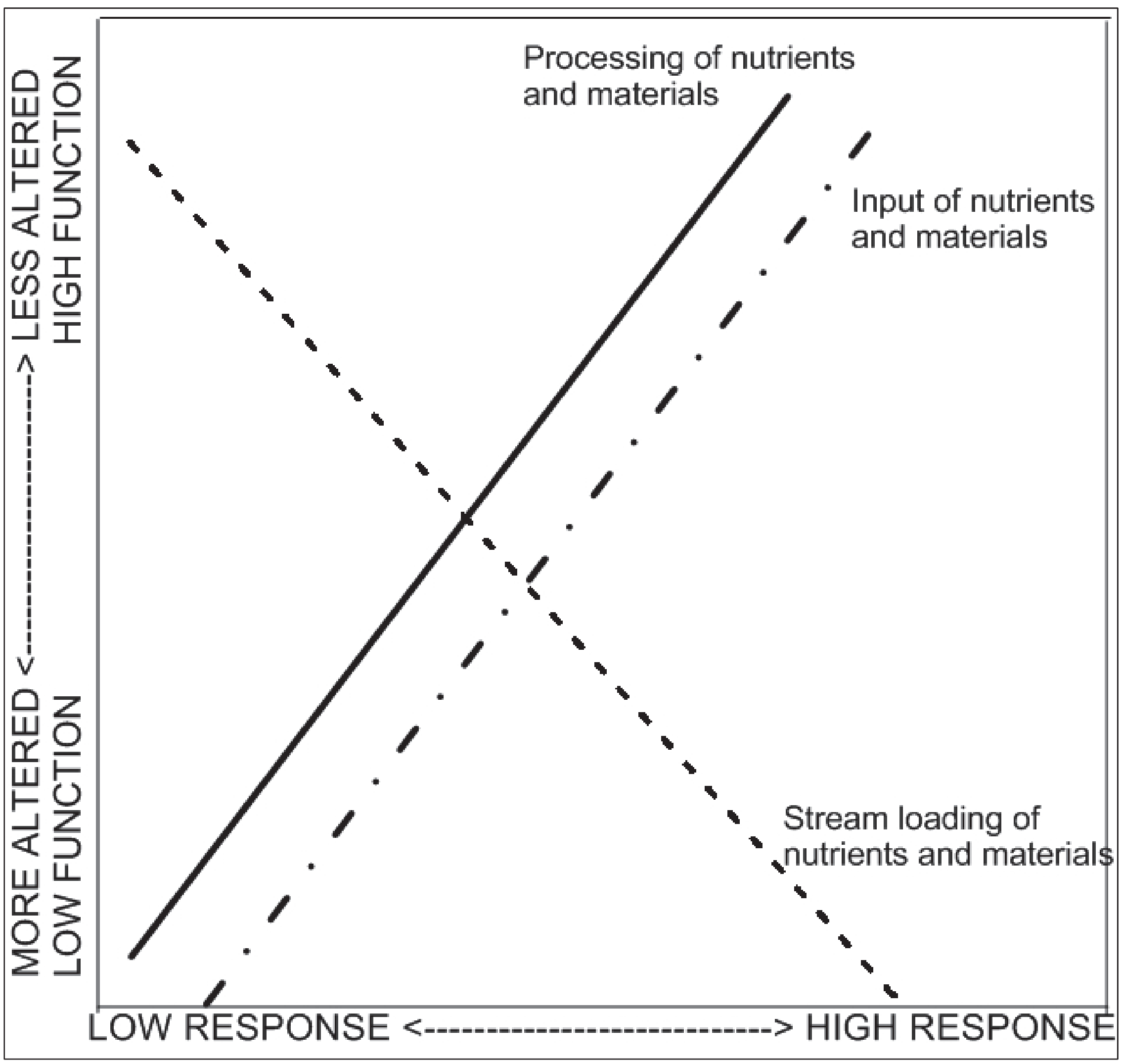
Table 4.1.2. Site Conditions and Biogeochemical Cycling FCI scores

Site	Site Condition	FCI Score
1	Silviculture, recreation	0.89
2	Mature second growth forest	0.93
3	Mature second growth forest	0.97
4	Surface mining	0.72
5	Mature second growth forest	0.91
6	Surface mining	0.61
7	Grazing	0.62
8	Silviculture, mature second growth forest	0.65
9	Mature second growth forest	0.90
10	Surface mining	0.29

4.1.2 Summary of findings

Each of the three aspects examined as part of the Biogeochemical Cycling model validation support the HGM model. Figure 4.1.1 depicts the predominant trends within the data set, providing a conceptual model of the relationship between material inputs, processing, and loading/transport across the alteration gradient observed in the study sites.

Figure 4.1.1. Conceptual model of the relationship between material inputs, processing, and loading/transport across the alteration gradient observed in the study sites. Sample sites characterized by unaltered stream channels and forested watershed received and processed large amounts of nutrients and materials, while transporting low levels of nutrients and materials. Altered sample sites received and processed few nutrients and materials, while displaying increased transport.



Leaf litter fall traps characterized the input of nutrients and other materials (e.g., leaves, twigs, particulates, etc.) into each study area (Section 4.2). Study sites characterized by mature forest canopy, channel attributes within or near the reference standard range, and containing few watershed alterations exhibited high levels of carbon and nitrogen inputs from leaf litter fall. Conversely, sites lacking developed forest structure, displaying channel attributes outside of reference standard range, and containing areas of alteration (impervious surface, agricultural inputs, past or ongoing mining operations) exhibited lower levels of carbon and nitrogen inputs from leaf litter fall.

The processing of nutrients and other materials introduced to each sample location was investigated using leaf litter bags (Section 4.3). Study sites characterized by mature forest canopy, channel attributes within or near the reference standard range, and containing few watershed alterations exhibited high levels of material processing through high rates of decomposition, carbon release, and nitrogen release. Conversely, sites lacking developed forest structure, displaying channel attributes outside of reference standard range, and containing areas of alterations (impervious surface, agricultural inputs, past or ongoing mining operations) exhibited lower rates of decomposition, carbon release, and nitrogen release.

The cycling of elements and compounds within each sample area and transport to adjacent areas was investigated by measuring the loading of nutrients and other materials at each sample location (Section 4.4). Study sites characterized by mature forest canopy, channel attributes within or near the reference standard range, and containing few watershed alterations exhibited decreased loading of nitrogen, carbon, phosphorus, and suspended sediments. These findings demonstrate that additional biogeochemical cycling is occurring within mature forested, more functional study sites. Conversely, sites lacking developed forest structure, displaying channel attributes outside of reference standard range, and containing areas of alterations (impervious surface, agricultural inputs, past or ongoing mining operations) exhibited higher rates of nutrient and sediment transport through increased loading of nitrogen, carbon, phosphorus, and suspended sediments. These findings demonstrate that decreased rates of biogeochemical cycling occurred within altered, lower functioning study sites.

The relationship between the HGM assessment model and independent measures is clear. Highly functional sample locations receive and process large amounts of nutrients and other compounds and perform biogeochemical cycling functions at high rates. This results in lower loading rates out of the headwater catchment. Low functioning sites receive and process lower amounts of nutrients and other materials and perform less biogeochemical cycling. As a result, higher loading rates are observed in altered catchments. The HGM Biogeochemical Cycling model effectively differentiates between high and low functioning headwater stream ecosystems, and validation data supports the continued use of this model. Sensitivity analysis results demonstrate that all variables incorporated into the Biogeochemical Cycling model affect rapid assessment outcomes and interact as intended by model developers (Section 4.5).

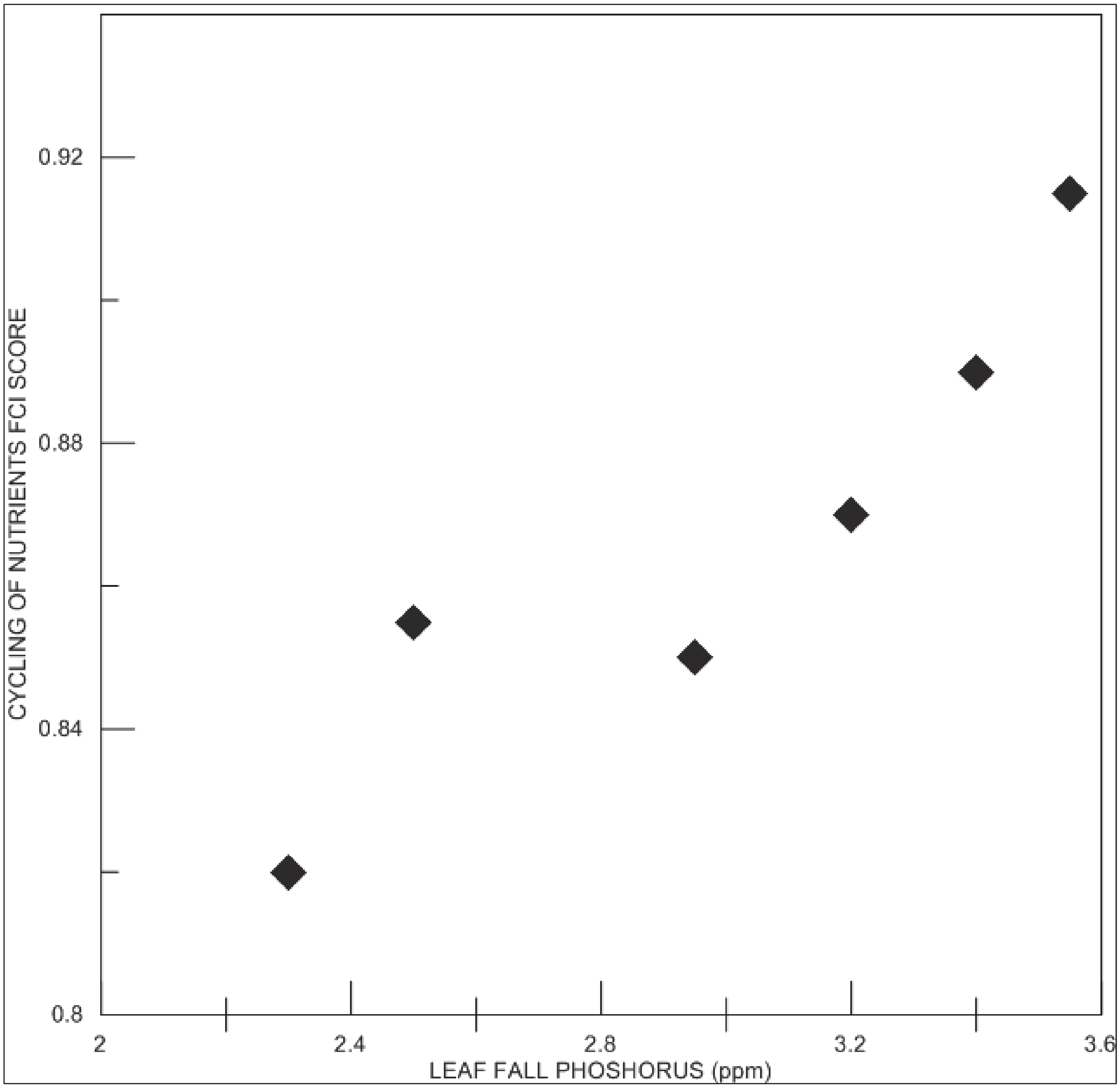
4.2 Leaf fall

4.2.1 Rationale for selecting the independent measure

Examinations of leaf litter fall and leaf litter decomposition rates provide insight into nutrient sources, and factors controlling organic matter transformations occurring through biogeochemical cycling (Benfield 1996; Aerts 1997). Organic carbon provides for maintenance of plant communities, including annual primary productivity, composition, and diversity while allowing for trophic transfer to higher organisms (Bormann and Likens 1970; Whittaker 1975; Perry 1994; Carpenter et al. 1998). Leaf litter fall represents a major biologic contributor to forest litter pools and annual leaf litter fall coincides with periods of increased nutrient output (Band et al 2001; Bormann and Likens 1967). Leaf litter represents a major source of coarse particulate organic matter to streams, resulting in leaf input and decomposition receiving considerable attention in the scientific literature (Webster and Benfield 1986, Allan 1995, Gessner et al. 1999).

Ferrari (1999) reported that leaf litter fall accounted for 69 percent of total nitrogen contributions to a mixed conifer-deciduous forest. Many studies link leaf fall decomposition products and leachate to soil and stream ecosystem biogeochemistry (Fisher and Likens 1973; McDowell and Likens 1988). Leaf litter inputs represent abundant and often reactive energy sources to streams and aquatic systems (Qualls 2004; 2005). Franklin et al. (2009) investigated six sites located within riverine wetlands in Tennessee, USA. Results indicate a significant correlation between the FCI score generated for the Biogeochemical Cycling function and leaf fall phosphorus (Figure 4.2.1). To the authors' knowledge, the six data points presented in Figure 4.2.1 represent the current extent of published studies examining HGM results and nutrient cycling or other biogeochemical functions.

Figure 4.2.1. Scatter diagram relating HGM Biogeochemical Cycling FCI score with leaf fall phosphorus (a proposed independent measure of assessment model validity); adapted from Franklin et al. (2009).



4.2.2 Methods

The leaf litter fall data collected during the current validation effort represent the amount of nitrogen (N) and carbon (C) input to each sample location per unit area during the study period. Calculations incorporated the total weight of leaves captured within four leaf traps of known area located in the riparian buffer zone adjacent to each stream (Figure 4.2.2). Traps were checked approximately weekly from October 2011 – January 2012. If leaves were present, leaf mass was measured and a 950-mL subsample was taken from each leaf litter trap for analysis of nutrient concentrations. All samples were

Figure 4.2.2. Leaf fall traps deployed during the study period within (clockwise from top left): 1) forested area, 2) open area, 3) leaf collection, and 4) leaf fall traps distributed throughout riparian/buffer zone.

homogenized prior to analysis. Sampling and analysis did not include determination of leaf fall species composition or species distribution. Following collection, contents of each litter sample were subsampled for total moisture content (105 °C), oven dried (60 °C), ground, and prepared for analysis (Klute 1986). Measurements included determination of nitrogen (N) and carbon (C) concentrations via combustion on an Elementar Vario Macro Carbon Nitrogen analyzer (900 °C) (Kahn 1998). The amount of leaf litter fall C and N was calculated using Equation 4.2.1.

Statistical procedures included normality testing applying the Shapiro-Wilk test (p-value > 0.05 represent normal distributions) followed by Pearson Product Moment Correlation analysis ($\alpha = 0.5$; SPSS IBM, Inc. Version 20).

$$\text{Mass of nutrient input at each site per unit area} = \sum(M_{xi} * C_{xi})/A \qquad (4.2.1)$$

where: M_x is the mass of leaf materials present in leaf trap x at time i, C_x is the concentration of nutrient in leaf trap x at time i, and A is the leaf trap area.

4.2.3 Results

Leaf C inputs ranged from 15.2 – 163 gC/m^2 with an average of 100 gC/m^2 and displayed normal distribution (Figure 4.2.3; Table 4.2.1). These results compare well with the findings compiled by Petersen and Cummins (1974) who report daily organic matter inputs to small streams between 0.97 – 4.2 $g/m^2/day$. The current study displays average inputs of 1.57 $g/m^2/day$ (range = 1.26 – 1.81 $g/m^2/day$) for forested locations based on a 90-day sampling period. Comparisons of leaf C input with predicted results of the HGM Biogeochemical Cycling FCI score yielded significant positive correlations ($r = 0.805$, $n = 10$, $P = 0.005$), supporting the current HGM model (Figure 4.2.4).

Leaf N inputs ranged from 0.29 – 3.87 gN/m^2 with an average of 2.40 gC/m^2 and displayed normal distribution (Figure 4.2.5; Table 4.2.1). Comparisons of leaf N inputs with HGM Biogeochemical Cycling FCI scores yielded significant positive correlations ($r = 0.777$, $n = 10$, $P = 0.008$) supporting the current HGM model (Figure 4.2.6).

4.2.4 Summary

Leaf litter fall data provides evidence that the HGM Biogeochemical Cycling model differentiated between sample locations displaying a range of watershed, channel, and riparian/buffer zone characteristics. Study sites displaying increased C and N leaf fall input exhibit mature second growth forest structure, watersheds lacking significant alterations (i.e., impervious surface, open ground), and stream channel characteristics (e.g., embeddedness) near the reference standard range. Study sites with lower C and N leaf fall inputs included areas lacking a developed forest structure, containing fill or open ground within the watershed, and displaying stream channel particles with low embeddedness scores.

Figure 4.2.3. Leaf carbon input per unit area collected at each sample location.

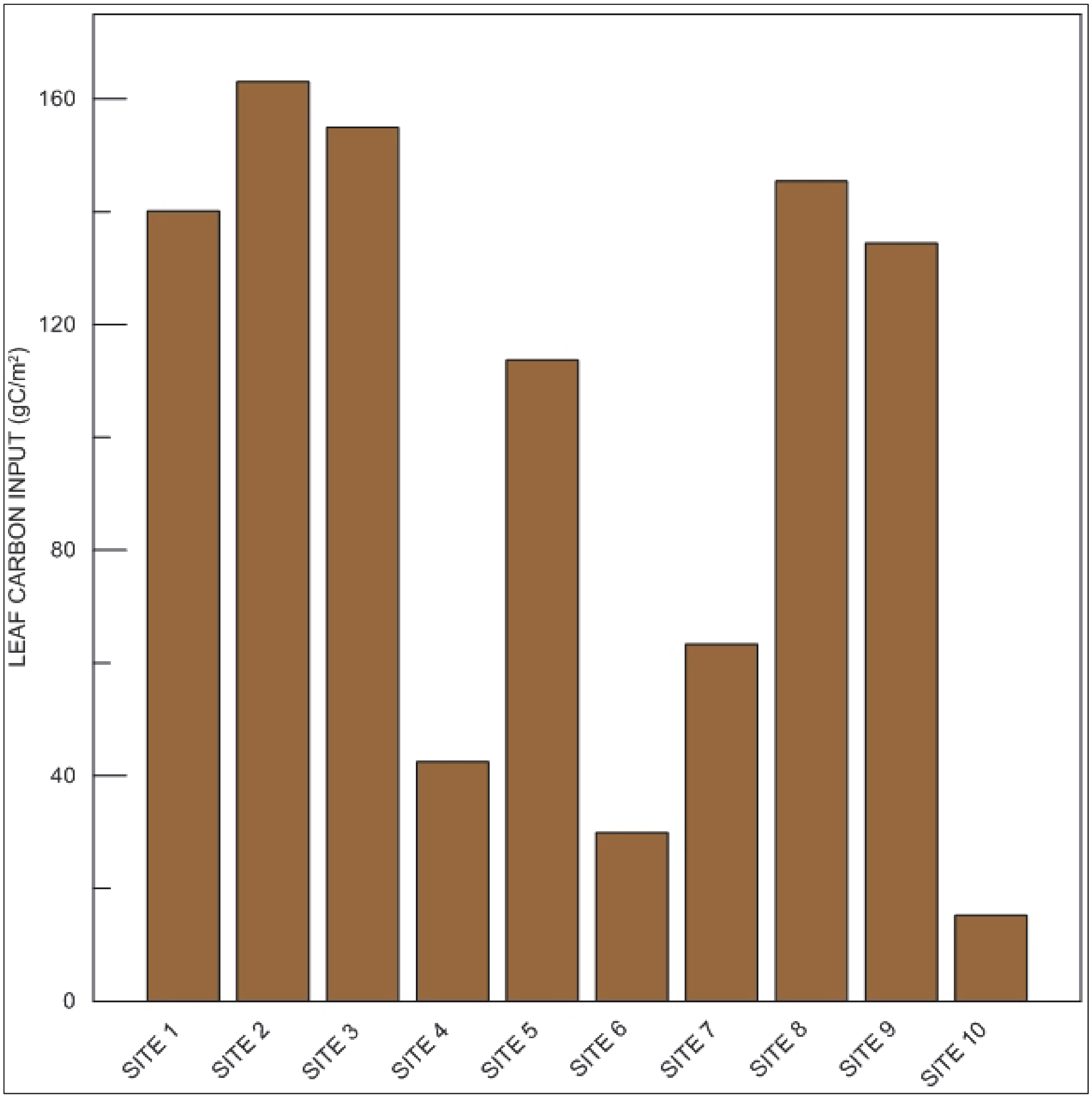


Table 4.2.1. Tests of Normality – Leaf fall Carbon and Nitrogen.

	Shapiro-Wilk		
Parameter	W	df	p
Carbon (g/m^2)	.870	10	.100
Nitrogen (g/m^2)	.897	10	.202

Figure 4.2.4. Leaf carbon input compared with HGM Biogeochemical Cycling FCI score.

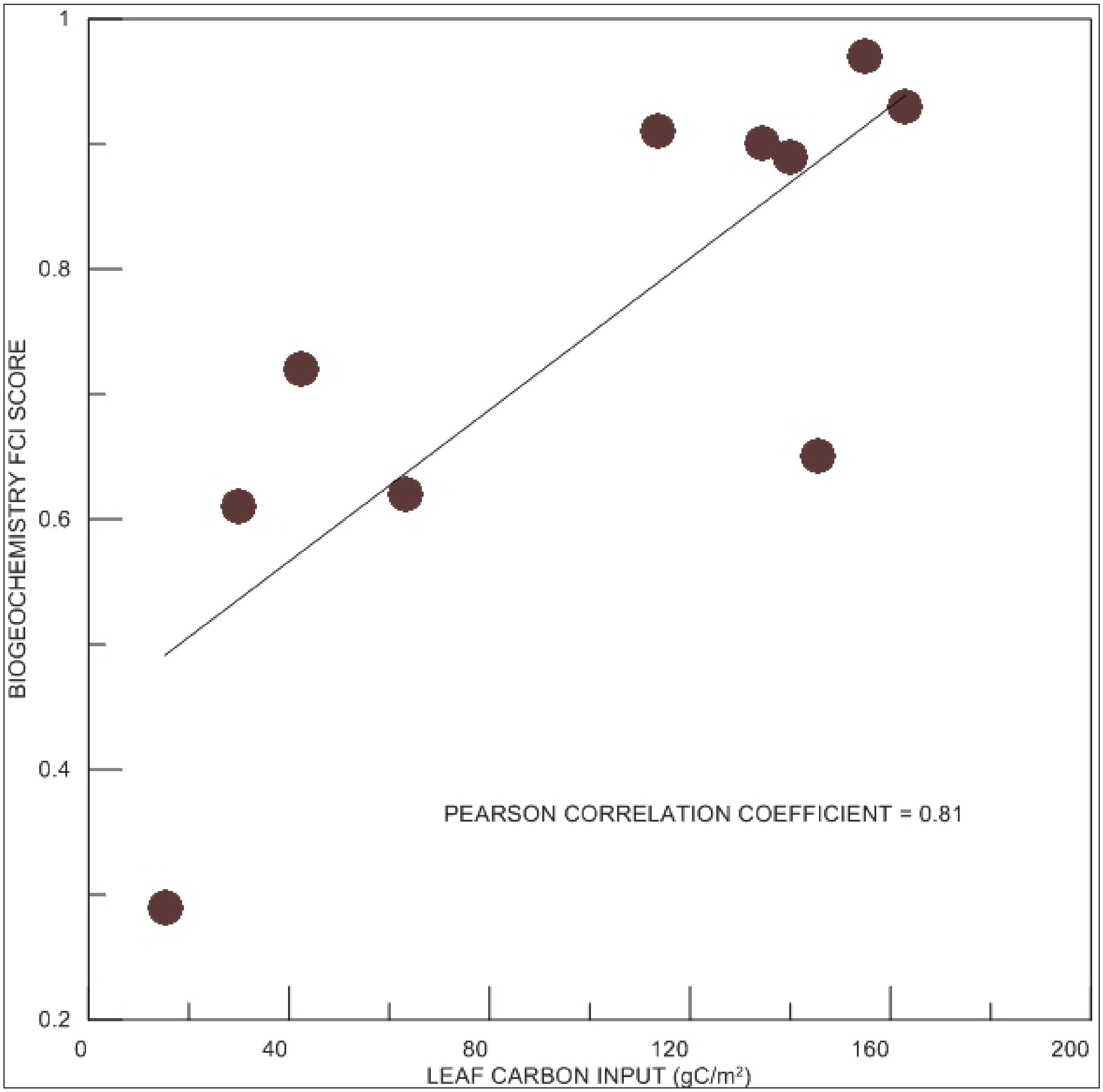


Table 4.2.2. Pearson Correlation Coefficients. **Correlation is significant at the 0.01 level.

Function	Statistic	C (g/m2)	N (g/m2)
Biogeochemical Cycling FCI Score	Pearson Correlation	.805**	.777**
	p-value (1-tailed)	.005	.008
	N	10	10

Figure 4.2.5. Leaf nitrogen input per unit area collected at each sample location.

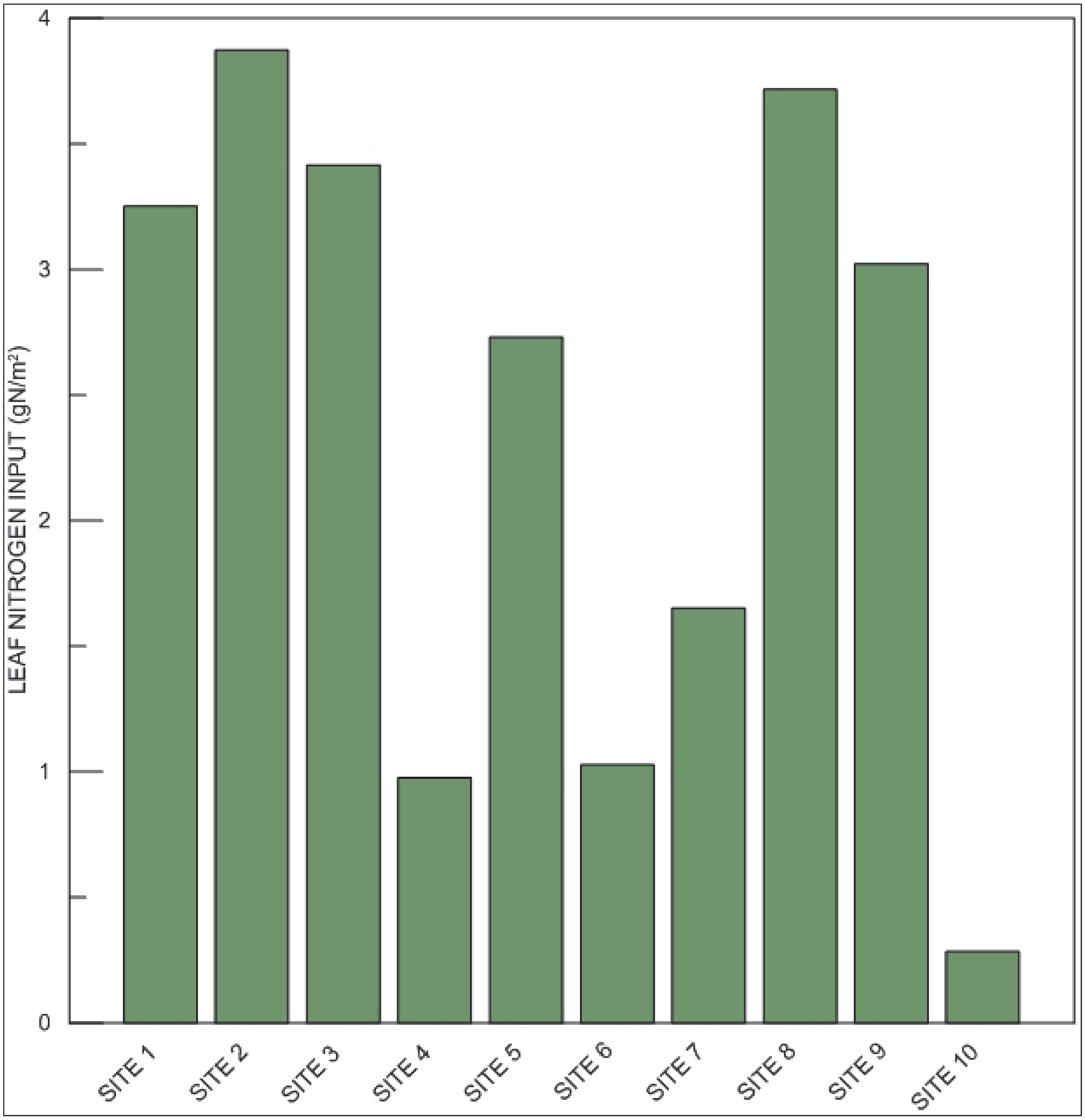
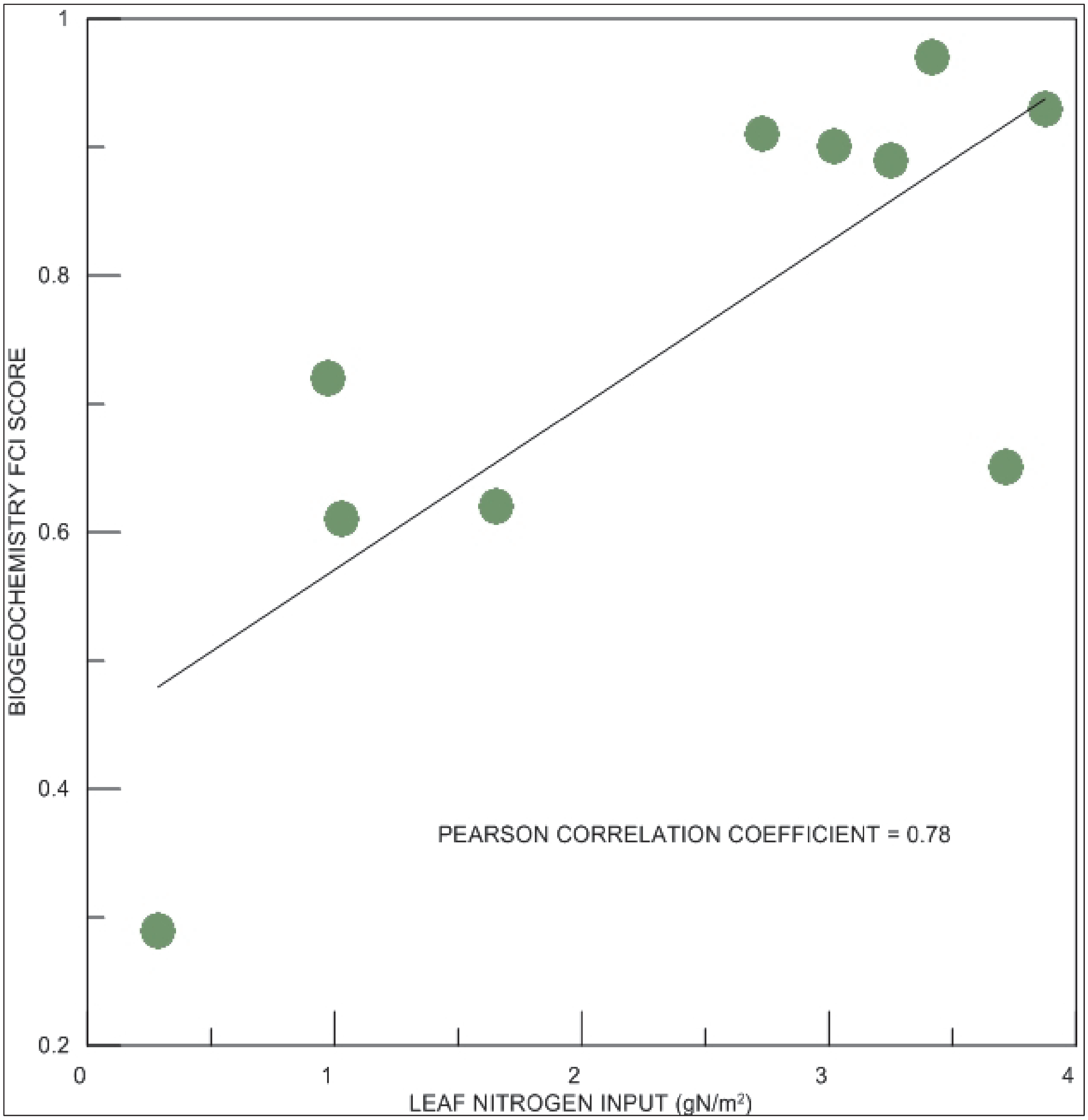


Figure 4.2.6. Leaf nitrogen input compared with HGM Biogeochemical Cycling FCI score.



4.3 Leaf litter bag decomposition

4.3.1 Rationale for selecting the independent measure

A number of researchers indicate that allochthonous inputs (e.g., leaf litter decomposition) provide the dominant energy source to small woodland stream systems (Nelson and Scott 1962; Hynes 1963; Minshall 1967; Cummins et al. 1973). Headwater streams exemplify these findings due to heavy forest canopy shading and the introduction of large amounts of coarse particulate organic matter from the terrestrial environment into the stream channel (Petersen and Cummins 1974; Figure 4.3.1).

Figure 4.3.1. Leaf litter filled stream channel shortly after autumnal abscission.

The leaching and decomposition of leaf litter and other detrital materials provides the basis of stream biogeochemical cycling and transfer of stream nutrients to higher trophic levels as exhibited by the synchronization of stream insect communities to seasonal leaf inputs (Petersen and Cummins 1974). The majority of research examining leaf litter decomposition and nutrient release to stream systems employs leaf litter bag decomposition studies (Benfield 1996; Vitousek et al. 1994). Since leaf litter decomposition occurs via physical, chemical, and biological processes (Meyer 1980), litter bag decomposition and nutrient release studies provide an appropriate independent measure of biogeochemical function.

Anthropogenic and natural disturbances affect stream biota and ecosystem functioning directly and indirectly, impacting leaf litter decomposition, nutrient release, and biogeochemical cycling (Gulis and Suberkropp 2003). Carpenter et al. (1998) identified nutrient inputs, agricultural runoff, and urban activities as areas of major concern in streams; all impacts occurring within the validation study area. Hagen et al. (2006) investigated leaf litter decomposition rates across a gradient of agricultural impacts, and Meyer (1980) compared leaf bag decomposition rates under high and low sedimentation regimes. Hough and Cole (2006) examined litter bag decomposition across two HGM wetland regional subclasses, but provided no data comparing HGM outcomes to leaf litter bag results. Recently, Gingerich and Anderson (2011) examined leaf decomposition rates within natural and constructed wetlands in West Virginia, USA. Again, the study included no examination of the relationships between leaf bag results and HGM FCI scores. The current work utilizes litter bag decomposition and nutrient release as an independent measure of biogeochemical cycling across an alteration gradient within headwater streams in the southern Appalachians.

4.3.2 Methods

Leaf litter decomposition bags consisting of 1-cm mesh PVC hardware cloth and measuring 23 x 23 cm were constructed in the laboratory. To ensure an equal distribution of leaf species composition and nutrient content within bags across study sites, all material used to fill the leaf litter bags was collected at one sample location soon after abscission and thoroughly mixed (Gulis and Suberkropp 2003). Species included: *Liriodentron tulipifera*, *Platanus occidentalis*, *Magnolia trepetala*, *Magnolia acuminata*, *Acer saccharum*, *Acer rubrum*, *Fagus grandifolia*, *Betula lenta*, *Carya cordiformis*, *Nyssa sylvatica*, *Tilia americana*, *Aesculus flava*, *Quercus rubra*, *Quercus prinus*, *Quercus alba*, and *Quercus velutina*.

The approach of collecting all leaves at one location prevents the determination of site-specific decomposition rates (which depends on site litter quality), but allows for investigation of environmental effects of decomposition and nutrient release across sampling locations (Hough and Cole 2009). Collected leaf material underwent drying until constant weight was reached. Each bag received 30 g of dried leaf litter. Eight control leaf litter bags underwent analysis (described below) for carbon (C) and nitrogen (N) prior to deployment, facilitating the calculation of nutrient release over the study period.

Leaf litter bag placement consisted of distributing four bags located in the riparian/buffer zone (within 25 feet of the stream) at each sample location (Figure 4.3.2). The collection of four replicate leaf litter bags (Harmon et al. 1999) from each location occurred following 6 months of exposure. Following collection, the contents of each litter bag were oven dried (60 °C), subsampled for total moisture content (105 °C), ground, and prepared for analysis (Klute 1986). Measurements included loss of material weight via decomposition and determination of nitrogen (N) and carbon (C) concentrations via combustion on an Elementar Vario Macro Carbon Nitrogen analyzer (900 °C) (Kahn 1998).

Figure 4.3.2. Leaf litter decomposition bags deployed during the study period within (clockwise from top left): 1) forested area, 2) open area, 3) leaf bag distributed throughout riparian/buffer zone, and 4) leaf bag collection.

Loss of mass from leaf litter bags was determined by summing the total amount of mass decomposed from replicate bags at each sampling location and dividing by the total mass of leaf litter deployed within each sample site (Equation 4.3.1).

Leaf mass decomposition = $(M_{(time=0)} - M_{(time=x)}/M_{(time=0)})*100\%$; where M = litter mass (4.3.1)

Leaf C and N release was measured by summing the total amount of C or N removed from replicate bags via processing at each sampling location and dividing by the total amount of nutrients deployed within each sample site (Equation 4.3.2).

Nutrient release = $(N_{(time=0)} - N_{(time=x)}/N_{(time=0)})*100\%$; where N = nutrient abundance (4.3.2).

Several studies (Petersen and Cummins 1974; Karberg et al. 2008; others) demonstrate that the mass loss from litter bags and decreases in leaf nutrients cannot be completely attributed to decomposition. In other words, leaf litter bag studies fail to differentiate between losses due to leaching, conversion to carbon dioxide, communition, and removal of leaf fragments by invertebrates. As a result, the data presented here represents leaf mass and nutrient "processing," providing a useful comparison of cycling across the gradient of validation study sites.

Statistical procedures included normality testing applying the Shapiro-Wilk test ($p>0.05$ indicates a normal distribution) followed by Pearson Product Moment Correlation analysis ($\alpha = 0.05$; SPSS IBM, Inc. Version 20).

4.3.3 Results

Leaf litter mass decomposition ranged from 23.1 – 61.7 percent with an average of 43.1 percent over six months of exposure and displayed normal distribution (Figure 4.3.3; Table 4.3.1). These results compare well with the findings of Gosz et al. (1973) who found approximately 30 percent leaf litter mass decomposed following six months of *Acer saccharum* exposure. Additionally, Gingerich and Anderson (2011) report mass decomposition of 45.4 – 56 percent over a period of one year. Although their study examined *Typha latifolia* decomposition and occurred over a one year exposure period, the study took place within West Virginia and included sample locations adjacent to headwater streams. Vargo et al. (1998) and

Kittle et al. (1995) also investigated *Typha latifolia* decomposition, reporting mass loss rates of 41.0 to 69.7 percent over a five month period. Comparisons of leaf litter mass decomposition with predicted results of the HGM Biogeochemical Cycling FCI score yielded significant positive correlations (r = 0.636, n = 10, P = 0.024) supporting the current HGM model (Figure 4.3.4).

Figure 4.3.3. Leaf bag mass decomposition at each sample location.

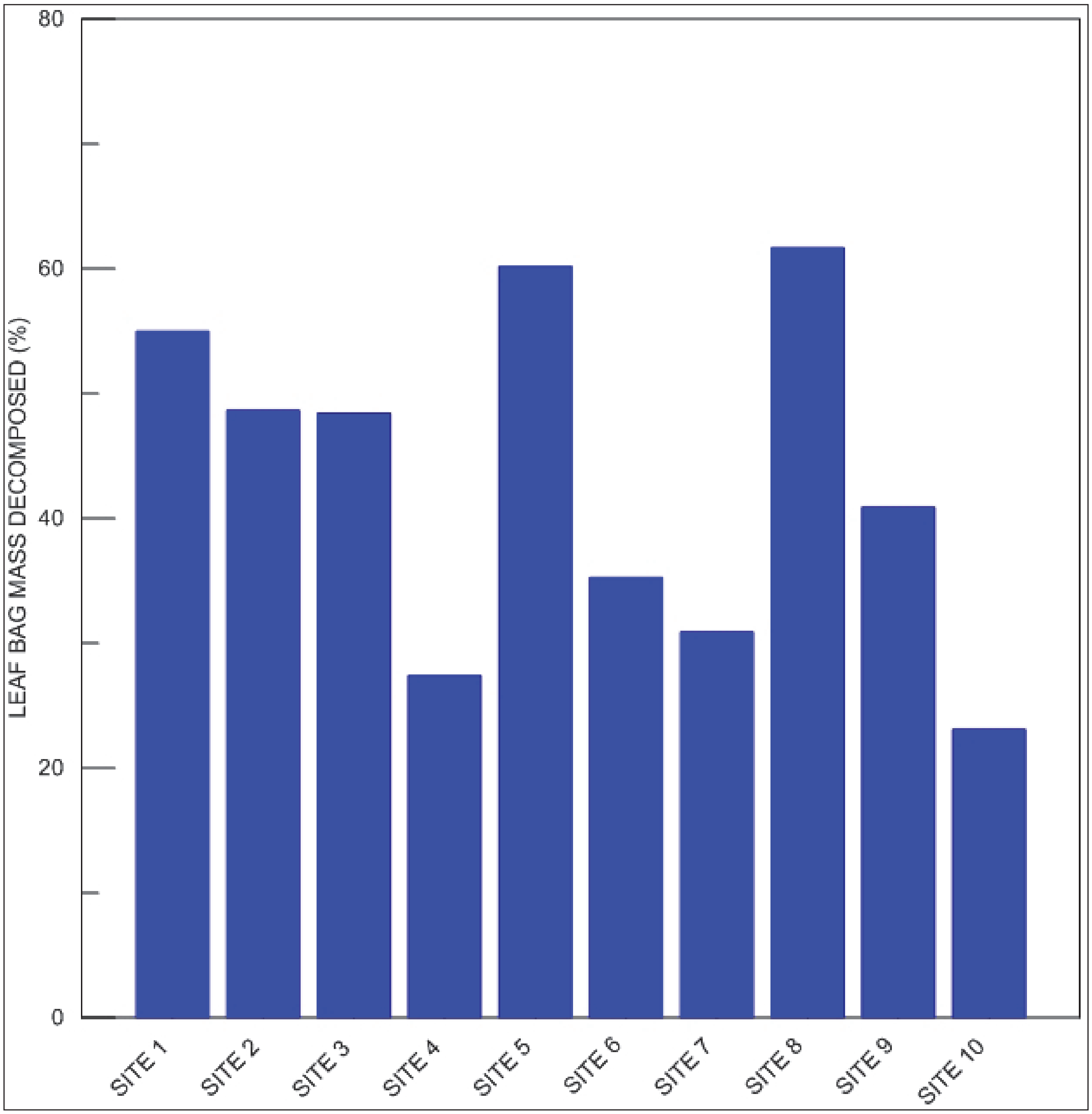
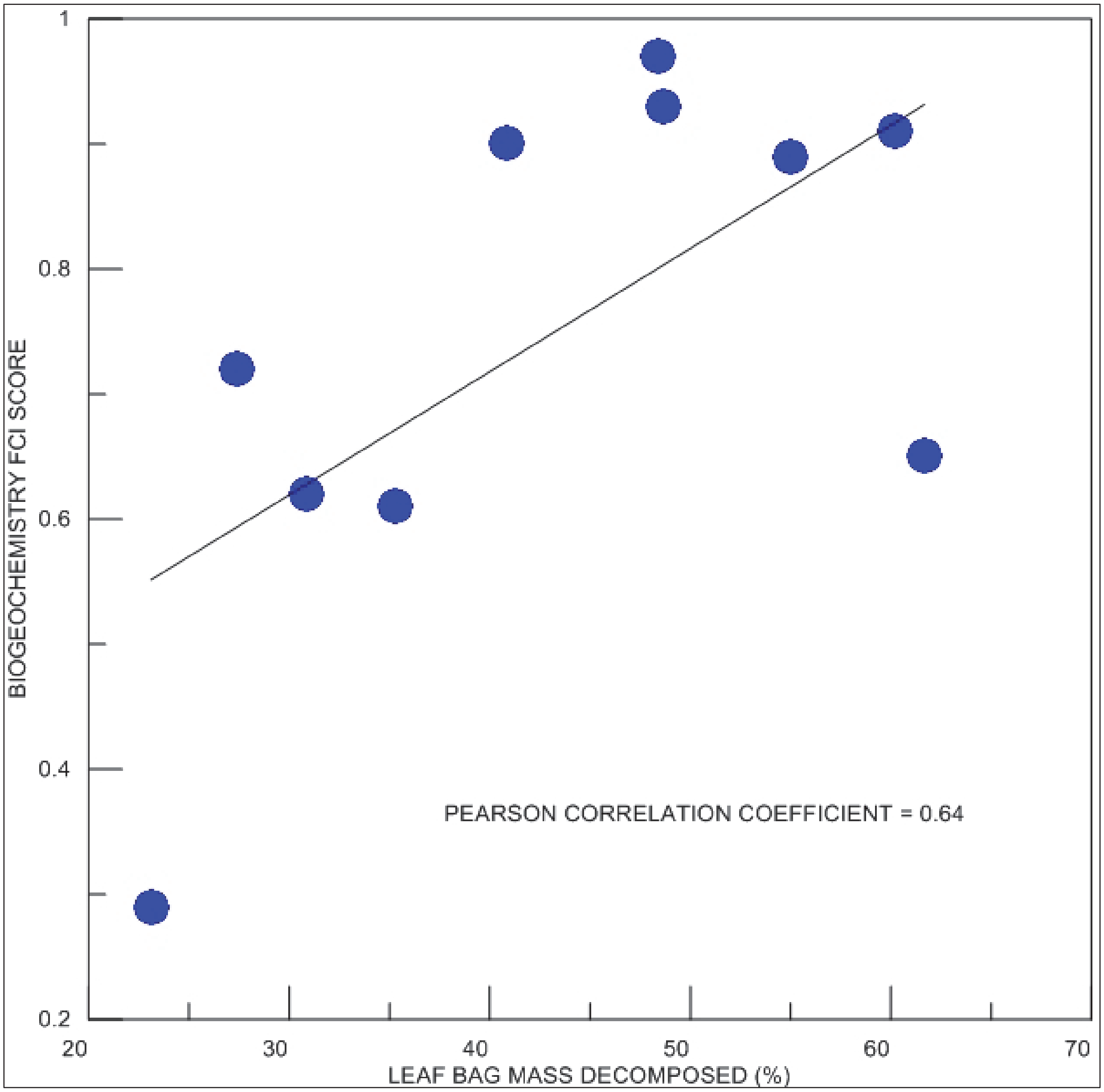


Figure 4.3.4. Leaf bag mass decomposition compared with HGM Biogeochemical Cycling FCI score.



Leaf litter bag C release ranged from 22.7 to 48.1 percent with an average of 37.6 percent and displayed normal distribution (Figure 4.3.5; Table 4.3.1). As pointed out above, the litter bag study does not address the fate of processed C, with possible outcomes including respiration as CO_2, removal as particulate organic matter, or leaching as DOC (Petersen and Cummins 1974). Comparisons of leaf C processing the HGM Biogeochemical Cycling FCI score yielded significant positive correlations (r = 0.674, n = 10, P = 0.016), supporting the current configuration of the functions presented in the HGM model (Figure 4.3.6).

Figure 4.3.5. Leaf carbon release collected at each sample location.

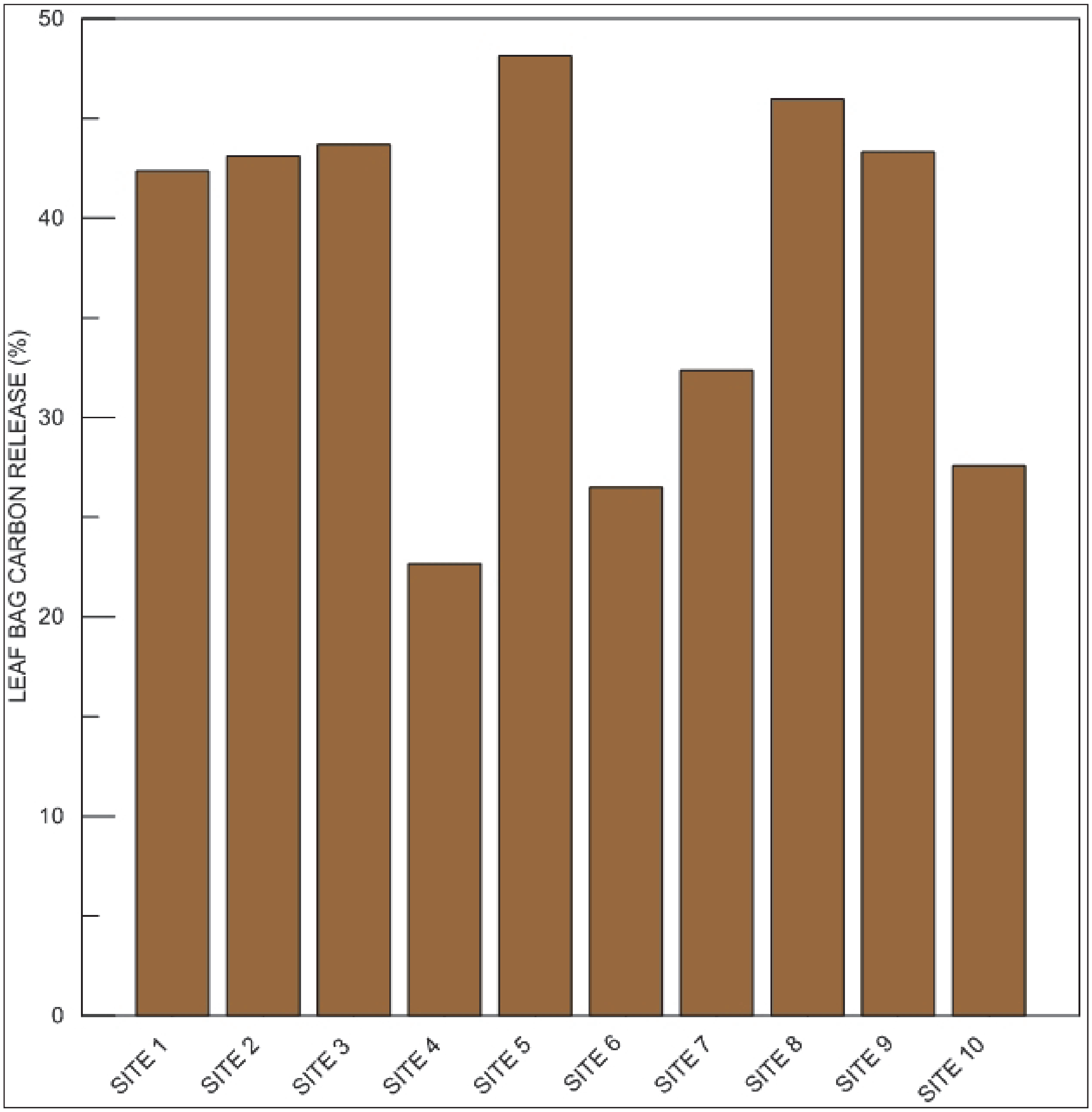
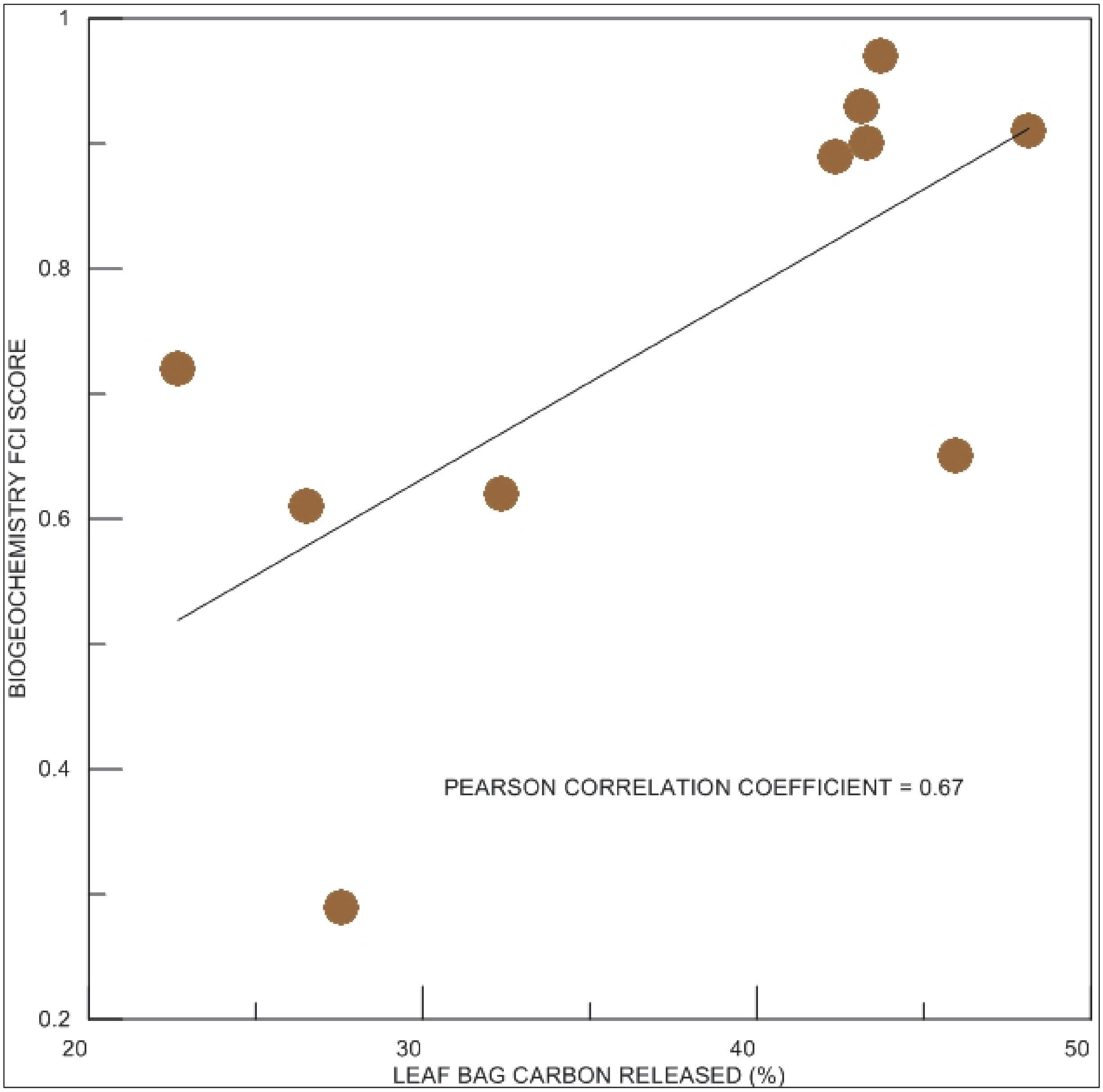


Table 4.3.1. Tests of Normality – Leaf bag percent weight decomposed, carbon, and nitrogen release.

	Shapiro-Wilk		
Parameter	W	df	p
Mass decomposed (%)	.943	10	.585
Nitrogen release (%)	.943	10	.586
Carbon release (%)	.859	10	.074

Figure 4.3.6. Leaf bag carbon release compared with HGM Biogeochemical Cycling FCI score.



Similarly, leaf litter bag N release ranged from 3.2 to 27.4 percent with an average of 16.0 percent and displayed normal distribution (Figure 4.3.7; Table 4.3.1). Simons and Seastedt (1999) reported 29 to 38 percent N release over 225 days for *Populus deltoides* riparian litter located in the Southwestern US. Comparisons of leaf N processing with the HGM Biogeochemical Cycling FCI score yielded significant positive correlations ($r = 0.812$, $n = 10$, $P = 0.002$), supporting the current configuration of the HGM model (Figure 4.3.8).

Figure 4.3.7. Leaf nitrogen release collected at each sample location.

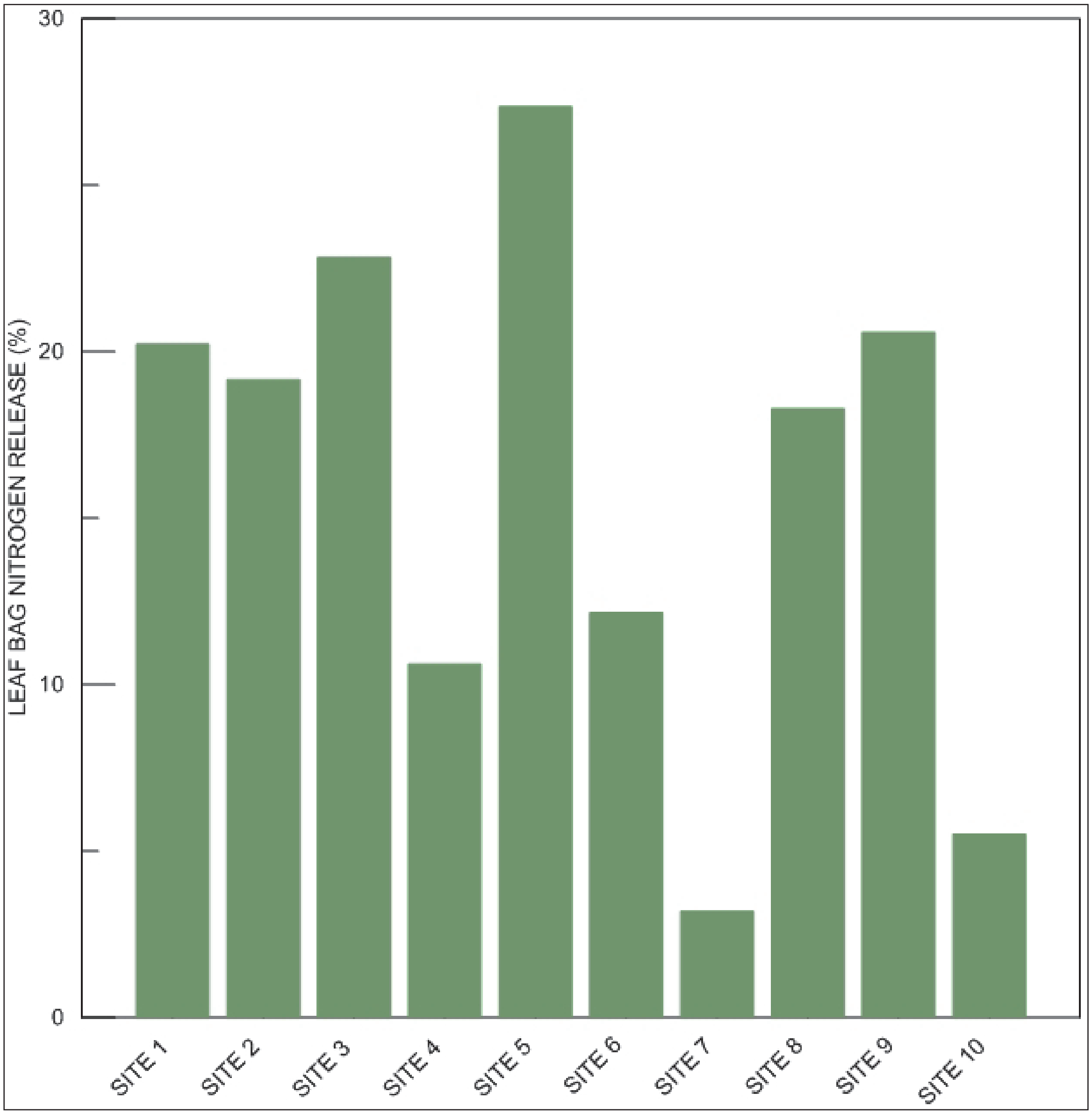
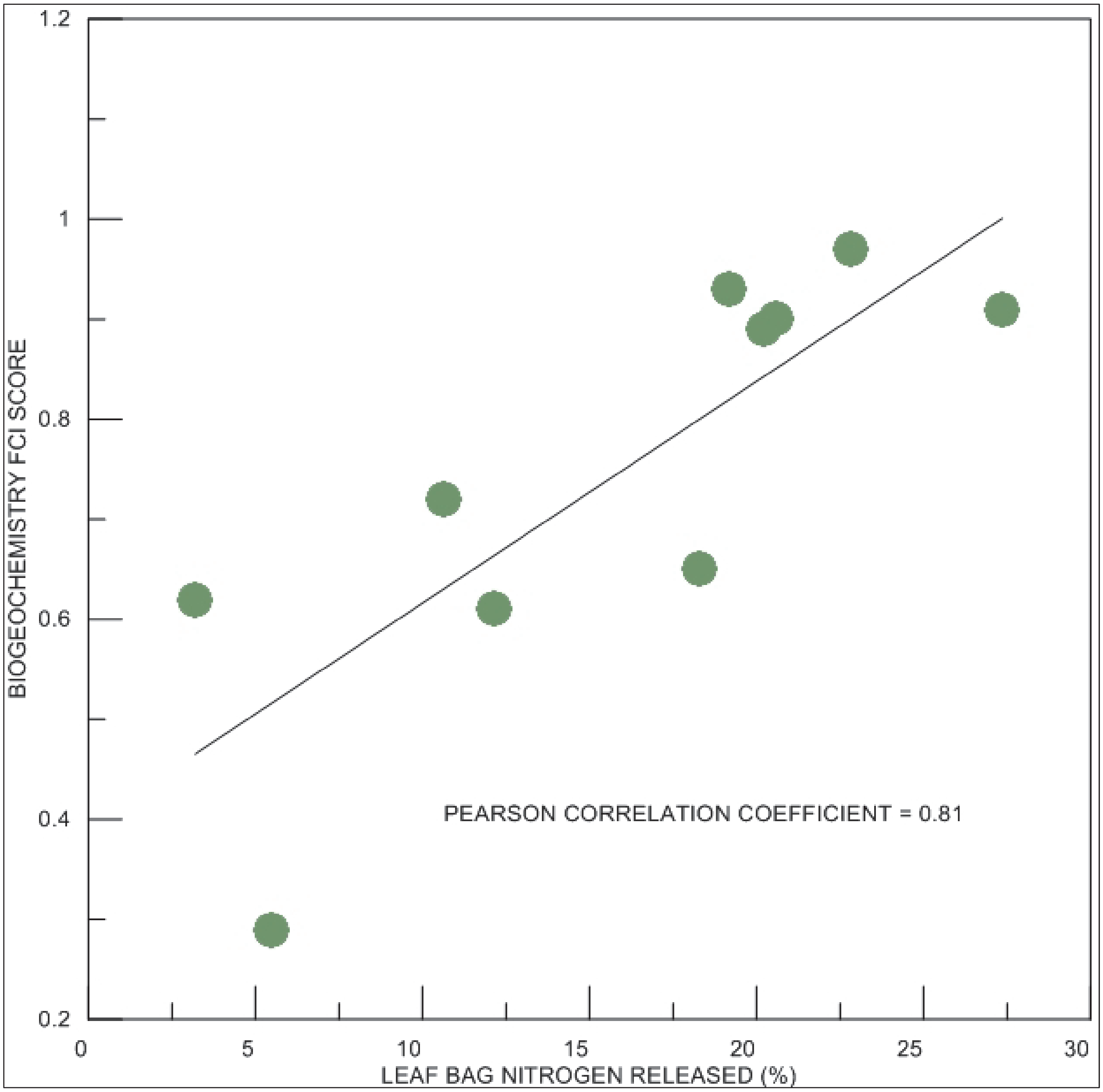


Table 4.3.2. Pearson Correlation Coefficients. Correlation is significant at the 0.01** and 0.05* level.

Function	Statistic	Mass decomposed (%)	C release (%)	N release (%)
Biogeochemical Cycling FCI Score	Pearson Correlation	.636*	.674**	.812**
	Significance (1-tailed)	.024	.016	.002
	N	10	10	10

Figure 4.3.8. Leaf bag nitrogen release compared with HGM Biogeochemical Cycling FCI score.



4.3.4 Summary

Leaf litter bag results provide evidence that the HGM Biogeochemical Cycling model differentiates between sample locations displaying a range of watershed and riparian/buffer zone characteristics. Validation study sites displaying increased leaf decomposition, C release, and N release exhibit mature second growth forest structure and watersheds lacking significant alterations (i.e.; impervious surface, open ground). The observed increases in C release and leaf processing agree with findings from Fennessey et al. (2008) and Atkinson and Cairns (2001) who reported higher rates of decomposition in older created and natural systems compared to recently created and highly altered areas. Study sites with lower leaf processing rates, C release, and N release included areas lacking a developed forest structure and containing fill or open ground within the watershed.

4.4 Stream loading

4.4.1 Rationale for selecting the independent measure

The concentration and loading of nutrients and sediments serve as a proxy measurement for the degree of biological, hydrologic, and morphological processing occurring within a stream catchment (Roberts and Mulholland 2007). Headwater streams contribute to nutrient processing, attenuation, and delivery to downstream environments (Peterson et al. 2001; Wipfli et al. 2007).

Several studies (Lowrance et al. 1984; Osborne and Kovacic 1993; Richards et al. 1996) link natural and anthropogenic disturbance regimes to a change in the loading of nutrients and sediments within headwater stream systems. Generally, findings suggest that higher levels of agriculture, urban, and forestry inputs within headwater catchments increase loading of nutrients (e.g., nitrogen and phosphorus) and sediments to downstream environments (Gurtz et al. 1980; Likens et al., 1970). As a result, examining the area-weighted loading of nutrients and sediments from headwater stream ecosystems provides an independent measure of biogeochemical cycling across an alteration gradient of sample locations.

Houser et al. (2006) reported relationships between the percentage-altered area within a headwater catchment and the amount of suspended solids, dissolved organic carbon, soluble reactive phosphorus, and ammonia discharged from the system (R^2 = 0.32 – 0.79). The outputs of the HGM model (FCI scores based on structural parameters measured within the project catchment) represent a surrogate measure of catchment alteration because HGM scores are scaled to the least-altered reference standard ecosystems found within the study area (Brinson 1993, 1995).

Examining relationships between model outputs and measurements of stream loading provides independent measures of biogeochemical functional capacity (Hanson et al. 2004). The loading of suspended solids has been linked to nutrient transport (Horowitz 2008), providing an additional measure of biogeochemical cycling as many nutrients are transported in association with particulates. Further, water quality parameters (e.g., specific conductivity, pH) provide insight into biogeochemical cycling and the availability of many materials, including nutrients and trace elements, that often depend on chemical conditions within the water column (Reddy and DeLaune 2008).

The following section presents the results of water quality parameters gathered throughout the study period. The ability of HGM model outputs to differentiate between sample locations exhibiting varying levels of alteration is evidence that the rapid assessment approach operates as intended.

4.4.2 Methods

Water sampling occurred throughout the study period (March 2011 – July 2012). Water samples were collected directly upstream of the trapezoidal flume (Figure 1.5.1, Figure 4.4.1). Due to the ephemeral nature of the study sites (i.e., water does not flow year round), following a set water sampling schedule was impracticable. Therefore, sampling took place whenever flow occurred, with emphasis placed on sampling a variety of flow regimes (i.e., high, moderate, and low flow conditions) to capture the range of flow at each sample location. Staff scheduled sampling events based on local rainfall patterns, returning to each site as frequently as possible (Figure 4.4.2). The availability of water within individual study sites determined the number of times each site was sampled. For example, water flowed more often in sites 5, 8, and 9 and therefore underwent sampling and analysis more often during the study period.

The water quality parameters examined during the study period included: inorganic nitrogen (i.e., nitrate, nitrite, and ammonia), total kjeldahl nitrogen, total organic carbon, total phosphorus, total suspended solids, specific conductivity, and pH. Table 4.4.1 presents the methods utilized for each analyte, detection limit, and reported units. All laboratory analysis occurred at a certified facility following published quality assurance and quality control procedures. Specific conductivity (conductivity) measurements occurred in the field utilizing automated data loggers. Additionally, conductivity measurements were examined for each collected water sample returned to the laboratory.

Figure 4.4.1. Water sampling occurred throughout the study period under a variety of conditions, including (clockwise from top left): 1) high flow, 2) moderate flow, and 3) low-flow conditions; 4) water samples could not be collected under very low flow conditions.

The calculation of stream loading has been widely applied to studies of stream water quality and biogeochemical cycling (Vanderbilt et al. 2003; Campbell et al. 2004) providing a method to estimate the total output of a component from a catchment. As a result, loading measures provide a better measure than traditional concentration measurements (Horowitz 2008). Loading data can also account for catchment area, allowing for comparison of results across varying landscape scales. Generally, loading calculations interpolate between instantaneous data points to estimate stream loading when concentration data collection occurs infrequently (Hope et al. 1997). The loading calculations applied in the current study employed the “Method 5” equations of Verhoff et al. (1980) and Walling and Webb (1985), as recommended by Littlewood (1992), when continuous discharge data is available (Equation 4.4.1).

$$\text{Load per unit area} = \left[K \cdot Qr \cdot \sum_{n}^{i=1}[CiQi] / \left(\sum_{n}^{i=1} Qi\right)\right] / A \qquad (4.4.1)$$

where K represents a conversion factor for the period of record, Qi is the instantaneous discharge at the time of sampling, Ci is the instantaneous analyte concentration, Qr is the mean discharge for the period of record, n is the number of samples, and A represents the headwater catchment area. Loading rates were calculated for both nutrients and suspended sediment loads.

Figure 4.4.2. Distribution of water samples collected and analyzed during the study period. Study sites exhibiting more sampling events contained flowing surface water more frequently.

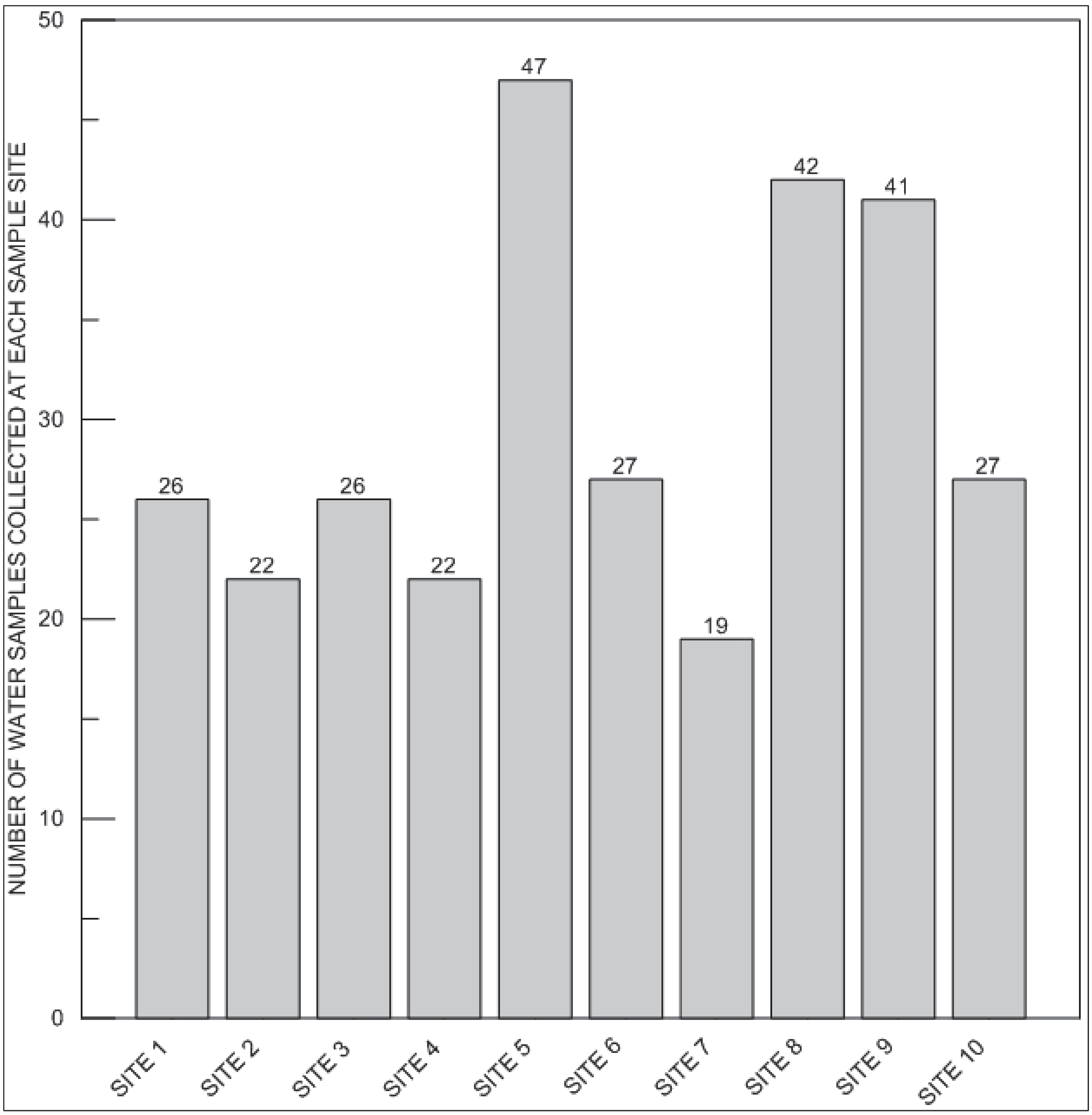


Table 4.4.1. Stream water quality parameters, methods, detection limits, and reported units.

Parameter	Method	Detection limit	Reported units
Ammonia as N	HACH 8038	0.02	mg/L
Nitrate and Nitrite as N	EPA 353.2	0.01	mg/L
Total Kjeldahl Nitrogen	SM4 500NorgC+ HACH 8038	0.02	mg/L
Total Organic Carbon	SM 5310C	0.50	mg/L
Total Suspended Solids	SM 2540D	2.0	mg/L
Total Phosphorus	AWWA 4500-P	0.01	mg/L
Specific Conductivity	EPA 120.1	N/A	*u*S/cm
pH	SM 4500H+ B	N/A	pH units

Analysis of pH and conductivity examined averages, maximum observed values, and the number of events exceeding a proposed standard for stream conductivity (USEPA 2011). The proposed standard is used as a basis or reference point only, allowing for comparison between sites. The proposed standard was selected because other numerical threshold conductivity standards were not found within the study area. Soil temperature measurements utilized TidbiT data loggers (Onset Corporation; Bourne, MA) placed 10 cm below the soil surface and temperature was recorded every 8 hours. Daily temperature ranges were averaged for the entire study period.

Statistical procedures included normality testing applying the Shapiro-Wilk test ($p>0.05$ indicates a normal distribution) followed by Pearson Product Moment Correlation analysis. In cases where data deviated from normal distributions, Spearmans Rank Order Correlation was applied (SPSS IBM, Inc. Version 20). Significance was determined at the $\alpha = 0.05$ level. Three outliers were removed from analysis based on extreme values significantly higher than all other observed dataset values collected across all other sample locations. Within the inorganic nitrogen loading data, one highly altered reach subjected to valley filling, displayed levels >12 times higher than all other observed values. Within the total phosphorus loading data, an altered reach subjected to ongoing agricultural inputs, including cattle directly accessing the stream above the water sampling location, displayed levels >74 times higher than all other values. In the case of maximum conductivity, one outlier was removed based on a value >4 times higher than all other values.

4.4.3 Results

Total inorganic nitrogen loading ranged from 0.92 – 75.3 kg/ha with an average of 9.8 kg/ha (Figure 4.4.3). Values observed for nitrate loading correspond well with the findings presented in Groffman et al. (2004) who reported values between 0.11 and 4.8 kg nitrate/ha/yr in forested watersheds; current study values range from 0.37 – 3.3 kg nitrate/ha/yr in forested watersheds. Site 6, a highly altered area which was formerly mined and subjected to valley filling, displayed levels >12 times higher than all other values and was removed from statistical analysis as an outlier, resulting in normal distribution for the remaining samples (Table 4.4.2). Comparisons of total inorganic nitrogen loading with predicted results of the HGM Biogeochemical Cycling FCI score yielded significant negative correlations (r = -0.696, n = 9, P = 0.019), supporting the current HGM model (Figure 4.4.4).

Organic nitrogen loading (represented by Kjeldahl nitrogen) ranged from 0.52 – 10.2 kg/ha with an average of 2.3 kg/ha (Figure 4.4.5). Data were not normally distributed (Table 4.4.2). Comparisons of organic nitrogen loading with predicted results of the HGM Biogeochemical Cycling FCI score yielded significant correlations (r_s = -0.624, n = 10, P = 0.027; Figure 4.4.6). The distribution of data is dominated by two highly altered study sites subject to valley filling and previous or ongoing mining activities. However, the ability of the HGM model to differentiate between mature forested sites and altered sites promotes confidence in the appropriateness of the model configuration based on the weight of evidence approach applied. For example, all sample locations characterized by HGM assessment scores >0.7 display organic nitrogen loading values <2.0 kg/ha.

Dissolved organic carbon loading results exhibited similar patterns to the organic nitrogen values observed above. Values ranged from 6.2 – 122 kg/ha with an average of 38.3 kg/ha (Figure 4.4.7). Data were not normally distributed (Table 4.4.2). Comparisons of dissolved organic carbon loading with the HGM Biogeochemical Cycling FCI score were not significantly correlated (r_s = -0.394, n = 10, P = 0.130; Figure 4.4.8). The distribution of data is dominated by two highly altered study sites subject to valley filling and previous or ongoing mining activities; however, the ability of the HGM model to differentiate between mature forested and altered sites promotes confidence in the appropriateness of model configuration.

Figure 4.4.3. Total inorganic nitrogen loading at each sampling site. Note the broken axis utilized to accommodate the outlier value observed at Site 6.

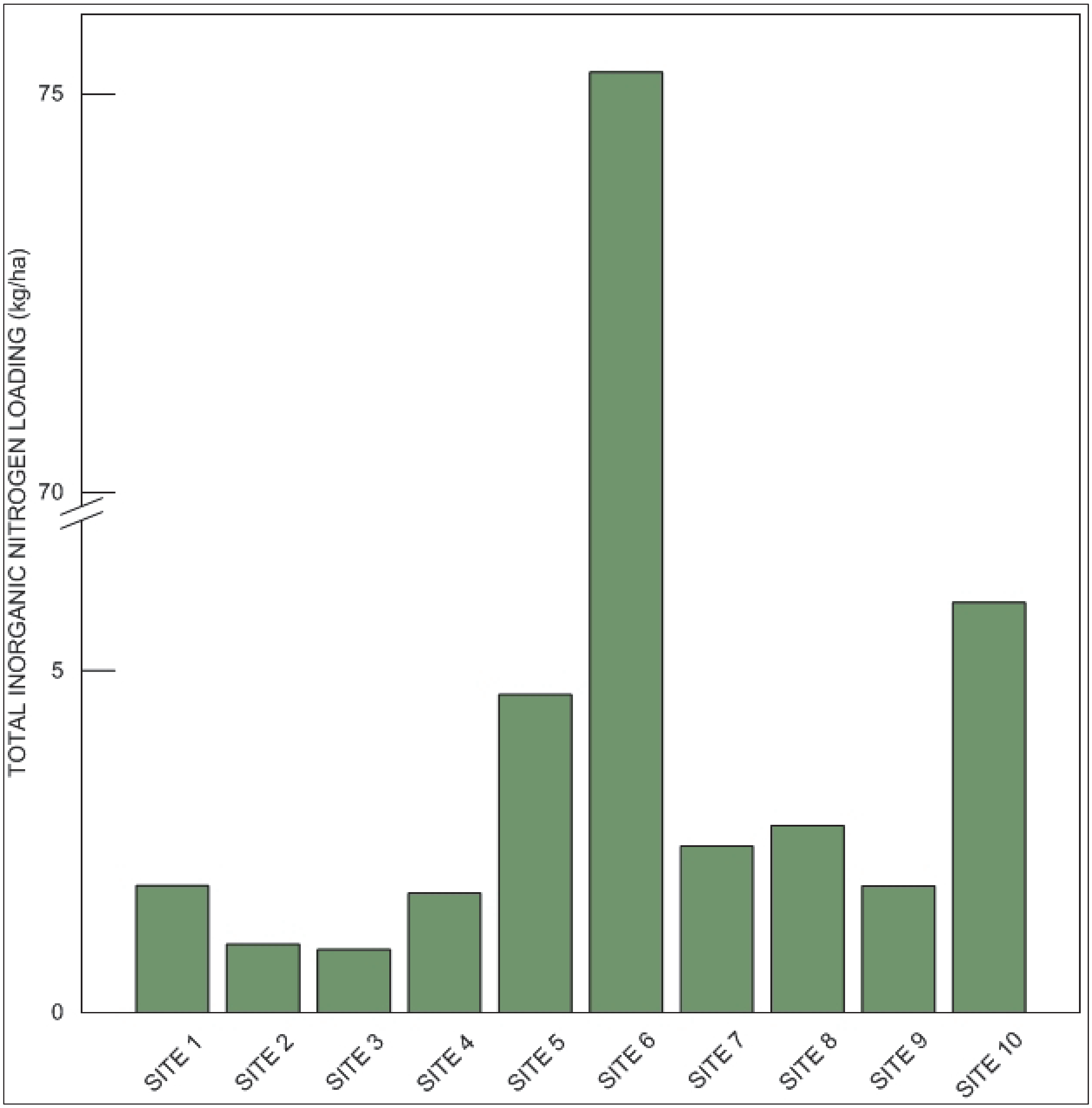


Table 4.4.2. Tests of Normality – Nutrient and suspended sediment, loading, and maximum conductivity.

Parameter	Shapiro-Wilk W	df	p
Inorganic nitrogen loading (g/ha)	.850	9	.075
Kjeldahl nitrogen loading (g/ha)	.654	10	.001
Dissolved organic carbon loading (kg/ha)	.694	10	.001

Parameter	Shapiro-Wilk		
	W	df	p
Total phosphorous loading (kg/ha)	.859	9	.094
Total suspended solids loading (kg/ha)	.871	10	.103
Maximum conductivity (uS/cm)	.886	9	.181
Soil Temperature, daily range (oC)	.856	10	.052

Figure 4.4.4. Total inorganic nitrogen loading at each sampling site compared with HGM Biogeochemical Cycling FCI score.

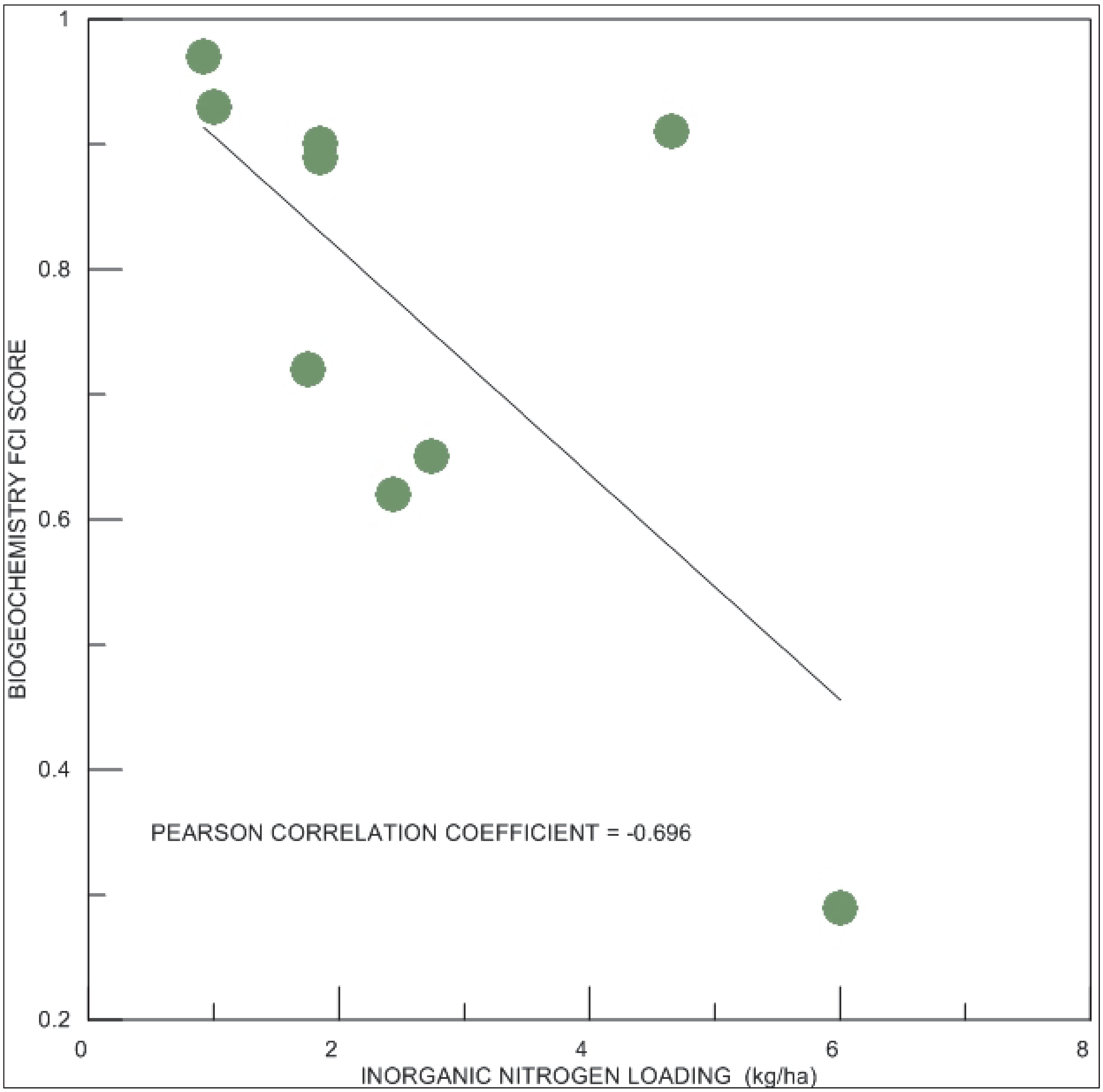


Figure 4.4.5. Total Kjeldahl nitrogen loading at each sampling site.

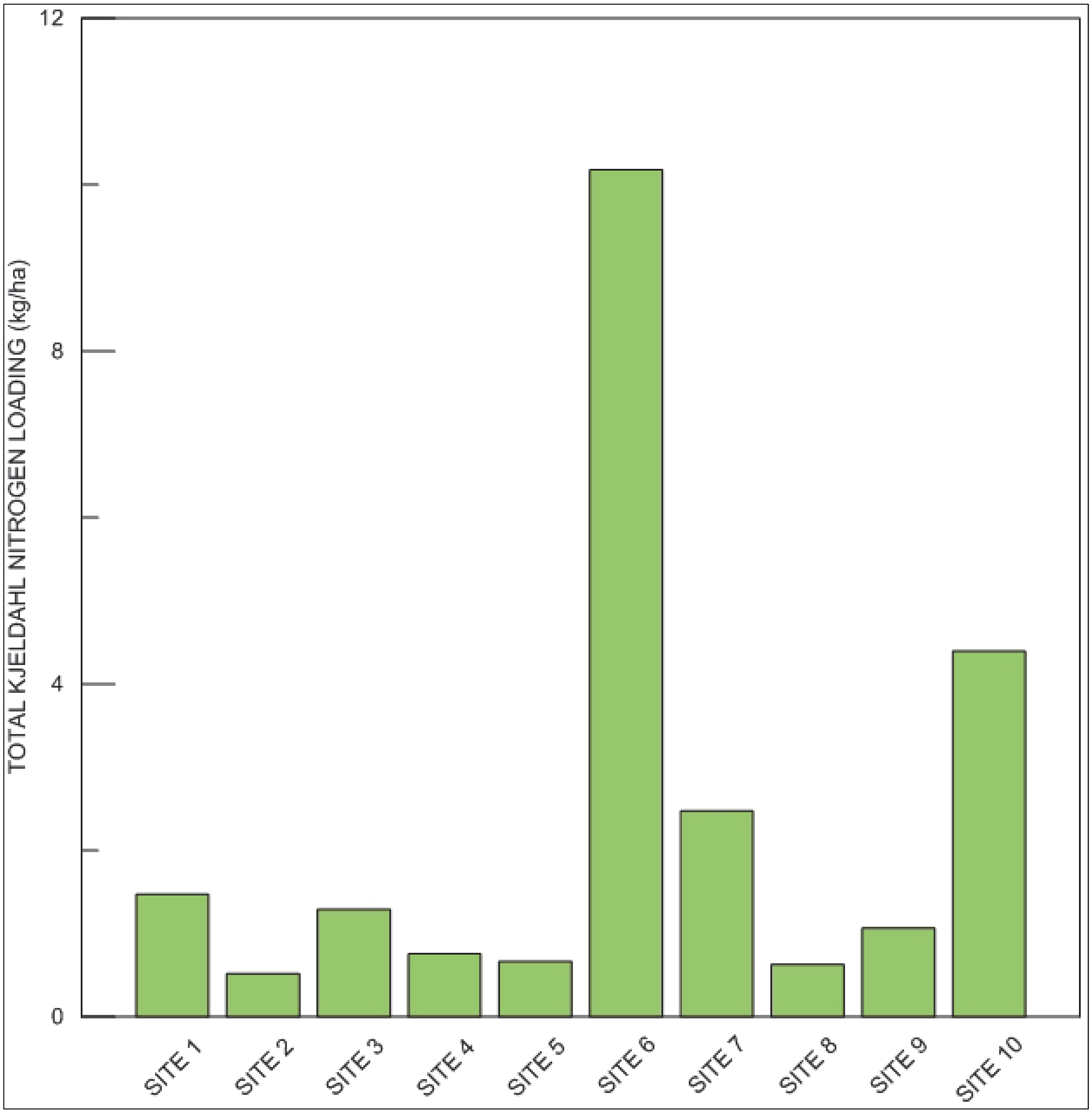


Figure 4.4.6. Kjeldahl nitrogen loading at each sampling site compared with HGM Biogeochemical Cycling FCI score.

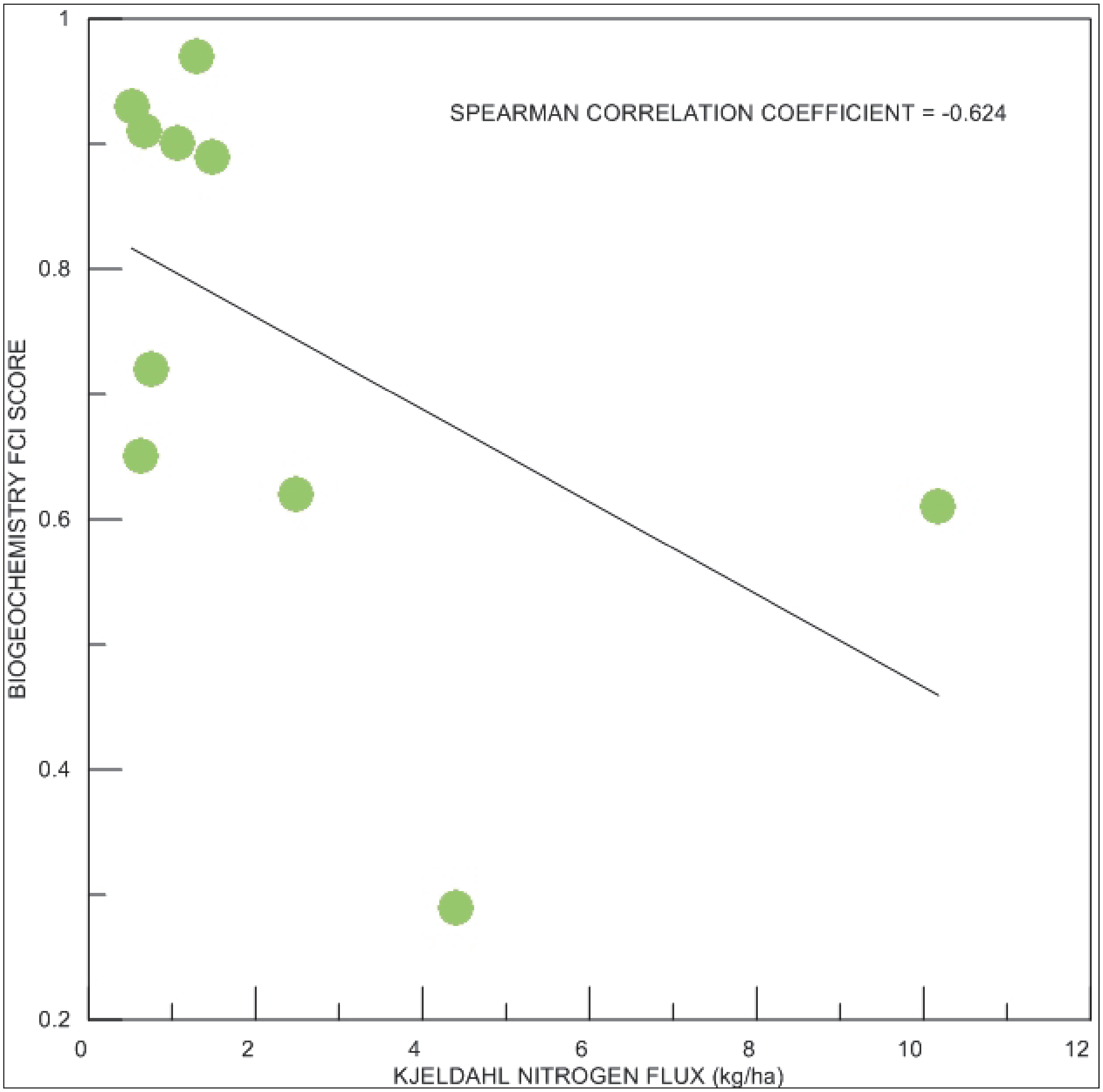


Figure 4.4.7. Dissolved organic carbon loading at each sampling site.

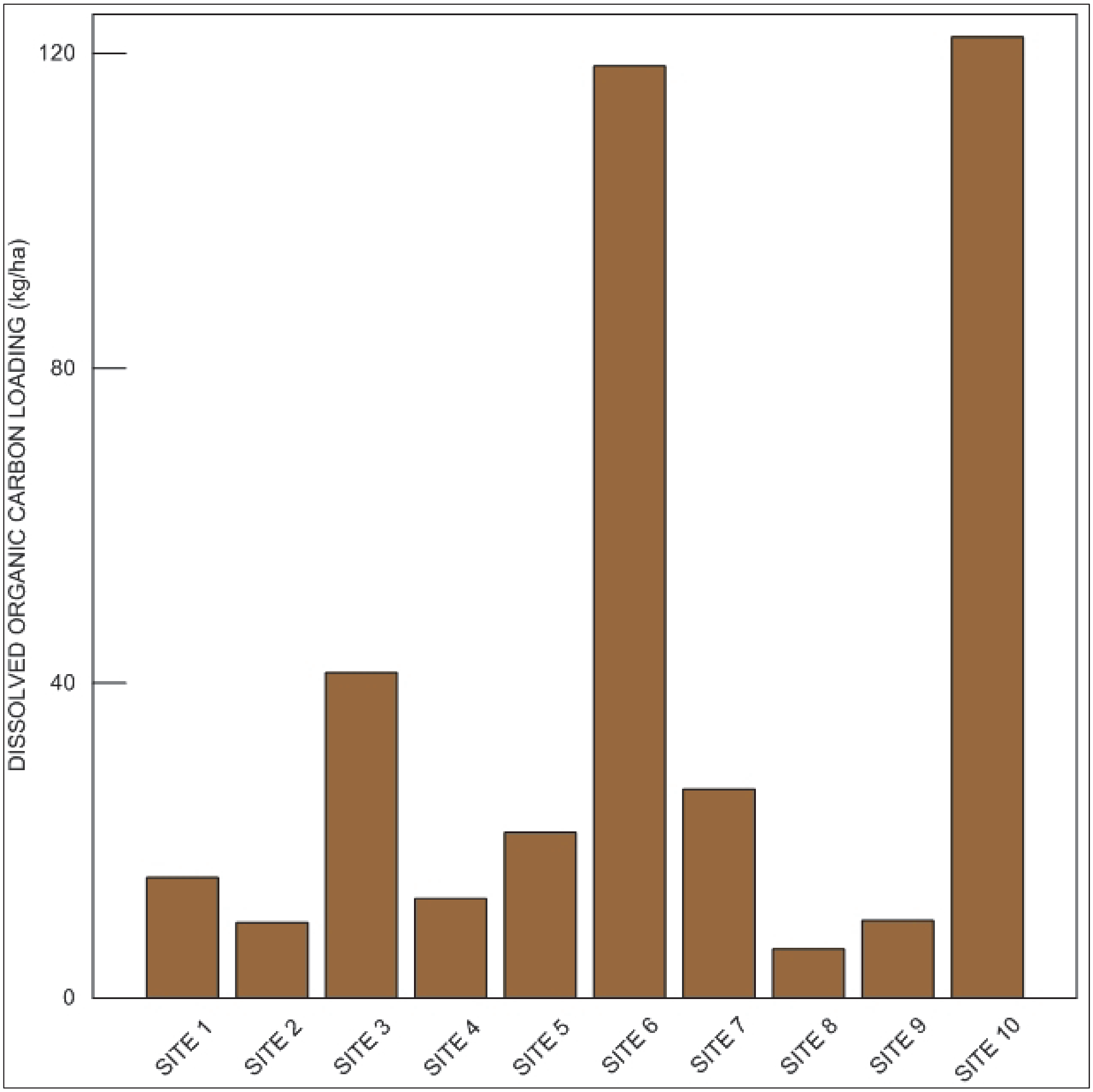
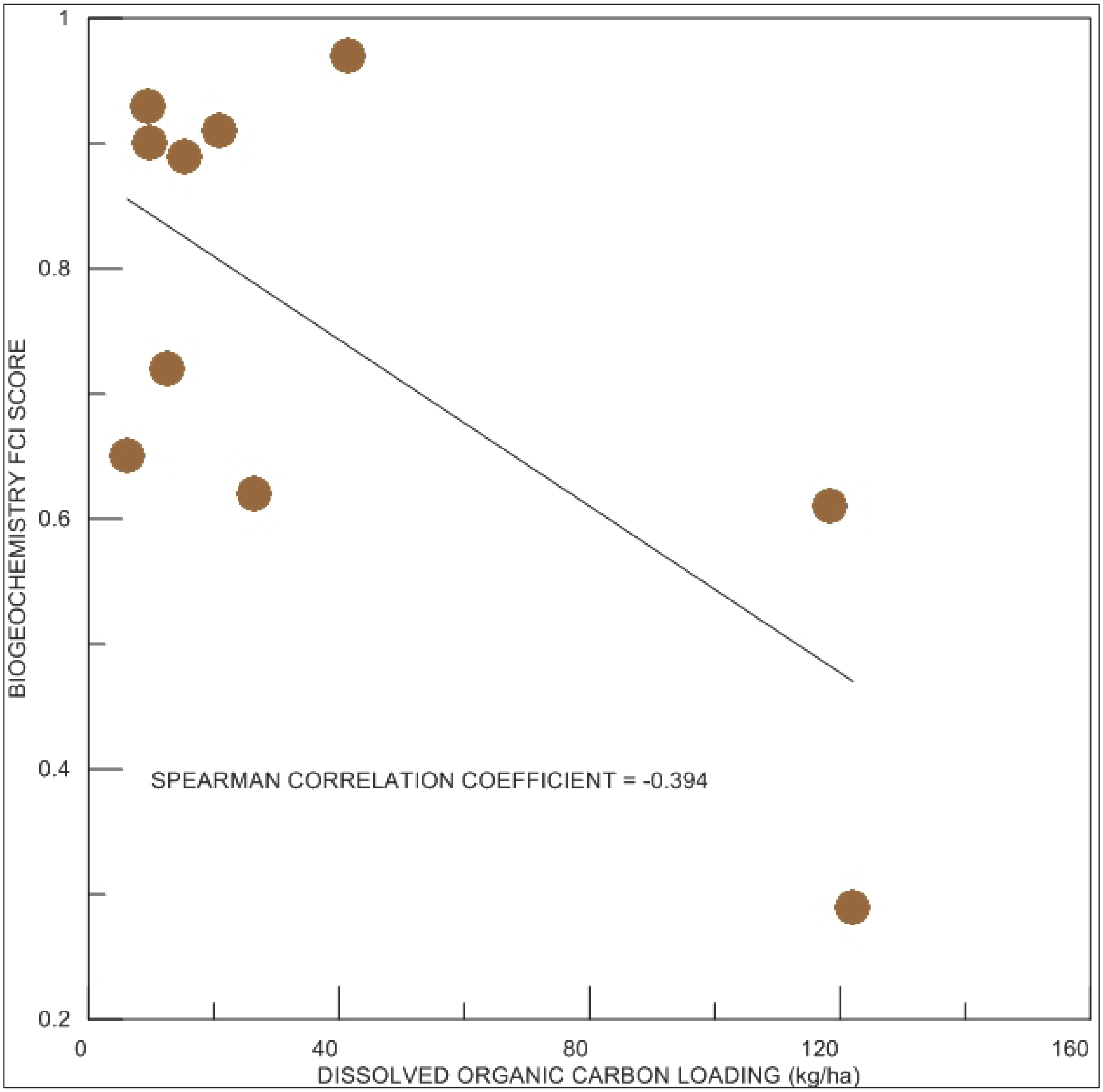


Figure 4.4.8. Dissolved organic carbon loading at each sampling site compared with HGM Biogeochemical Cycling FCI score. The correlation was not statistically significant.



Total phosphorus loading ranged from 0.045 – 36.8 kg/ha, with an average of 3.87 kg/ha (Figure 4.4.9). Site 7, an altered area subject to ongoing agricultural inputs including cattle directly accessing the stream above the water sampling location, displayed levels >74 times higher than all other values. As a consequence, Site 7 was removed from statistical analysis as an outlier, resulting in normal distribution (Table 4.4.2). Comparisons of total phosphorus loading with predicted results of the HGM Biogeochemical Cycling FCI score yielded significant correlations (r = -0.661, n = 9, P = 0.026), supporting the current HGM model (Figure 4.4.10).

Figure 4.4.9. Total phosphorus loading at each sampling site. Note the broken axis utilized to accommodate the outlier value observed at site 7.

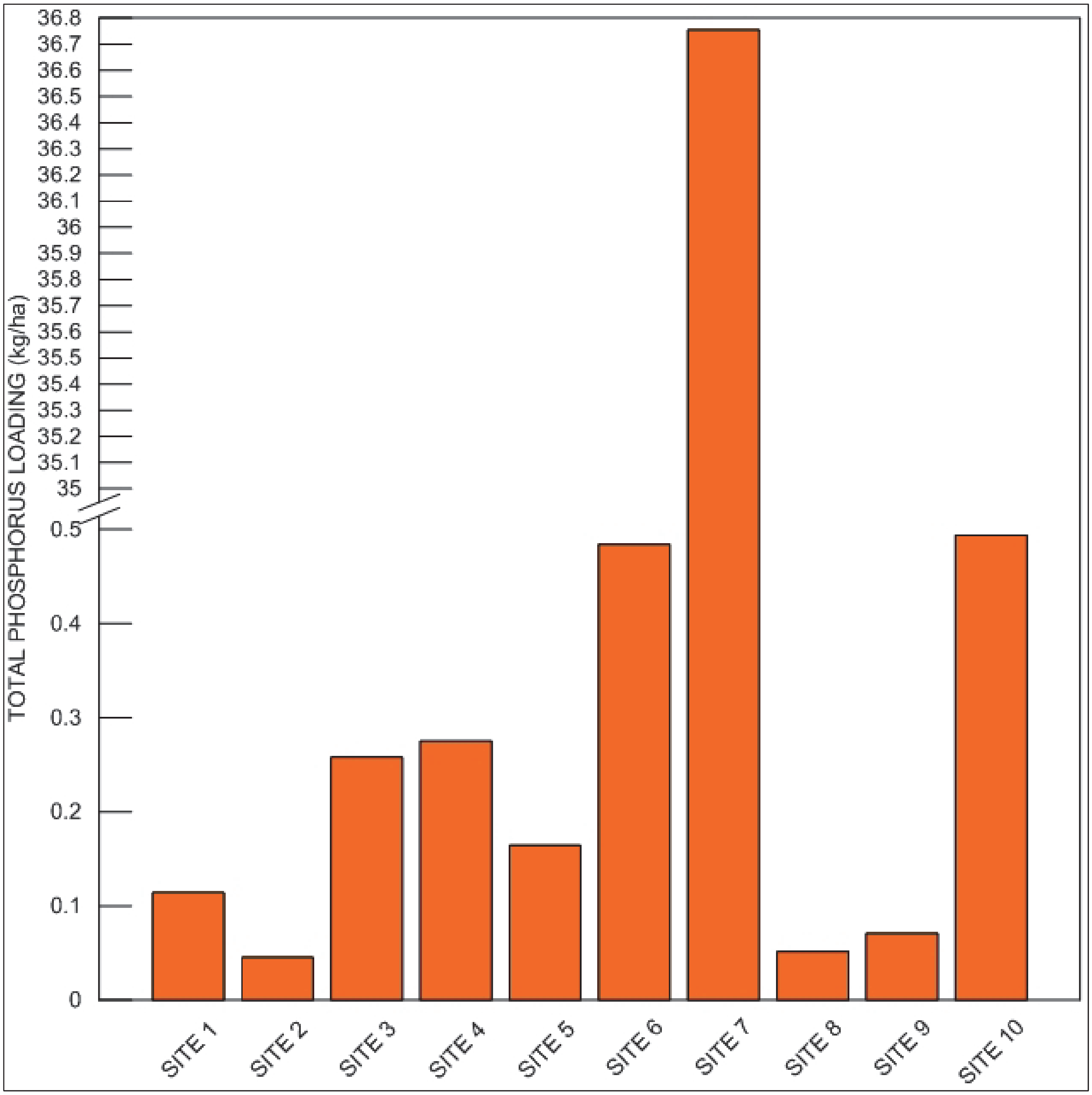
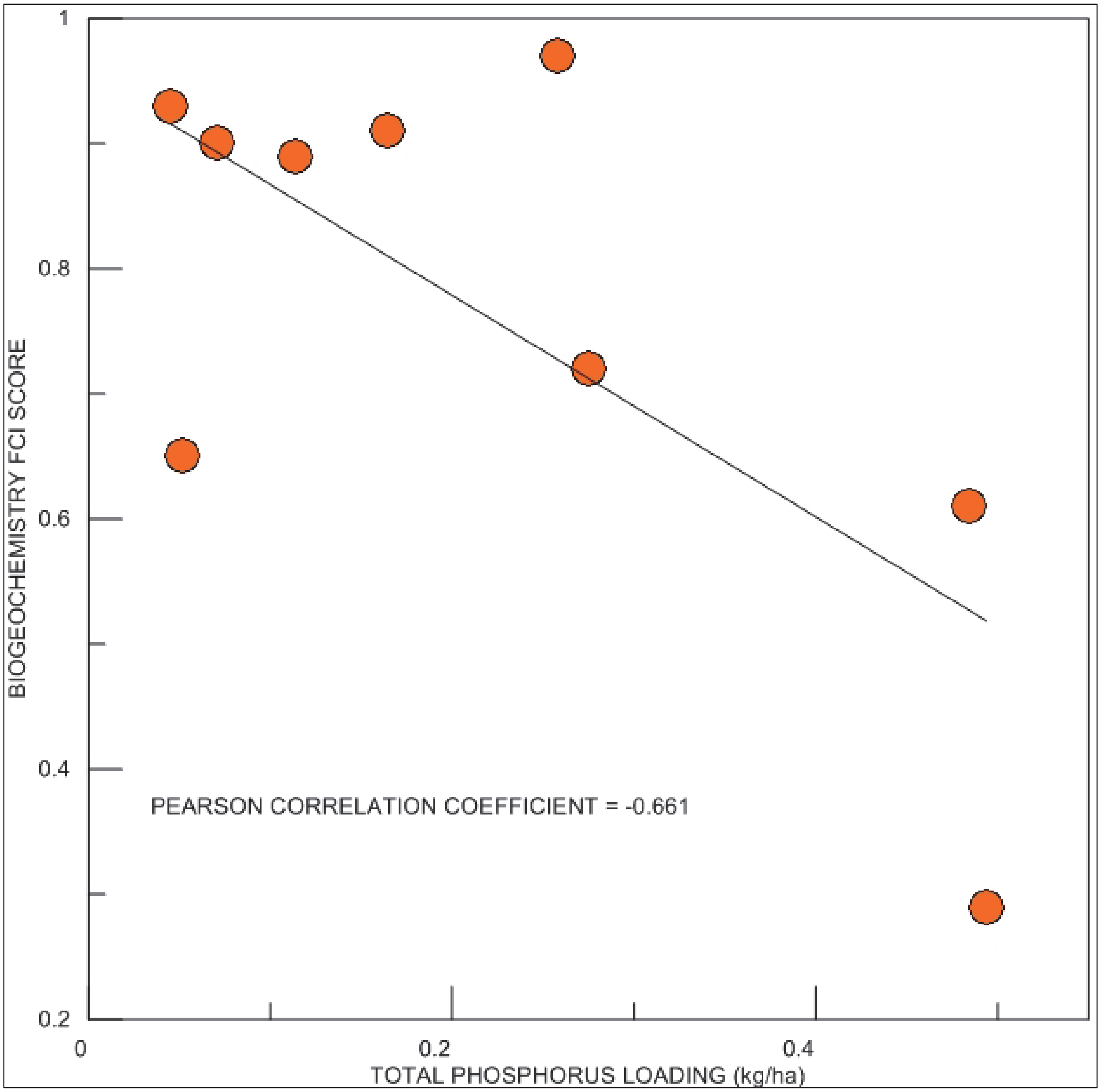


Figure 4.4.10. Total phosphorus loading at each sampling site compared with HGM Biogeochemical Cycling FCI score.



Total suspended solids loading ranged from 22.2 – 375 kg/ha with an average of 167 kg/ha (Figure 4.4.11). Data were normally distributed (Table 4.4.2). Comparisons of total suspended solid loading with the HGM Biogeochemical Cycling FCI score yielded significant correlations (r = -0.556, n = 10, P = 0.048; Table 4.4.3, Figure 4.4.12). These findings support the current HGM model.

Figure 4.4.11. Total suspended solids loading at each sampling location.

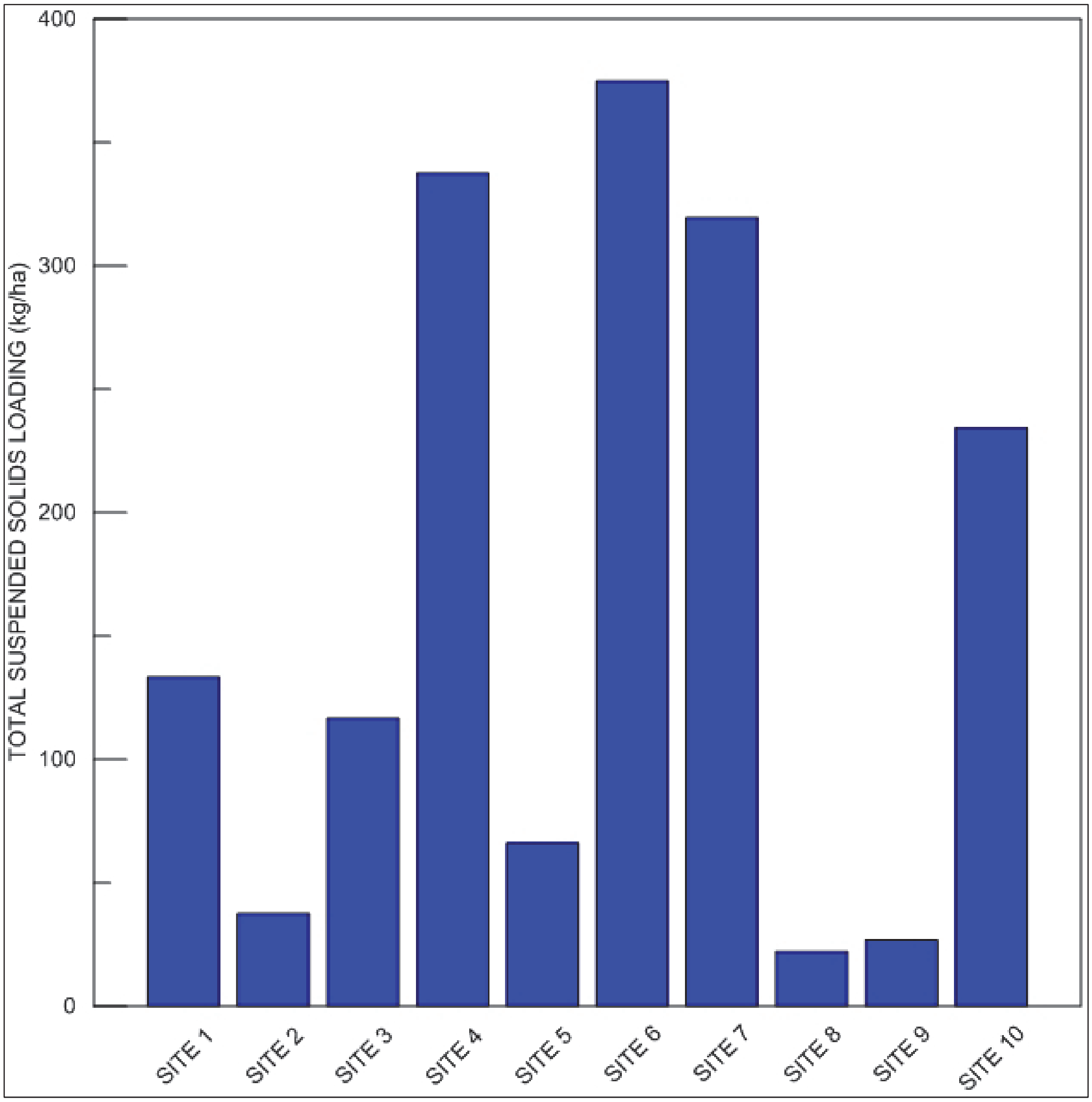
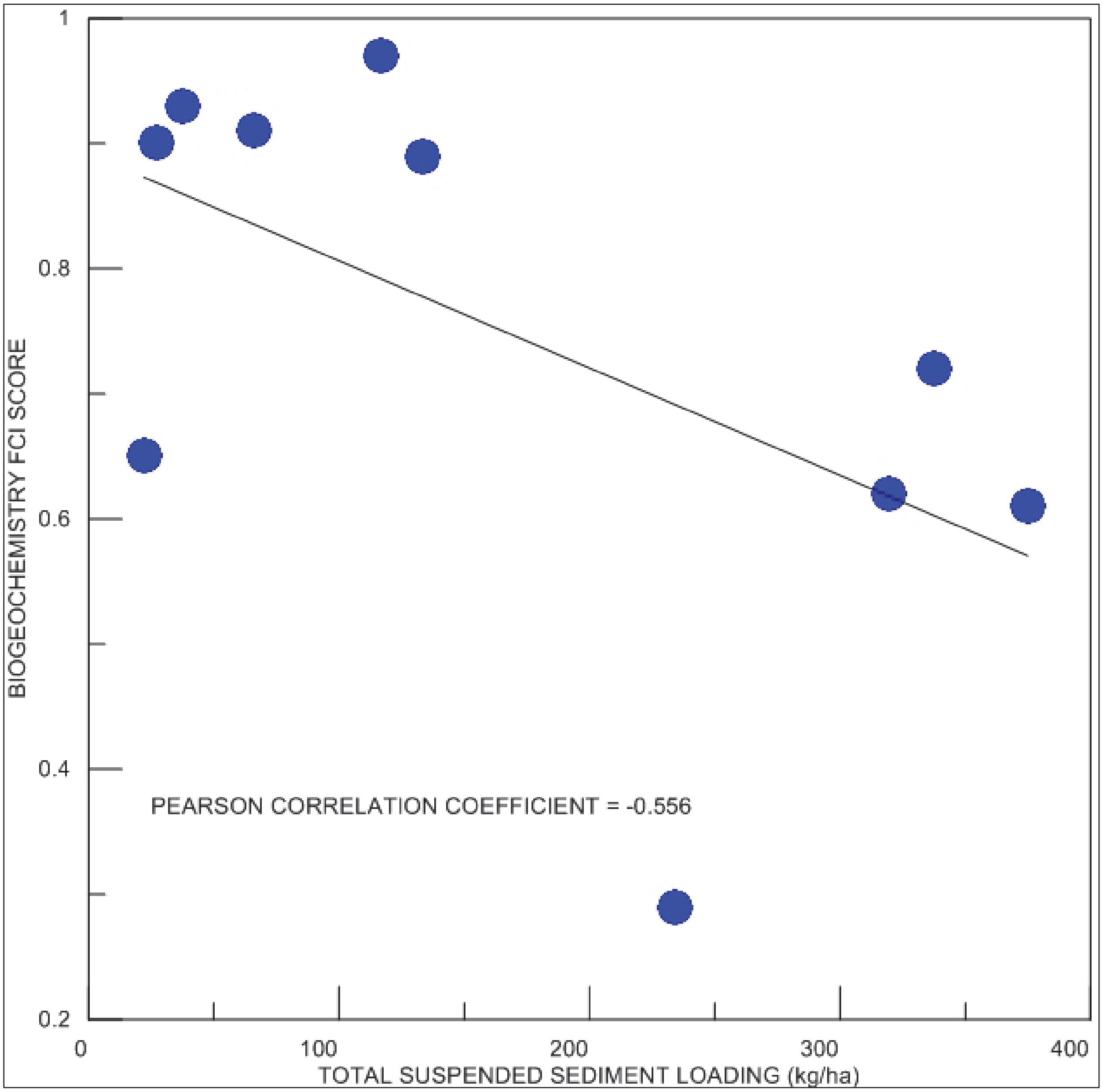


Table 4.4.3. Correlation Coefficients. Correlation is significant at the 0.05* level or the 0.01** level.

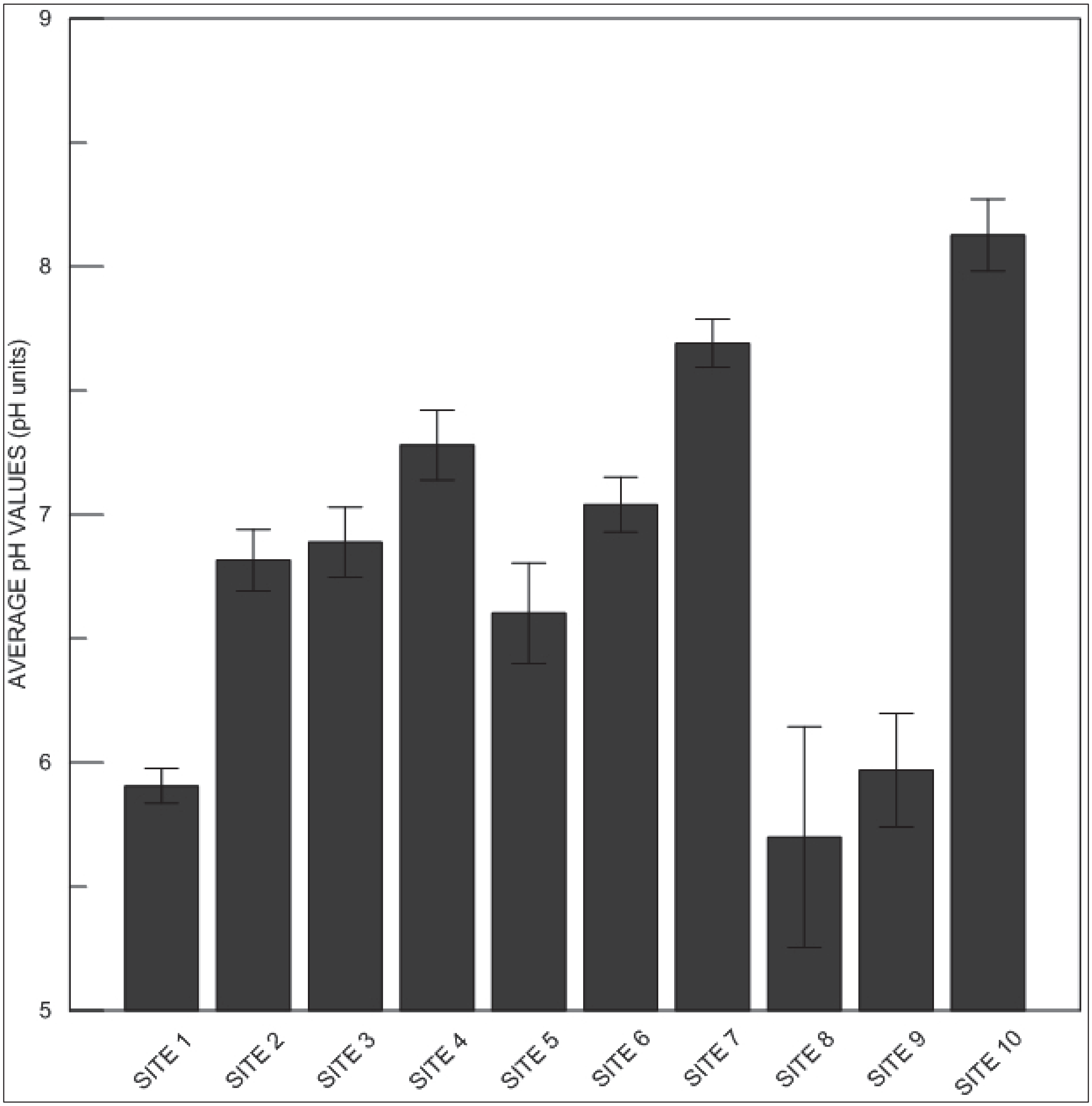
Parameter	Test	Correlation	N	*P*-value (1-tailed)
Inorganic nitrogen loading (g/ha)	Pearson	-.696*	9	.019
Kjeldahl nitrogen loading (g/ha)	Spearman	-.624*	10	.027
Dissolved organic carbon loading (kg/ha)	Spearman	-.394	10	.130
Total phosphorous loading (kg/ha)	Pearson	-.661*	9	.026
Total suspended solids loading (kg/ha)	Pearson	-.556*	10	.048
Maximum conductivity (*u*S/cm)	Pearson	-.693*	9	.019
Soil Temperature, daily range (^{o}C)	Pearson	-.812**	10	.002

Figure 4.4.12. Total suspended solids loading at each sampling site compared with HGM Biogeochemical Cycling FCI score.



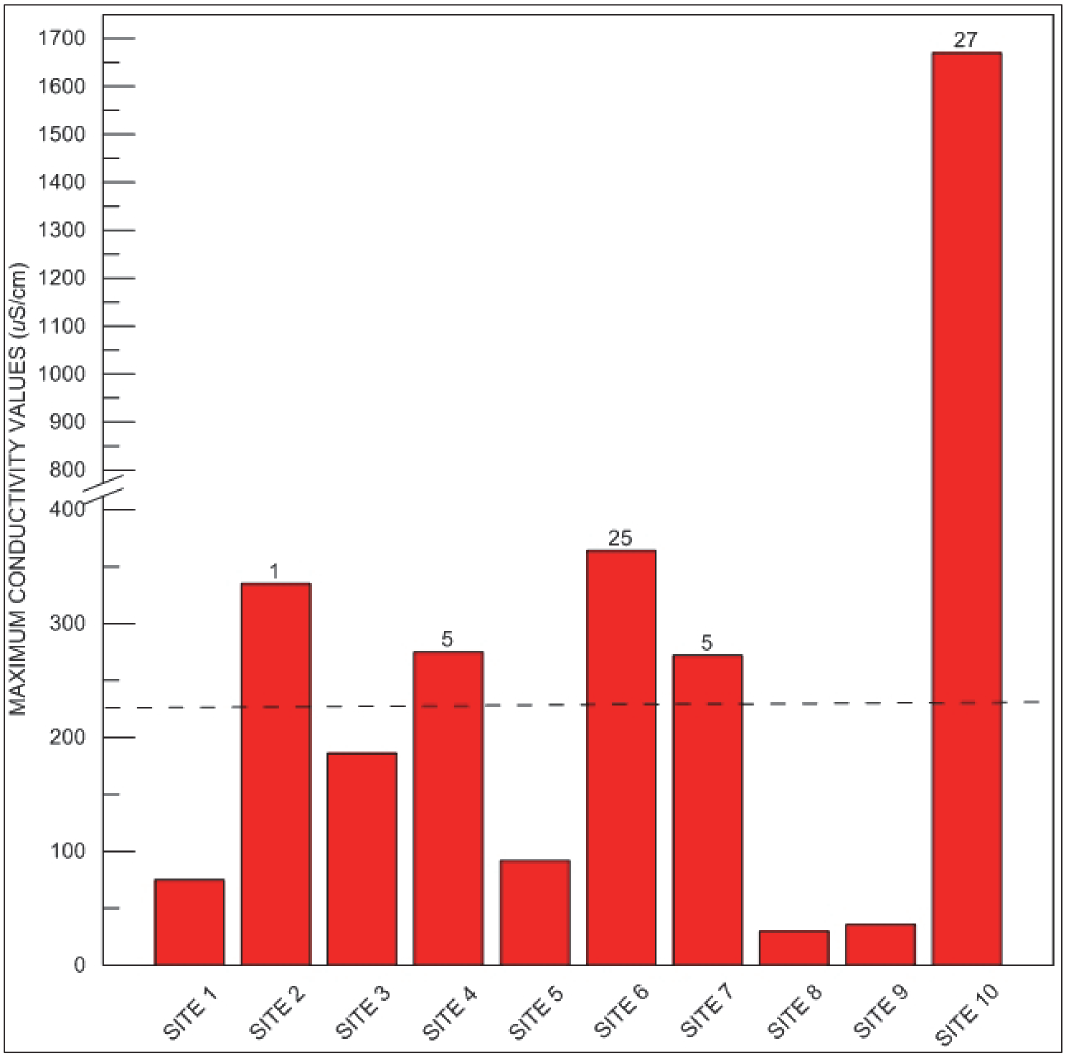
Hartman et al. (2005) and others have reported increases in headwater stream pH and conductivity resulting from alterations occurring within the study area. Collected data displays average pH values >7.0 in sampled locations impacted by valley filling, past or ongoing mining activities, and ongoing agricultural operations (Figure 4.4.13). Mature forested sample locations exhibit average pH values <7.0.

Figure 4.4.13. Average pH values observed at each sampling site. Error bars represent one standard deviation.



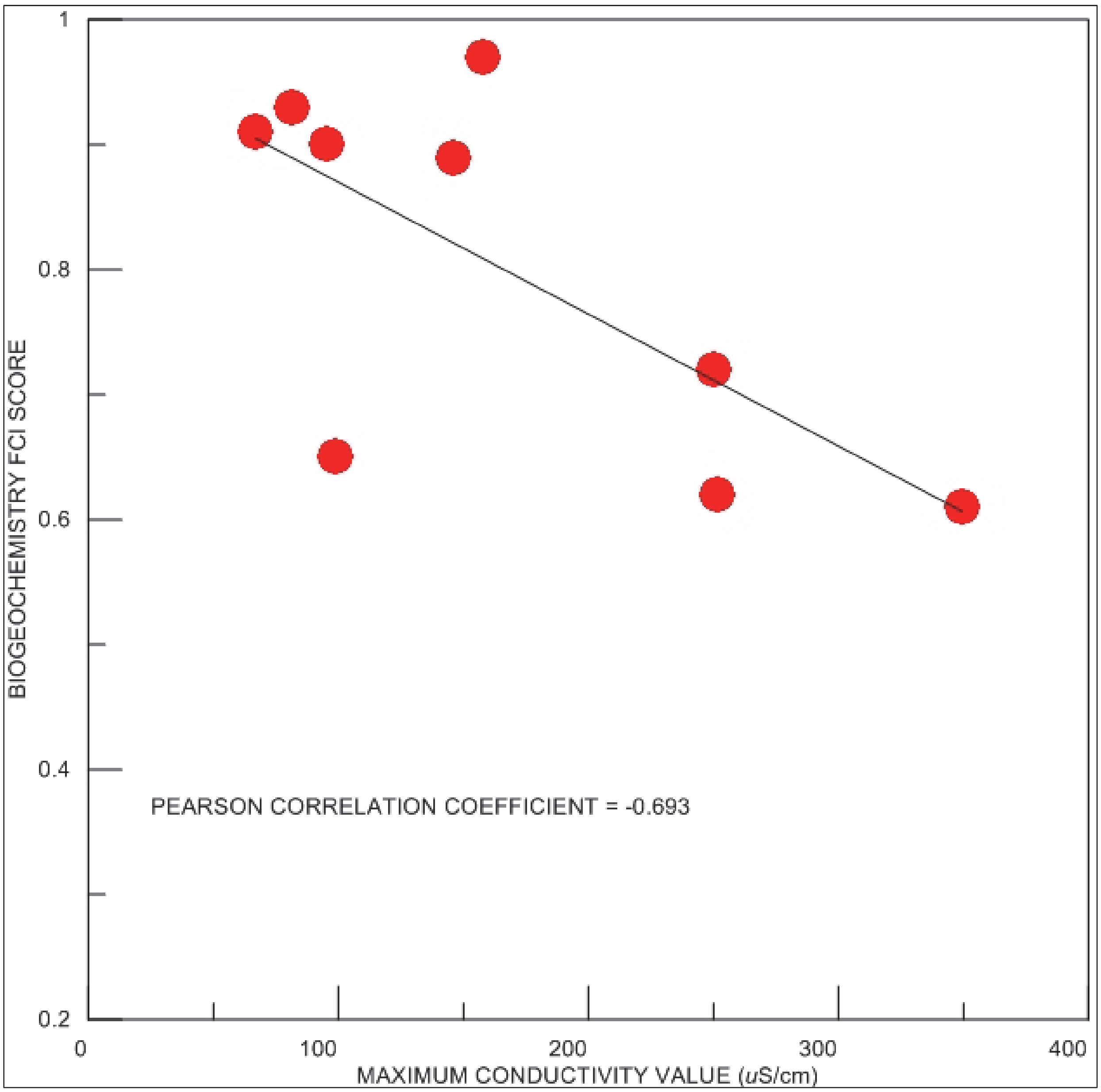
Conductivity values observed throughout the study ranged from 18 – 1670 *u*S/cm with an average of 199 *u*S/cm. Evaluating average conductivity values provides limited utility, and threshold or maximum values are often examined. The EPA threshold conductivity level is 300 *u*S/cm, with a lower 95 percent confidence band of 225 *u*S/cm (USEPA 2011). Conductivity values at several sites exceeded the proposed threshold, with sites characterized by lower HGM FCI scores exhibiting more high conductivity events (Figure 4.4.14). The proposed standard is used as a

Figure 4.4.14. Maximum conductivity values observed at each sampling site as measured using automated data loggers. Note the broken axis utilized to accommodate the outlier value observed at site 10. Column label values represent the number of sampling events during which conductivity levels surpassed the 225 *u*S/cm threshold proposed by EPA (threshold represented by the horizontal dashed line). The proposed standard is used as a reference point only, allowing for comparison between sites. The proposed standard was selected because other numerical threshold conductivity standards were not found within the study area.



basis or reference point only, allowing for comparison between sites. The proposed standard was selected because other numerical threshold conductivity standards were not found within the study area. Maximum stream conductivity values measured using automated data loggers ranged from 67 – 1458 *u*S/cm, with an average of 296 *u*S/cm. Lindberg et al. (2011) reported similar results with maximum conductivities of 253 uS/cm in unaltered Appalachian headwater streams, while catchments impacted by surface mining ranged from 502 – 2540 uS/cm. Site 10, an altered area subject to valley fill and previous mining impacts upstream of the water sampling location, displayed levels >4 times higher than all other values. As a consequence, site 10 was removed from statistical analysis as an outlier; the remaining data were normally distributed (Table 4.4.2). A comparison of maximum conductivity measured in the laboratory with predicted results of the HGM Biogeochemical Cycling FCI score yielded significant correlations (r = -0.693, n = 9, P = 0.019; Table 4.4.3). These findings support the current HGM model (Figure 4.4.15).

Figure 4.4.15. Maximum observed conductivity values in laboratory samples at each sampling site compared with HGM Biogeochemical Cycling FCI score.



Several published studies show that changes in temperature regimes of headwater streams are related to alterations to their contributing watersheds, in that greater ranges in temperature occur in streams with watersheds exposed to silvicultural, agricultural, and urban landuses (Swift and Messer 1971; Rischel et al. 1982). A number of aquatic and terrestrial processes associated with decomposition and nutrient cycling exhibit temperature dependence (Allan 2001). Thus, an increase in water temperature affects metabolic process rates and microbial community assemblages (Dodds 2002). The mean daily temperature range observed within soils in the riparian zone (at 10 cm depth) varied between 0.37 and 8.0 ^{o}C among study sites, with an average range of of 2.9 ^{o}C (Figure 4.4.16). Temperature range data were normally distributed (Table 4.4.2) and negatively correlated with HGM Biogeochemical Cycling FCI score (r = -0.812, n = 10, P = 0.002; Table 4.4.3). These findings demonstrate that the HGM model effectively differentiates between least-altered and altered reaches, where the daily range in soil temperatures are much more variable (Figure 4.4.17). Soils from riparian zones with lower daily temperature variations were associated with watersheds of closed-canopied forest; sites with higher daily temperature ranges in soil temperatures were associated with watersheds with decreased canopy cover.

4.4.4 Summary

Nutrient and sediment loading measurements, water pH and conductivity, and soil temperature provide evidence that the HGM Biogeochemical Cycling model differentiates between locations exhibiting a range of watershed, channel, and riparian/buffer zone characteristics. Validation study sites displaying lower nutrient and sediment loading rates, lower maximum conductivity, temperature ranges, and pH values are associated with mature, second-growth forest, watersheds lacking significant alterations to the surrounding watershed (i.e., impervious surface, open ground), and stream channel characteristics (e.g., embeddedness) near the range of values exhibited by reference standard sites. Higher nutrient and sediment loading rates occurred in reaches exhibiting alteration from forestry, mining, and agricultural impacts within the watershed and received low HGM scores. Additionally, low scoring study sites also displayed higher maximum conductivity values, pH values, and a greater daily range in soil temperatures. Study sites exhibiting few landscape alterations displayed physical characteristics within the range of reference standard reaches, and received higher HGM scores.

Figure 4.4.16. Mean daily range of soil temperature in the riparian zones of instrumented reaches. Error bars represent one standard deviation.

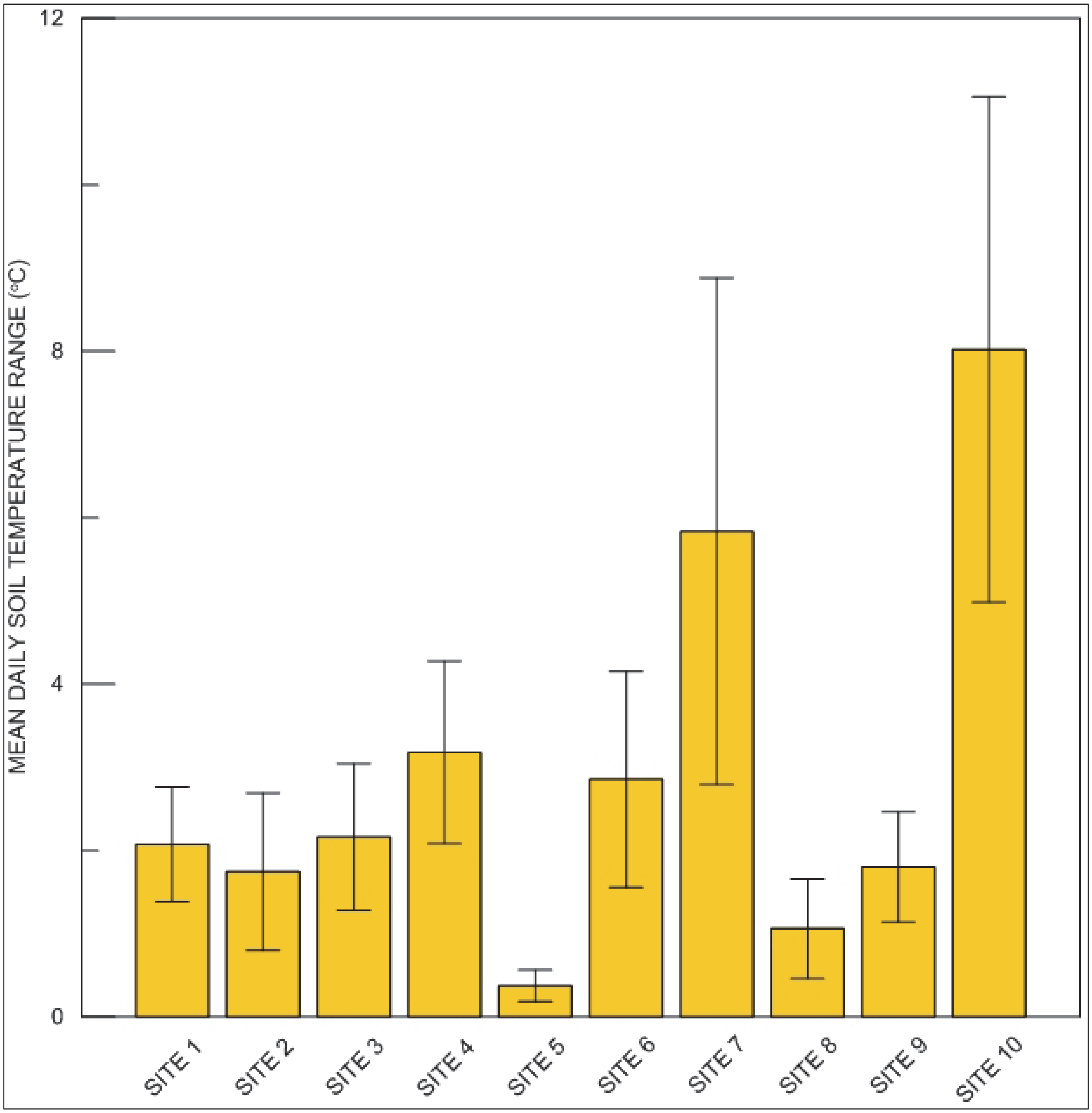
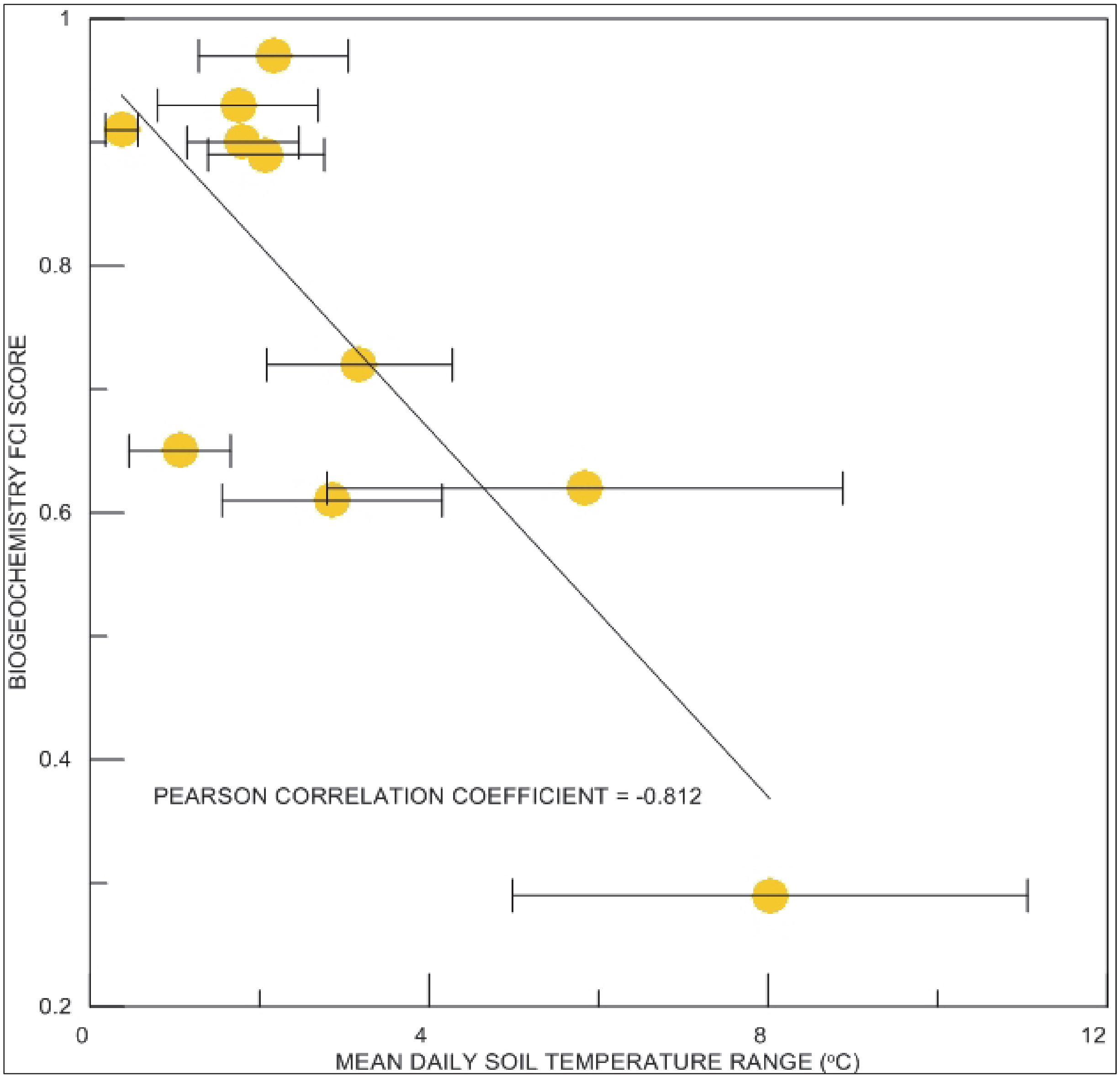


Figure 4.4.17. Mean daily range of soil temperature at each sampling reach compared with HGM Biogeochemical Cycling FCI score. Error bars represent one standard deviation.



4.5 Biogeochemical Cycling function sensitivity analysis

Sensitivity analysis results demonstrate that all variables incorporated into the Biogeochemical Cycling model exhibit appropriate relative influences on FCI scores. Appendix A presents sensitivity testing results for all variables, with only a representative subset examined in this section. Examples and data analysis are provided for one variable displaying a strong influence over FCI scores and one variable displaying a weaker impact on FCI scores.

In sites with canopy cover ≥20 percent (Equation 4.5.1), sensitivity analysis results demonstrate that stream particle embeddedness (V_{EMBED}) maintains the strongest effect on model FCI outputs (Figure 4.5.1; Table 4.5.1). When embeddedness values remain at the lowest allowable subindex score (0.1), the maximum attainable model FCI output equals 0.32 (Figure 4.5.2). When embeddedness values reach a maximum value of 1.0, the range of potential FCI values is 0.13 – 1.0. Increasing the embeddedness score incrementally from 0.1 to 1.0 while holding all other variables constant results in an increase in FCI scores ranging from 0 – 216%. The selection and scaling of sediment particle embeddedness as a controlling factor with the model is based on the potential of stream substrates to temporarily sequester organic matter and inflowing water, providing biologically reactive surfaces capable of supporting the occurrence of biogeochemical cycling.

$$FCI = \left\{ V_{EMBED} \times \left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{TDBH}}{3} \right) + V_{WLUSE}}{2} \right] \right\}^{1/2} \quad (4.5.1)$$

Figure 4.5.1. Conceptual distribution of variable influence on Biogeochemical Cycling FCI scores in areas displaying ≥20 percent canopy cover.

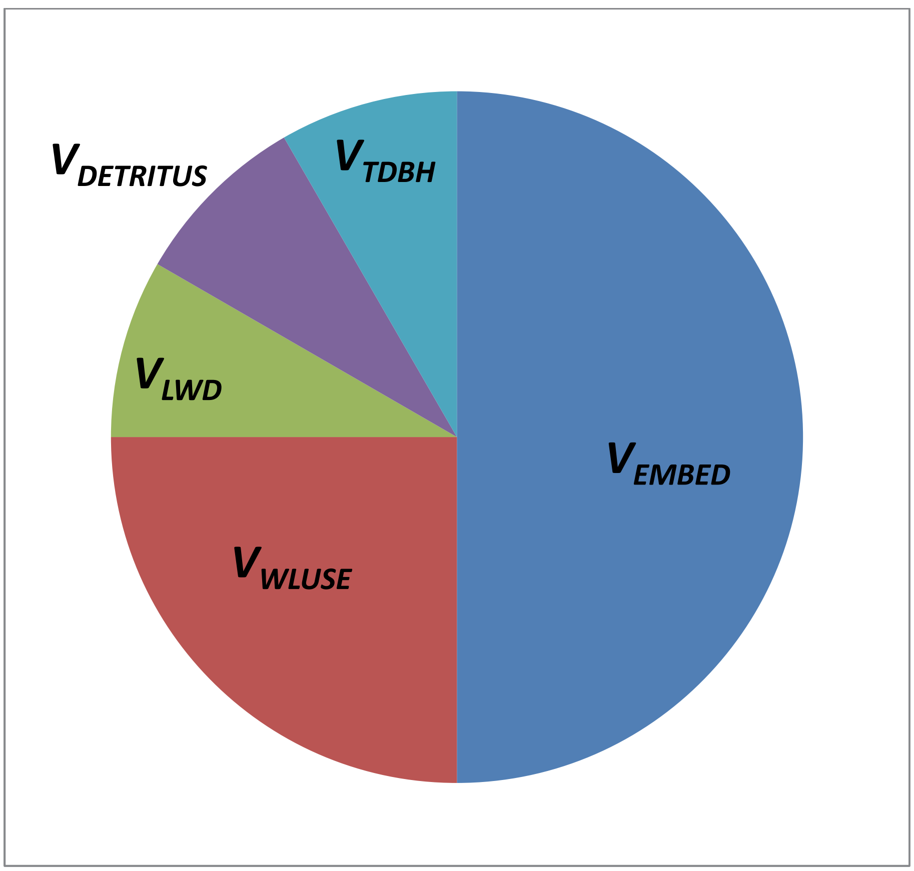


Table 4.5.1. Range of FCI scores attainable based in sensitivity analysis (≥20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.10, 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.09	0.04	0.13	0.22	0.10	0.32	0.34	0.16	0.50
V_{LWD}	0.09	0.04	0.14	0.07	0.09	0.16	0.08	0.23	0.31
$V_{DETRITUS}$	0.09	0.04	0.14	0.07	0.09	0.16	0.08	0.23	0.31
V_{TDBH}	0.09	0.04	0.13	0.07	0.09	0.16	0.08	0.23	0.31
V_{WLUSE}	0.19	0.04	0.23	0.16	0.07	0.23	0.22	0.18	0.40
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.48	0.22	0.71	0.59	0.27	0.87	0.68	0.32	1.00
V_{LWD}	0.08	0.46	0.54	0.09	0.68	0.77	0.09	0.91	1.00
$V_{DETRITUS}$	0.08	0.46	0.54	0.09	0.68	0.77	0.09	0.91	1.00
V_{TDBH}	0.08	0.46	0.54	0.09	0.68	0.77	0.09	0.91	1.00
V_{WLUSE}	0.26	0.35	0.61	0.28	0.53	0.81	0.29	0.71	1.00

The remaining variables exhibit a smaller effect on FCI scores; however, each variable reacts appropriately across the range of potential model outcomes and, in combination, the variables display an appreciable impact on model results. For example, when V_{TDBH} remains at the minimum possible value (0.1) potential FCI scores equal 0.04 – 0.91 (Figure 4.5.3). When V_{TDBH} is maximized (subindex = 1.0) the observed range of FCI values occur between 0.13 and 1.0. Increasing the V_{TDBH} subindex incrementally from 0.1 to 1.0 while holding all other variables constant results in increases in FCI scores ranging 10 – 216%.

The minimum attainable FCI score of 0.04 requires V_{WLUSE}, V_{LWD}, and $V_{DETRITUS}$ all equaling 0.0 with the remaining variables (V_{TDBH} and V_{EMBED}) equaling 0.1. Achieving an FCI score of 1.0 requires all five variables (V_{EMBED}, V_{WLUSE}, V_{LWD}, $V_{DETRITUS}$, and V_{TDBH}) exhibiting values equaling 1.0.

For the Biogeochemical Cycling function with canopy cover <20 percent (Equation 4.5.2; Figure 4.5.4), sensitivity analysis results demonstrate that stream particle embeddedness (V_{EMBED}) maintains the strongest effect on model FCI outputs (Figure 4.5.5; Table 4.5.2). When embeddedness values independently remain at the lowest allowable subindex score (0.1), the

Figure 4.5.2. Results of sensitivity test of V_{EMBED} (canopy cover ≥20 percent). Each line in the figure represents a test scenario at which the target variable was increased from 0.0 to 1.0 in increments of 0.1 while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

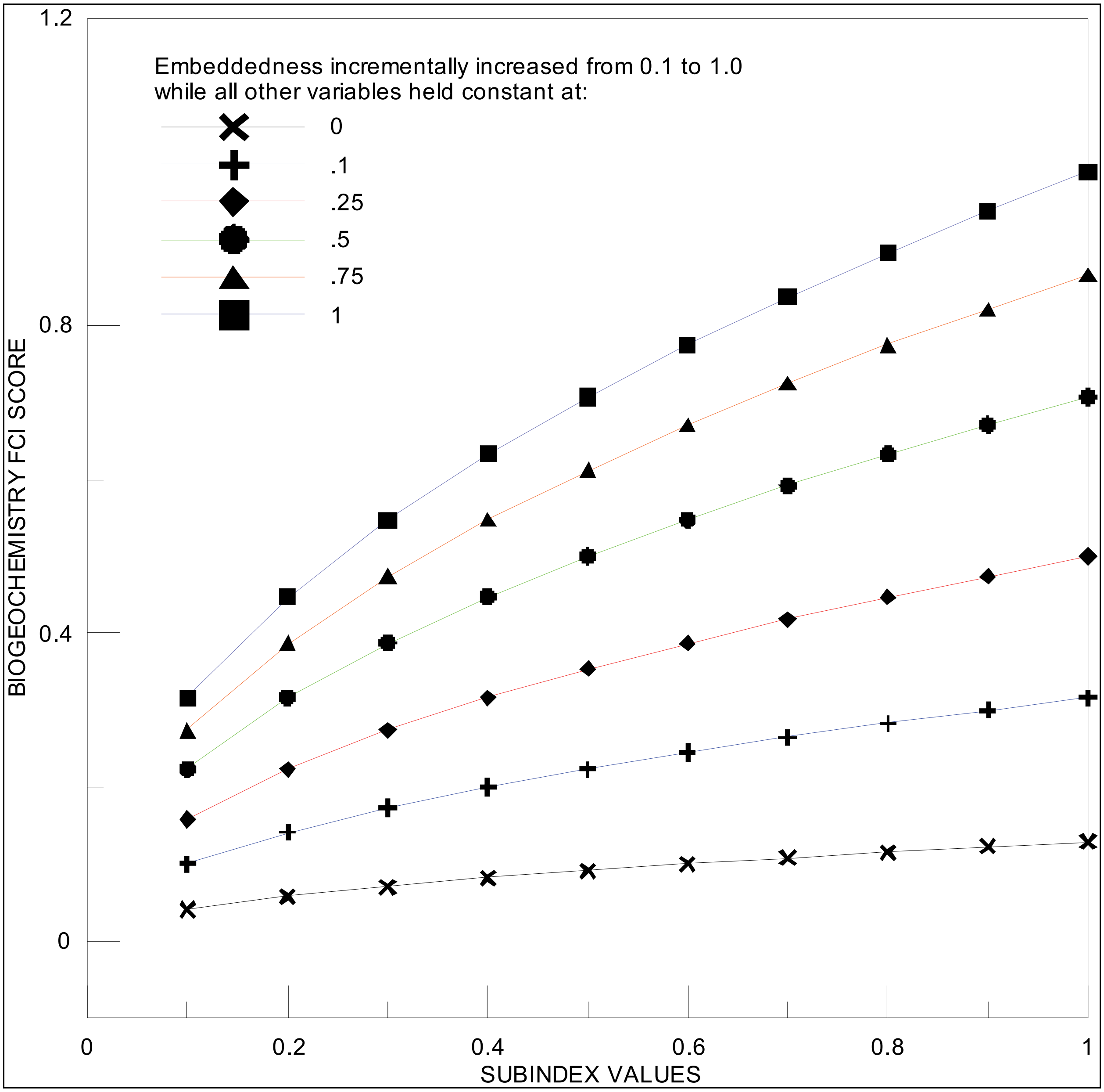
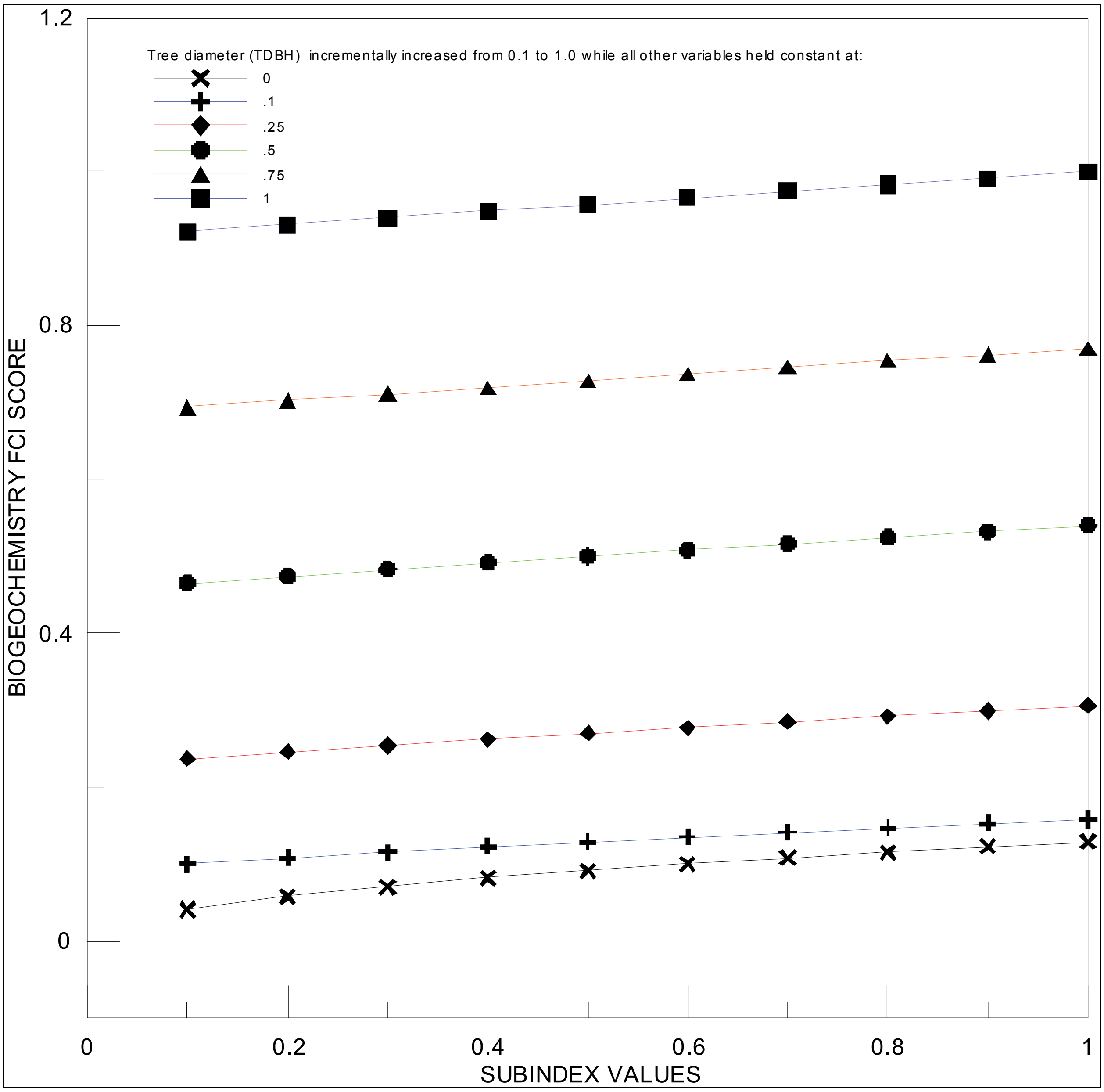
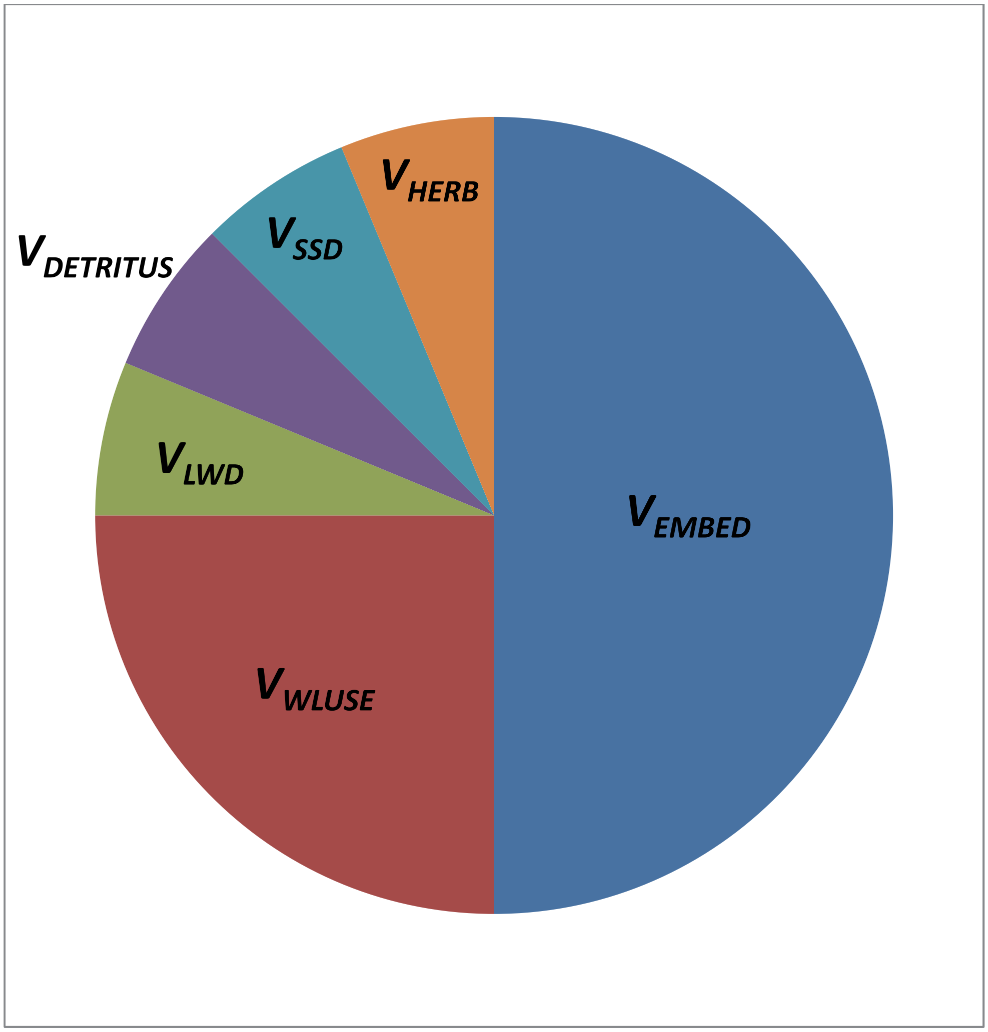


Figure 4.5.3. Results of sensitivity test of V_{TDBH} (canopy cover ≥20 percent). Each line in the figure represents a test scenario at which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



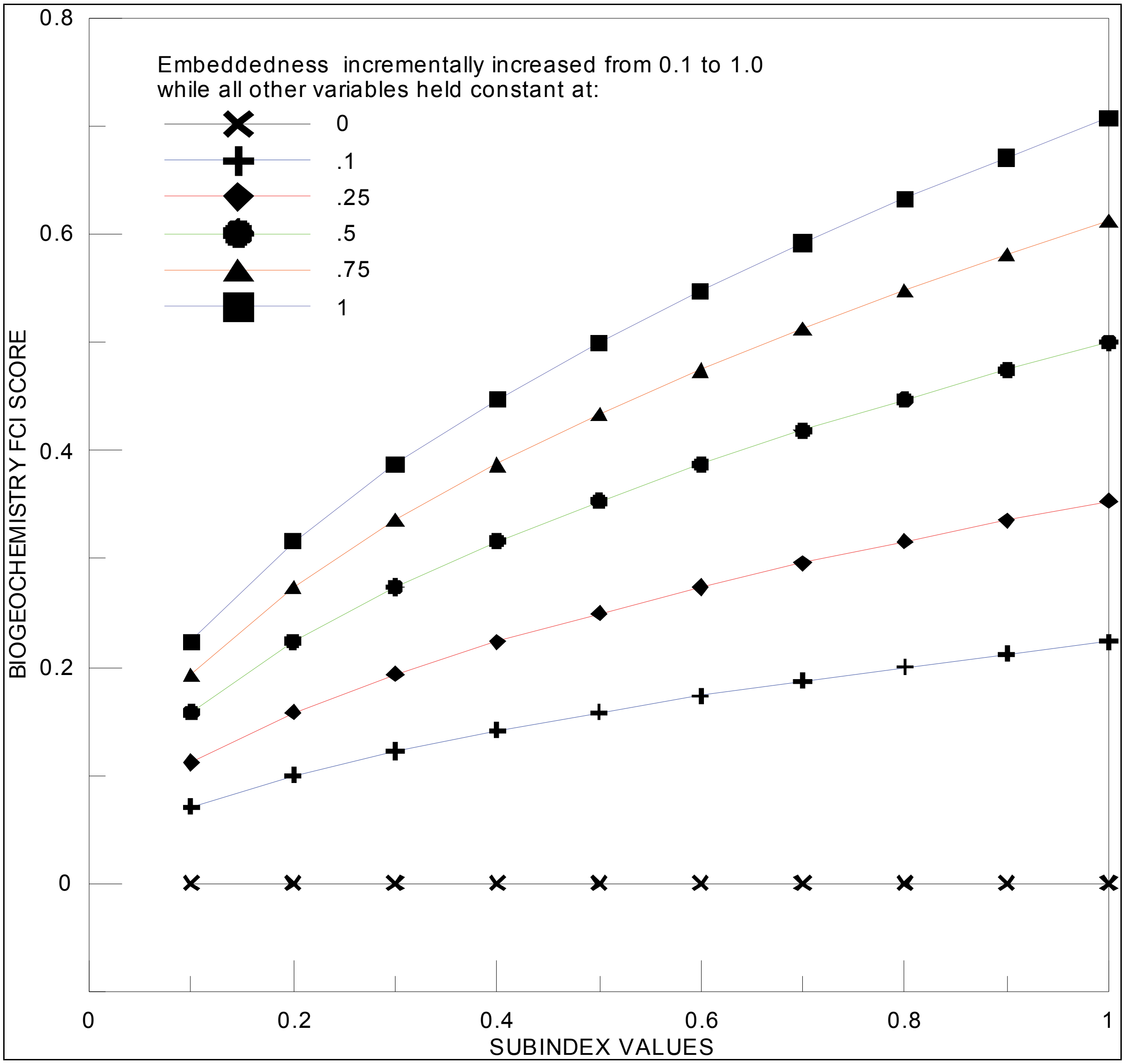
maximum attainable model FCI output equals 0.22. When embeddedness values reach a maximum value of 1.0, the range of potential values equals 0.0 – 0.71. FCI scores fail to reach 1.0 under any circumstance when canopy cover remains <20 percent. When embeddedness values reach a maximum value of 1.0, the range of potential values equals 0.13 – 0.71. Increasing the embeddedness score incrementally from 0.1 to 1.0 while holding all other variables constant results in an increase in FCI scores ranging from 0 – 216%.

Figure 4.5.4. Conceptual distribution of variable influence on Biogeochemical Cycling FCI scores in sites displaying <20 percent canopy cover.



The remaining variables exhibit a smaller effect on FCI scores; however, each variable reacts appropriately across the range of potential model outcomes and in combination maintain an appreciable effect on model outcomes. For example, when V_{LWD} remains at the minimum acceptable value (0.0) potential FCI scores equaling 0.0 – 0.66 are possible (Figure 4.5.6). When maximized (subindex = 1.0) the observed range of values equals 0.08 –.71. Increasing the V_{LWD} score incrementally from 0.1 to 1.0 while holding all other variables constant results in increases in FCI scores ranging 0 – 56%.

Figure 4.5.5. Results of sensitivity test of V_{EMBED} (canopy cover <20 percent). Each line in the figure represents a test scenario at which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



The equivalent treatment of V_{LWD}, $V_{DETRITUS}$, V_{HERB}, and V_{SSD} results from the necessity of capturing a variety of impacts and potential sources of nutrients, organic matter, and other materials to the stream system. For example, recently restored streams often display detrital ground cover representing short term source and storage of organic matter, but lack longer term storage represented by established sapling-shrub communities and large woody debris. FCI scores of zero require V_{LWD}, $V_{DETRITUS}$, V_{HERB}, and V_{SSD} all equaling 0.0. Achieving the maximum attainable FCI score of

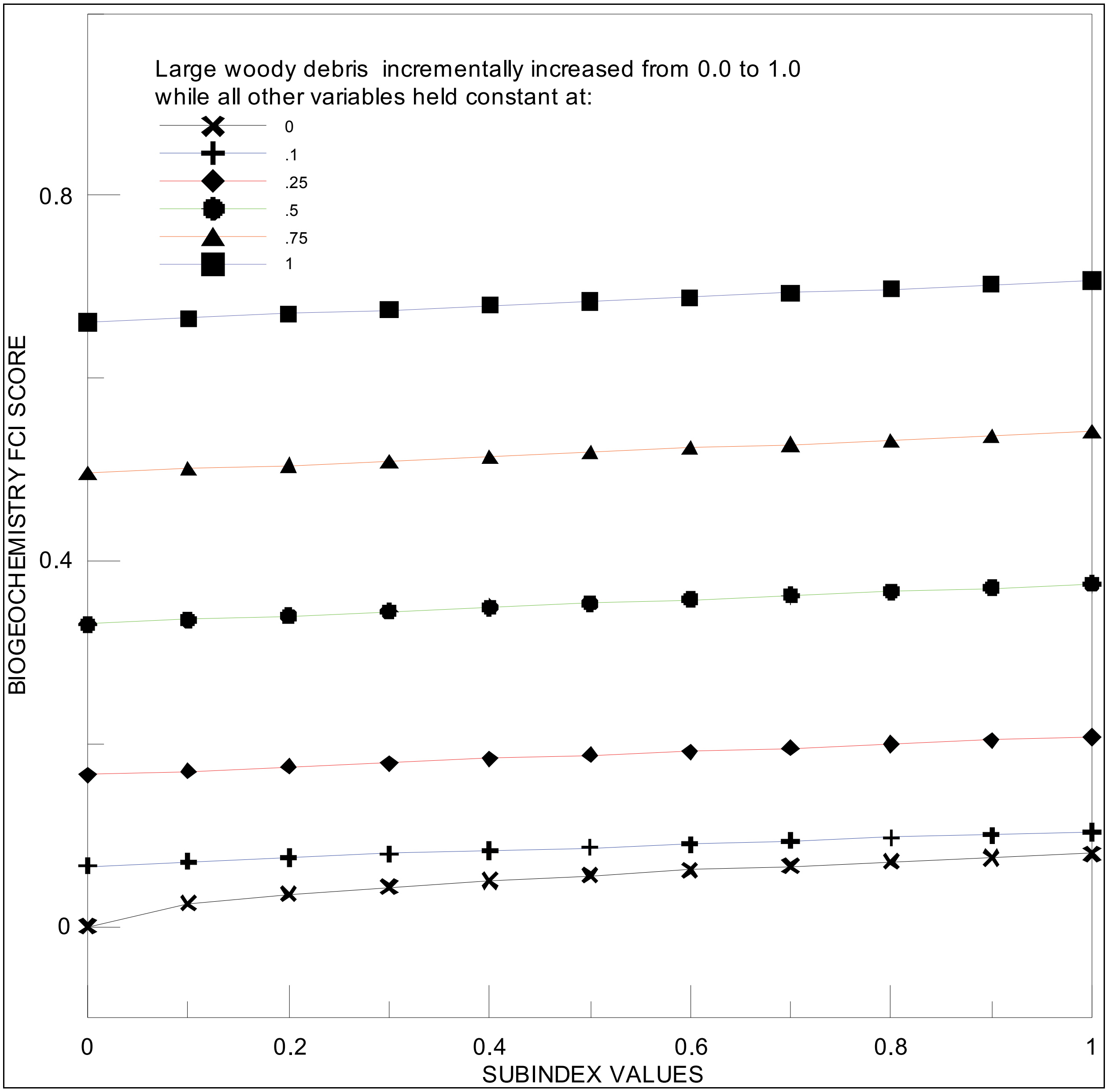
0.71 requires all six variables (V_{EMBED}, V_{LWD}, $V_{DETRITUS}$, V_{WLUSE}, V_{HERB} and V_{SSD}) equal 1.0.

$$FCI = \left\{ V_{EMBED} \times \left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{SSD} + V_{HERB}}{4} \right) + V_{WLUSE}}{4} \right] \right\}^{\frac{1}{2}} \tag{4.5.2}$$

Table 4.5.2. Range of FCI scores attainable based in sensitivity analysis (percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.1 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.00	0.00	0.00	0.15	0.07	0.22	0.24	0.11	0.35
V_{LWD}	0.08	0.00	0.08	0.04	0.07	0.10	0.04	0.17	0.21
$V_{DETRITUS}$	0.08	0.00	0.08	0.04	0.07	0.10	0.04	0.17	0.21
V_{DBH}	0.08	0.00	0.08	0.04	0.07	0.10	0.04	0.17	0.21
V_{WLUSE}	0.16	0.00	0.16	0.12	0.05	0.17	0.15	0.13	0.28
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.34	0.16	0.50	0.42	0.19	0.61	0.48	0.22	0.71
V_{LWD}	0.04	0.33	0.38	0.05	0.50	0.54	0.05	0.66	0.71
$V_{DETRITUS}$	0.04	0.33	0.38	0.05	0.50	0.54	0.05	0.66	0.71
V_{DBH}	0.04	0.33	0.38	0.05	0.50	0.54	0.05	0.66	0.71
V_{WLUSE}	0.18	0.25	0.43	0.20	0.38	0.57	0.21	0.50	0.71

Figure 4.5.6. Results of sensitivity test of V_{LWD} (canopy cover <20 percent). Each line in the figure represents a test scenario at which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



5 Validation of Hydrology function

5.1 Introduction

5.1.1 Functional definition and independent measures selected

The following chapter presents validation study results relating to the Hydrology function. The purpose of this section focuses on testing the ability of the Hydrology HGM rapid assessment model to differentiate between site conditions occurring along a gradient of alteration when compared against independent measures of ecological function. The study is not intended to provide a comprehensive evaluation of all potential hydrology functions, interactions, and relationships as they apply to general investigations evaluating stream or landscape ecological research.

The *Operational Draft Regional Guidebook for the Functional Assessment of High-gradient Ephemeral and Intermittent Headwater Streams in Western West Virginia and Eastern Kentucky* (Noble et al. 2010) defines the Hydrology function as:"…the ability of the high-gradient headwater stream to dissipate energy associated with discharge velocity and transport water downstream."

Few studies have focused on the hydrologic functioning of high-gradient ephemeral and intermittent headwater streams in the Appalachian Region. One study (Adams and Spotila 2005) found that high-gradient headwater streams fail to follow the typical hydrologic and sediment transport patterns of higher order streams within the Appalachian region.

Smith et al. (2013) describe procedures for using independent measures of function to validate HGM rapid assessment results. The Project Delivery Team (PDT) identified several independent measures that were used to quantify stream hydrology and validate stream characteristics along a gradient of alteration. Common approaches include direct measures of stream discharge, relationships between watershed size and discharge, sediment transport, relationships between watershed size and sediment transport, and channel geometry (Rosgen 1996; Leopold 1994; Leopold et al. 1992; Chang 2006; Golden and Springer 2006; Adams and Spotila 2005; Whiting et al. 1999). Available literature demonstrated that the direct measures listed above exhibit changes following alteration of the stream channel or the surrounding watershed. For example, multiple studies report that stream discharge and sediment transport increased following forest clearing within a headwater catchment (Gurtz et al. 1980, Patric 1973).

A basic concept of stream function is the interrelationship between the discharge of water, the movement of sediment, and channel geometry (Leopold 1994; Leopold et al. 1992; Cretaz and Barten 2007). The independent measures selected to test the validity of the Hydrology function address three aspects of stream hydrologic functions as defined above: 1) surface hydrology, 2) sediment transport (bedload and suspended), and 3) characteristic stream channel geometry. Guidance from the PDT and review of available literature helped determine the selection of stream measurements capable of validating the Hydrology function (Table 5.1.1).

Table 5.1.1. Independent measures of stream Hydrology function.

Functional proxy	Independent measures
Surface hydrology and transport of water downstream	Stream flow frequency
	Stream discharge and watershed area
Sediment transport	Bedload transport and watershed area
	Bedload transport and discharge
	Suspended sediment yield
Characteristic stream channel geomorphology	Channel width:depth ratios

The study examines the relationships between Hydrology FCI scores, surface hydrology, sediment transport, and stream channel geometry. Figure 5.1.1 contains the hydrograph of stream discharge, daily rainfall, and bedload measurements collected at one sample site to illustrate the type of data that was collected and used to validate the Hydrology function. High-gradient headwater streams within the study area display high levels of geomorphic, landscape, and ecological variability including a wide range of watershed sizes, riparian hillslopes, channel slopes, total precipitation, vegetative compositions, and other characteristics. Due to the number and complexity of variables affecting study site hydrology, it was difficult to make meaningful direct comparisons of one direct measure of stream Hydrology function (e.g., total discharge) with the FCI scores which incorporate several interrelated characteristics. FCI scores provide a relative comparison between reference standard sites, which display minimal alteration, and sites exhibiting the gradient of alterations described in the HGM guidebook.

The validation approach utilized here establishes relationships between independent measures of hydrologic function and FCI scores observed at the least-altered sites in the validation study. Forested sites with a natural channel and mature, stable riparian/buffer zone characterized the least-altered study sites (sites 2, 3, 5, 8, and 9; Table 5.1.2). The ability of the HGM Hydrologic model to differentiate between sites exhibiting a gradient of alteration defines validation success. By examining the departure of impaired sites from the relationships established at the least-altered sites, validation of the HGM model can be tested (Figure 5.1.2). For example, if all least-altered stream channels with forested watersheds exhibit discharge, sediment, and stream channel geometry relationships within a given range, a subset of sites with altered riparian watershed and channel characteristics would be expected to deviate from that range. As a result, a number of relationships can be examined to test the validity of the HGM Hydrology function by comparing FCI scores from altered and least-altered sites to determine whether the HGM outcomes differentiate among the study sites.

Figure 5.1.1. Example of discharge (blue line), rainfall (black bar), and bedload (points) for one site.

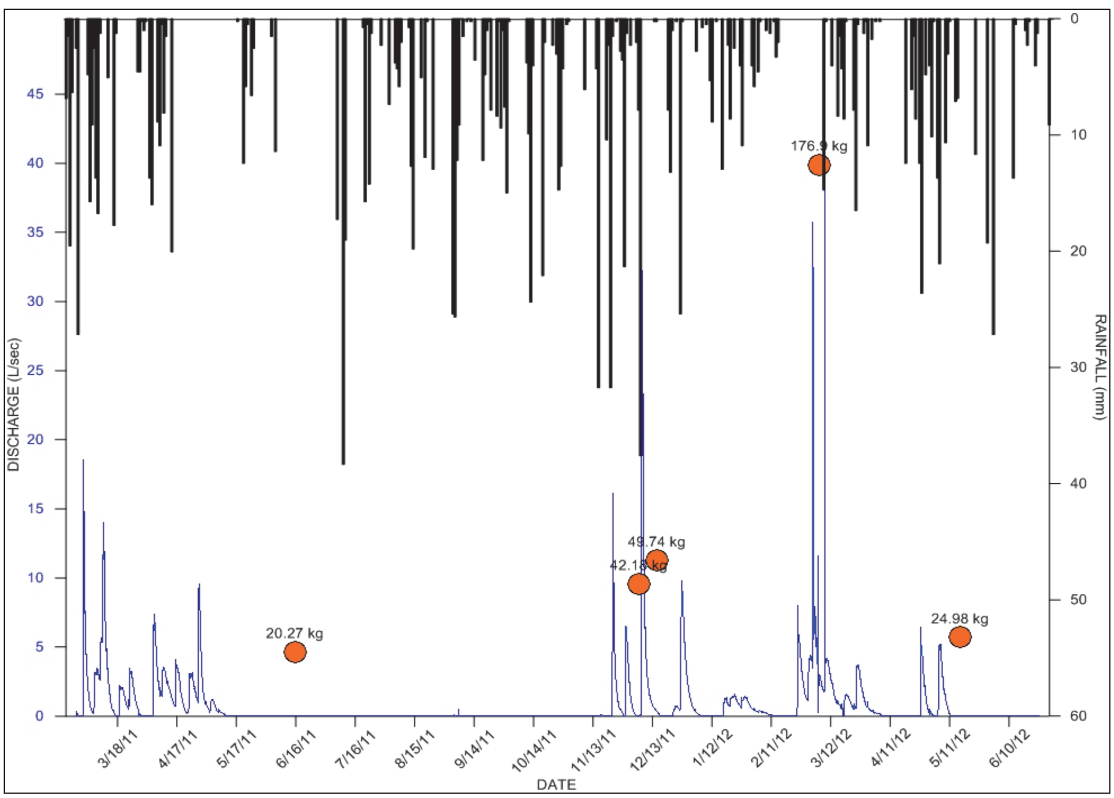


Table 5.1.2. Hydrology metrics which fall outside of the range observed at the least-altered sites*.

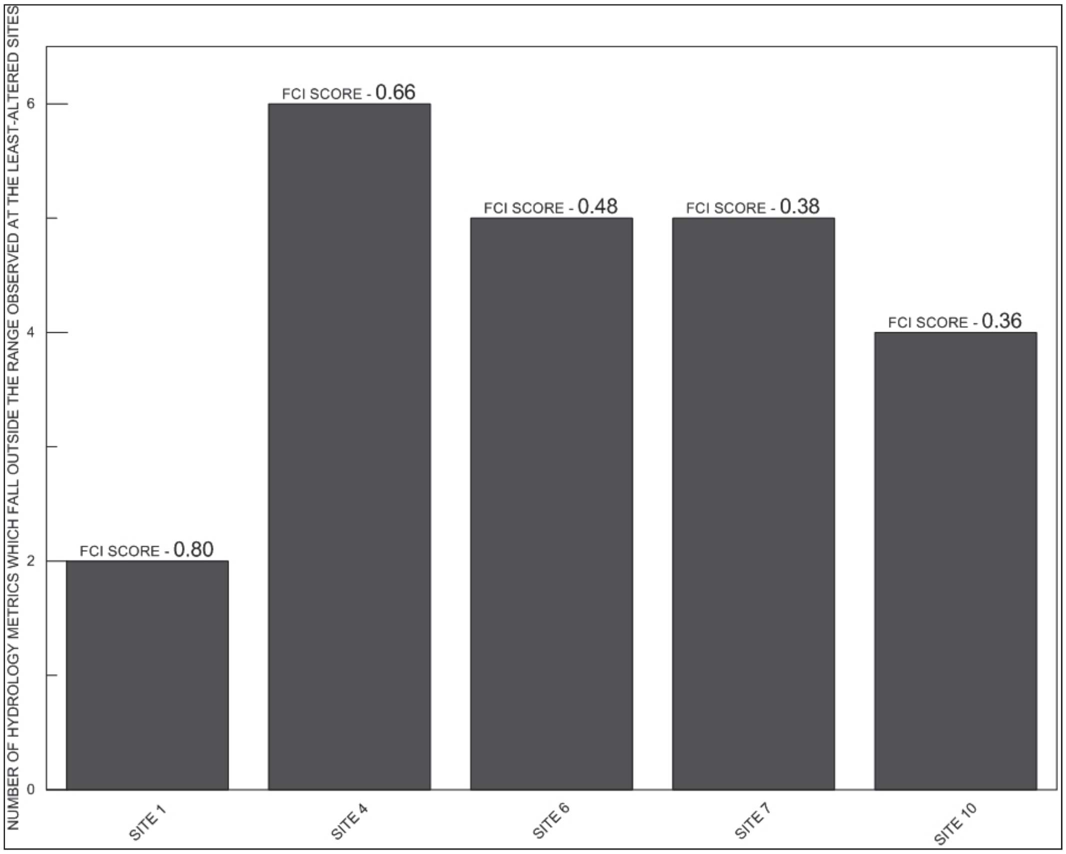
	Surface Hydrology and Transport of Water Downstream		Sediment Transport			Characteristic Stream Channel Morphology		
Site	Stream flow frequency (Fig 5.2.6)	Stream discharge and watershed area (Fig 5.2.8)	Bedload transport and watershed area (Fig 5.3.5)	Bedload transport and discharge (Fig 5.3.6)	Suspended sediment yield (Fig 5.3.7)	Channel width:depth ratio (Fig 5.4.1)	Site condition	Hydrology FCI
1	X					X	Silviculture	0.80
2							Least-altered forested watershed*	0.95
3							Least-altered forested watershed*	0.95
4	X	X	X	X	X	X	Valley fill; mining	0.66
5							Least-altered forested watershed*	0.96
6	X	X		X	X	X	Valley fill; mining	0.48
7	X		X	X	X	X	Agriculture pastureland	0.38
8							Silviculture*	0.76
9							Least-altered forested watershed*	0.87
10	X			X	X	X	Valley fill; mining	0.36

5.1.2 Summary of findings

Each of the three hydrologic characteristics examined as part of the hydrology validation process (surface hydrology, sediment transport, stream channel geometry) support the existing HGM Hydrology model (Equation 5.5.1). Table 5.1.2 and Figure 5.1.2 depict the predominant alterations within the data set, providing visual representation of the relationship between FCI scores and study sites.

The relationship between the HGM functional assessment model and independent measures is clear. Study sites receiving high FCI scores display discharge, sediment transport, and channel geometry consistent with highly functioning headwater streams. Study sites exhibiting alterations deviate

Figure 5.1.2. Relationship of altered sites to the number of alterations and Hydrology FCI scores.



from the observed patterns, for one or more independent measures of function. The HGM Hydrology model effectively differentiates between highly and poorly functioning headwater stream ecosystems, and validation data supports the continued use of this model. Sensitivity analysis results demonstrate that all variables incorporated into the Hydrology model affect rapid assessment outcomes.

5.2 Surface hydrology

5.2.1 Rational for selecting the independent measures

Hydrology is the primary factor controlling stream processes (Doyle et al. 2005). This section describes how measurements of stream discharge were used to test the validity of the Hydrology function. Stream discharge is a volumetric measure of water moving past a point over time (e.g., L/sec).

Stream discharge was used to develop discharge frequency curves. Discharge frequency curves illustrate the percentage of time that stream discharge exceeded some specified value of interest. Discharge frequency curves are a common method of comparing stream hydrology. Several studies have shown that alteration to the stream channel, riparian/buffer zone, or surrounding watershed can alter the streams discharge (Cretaz and Barten 2007). The difference in discharge frequency curves between least-altered and altered study sites and the comparison of stream discharge to watershed area represent independent measures used to validate the transport of water in the stream channel.

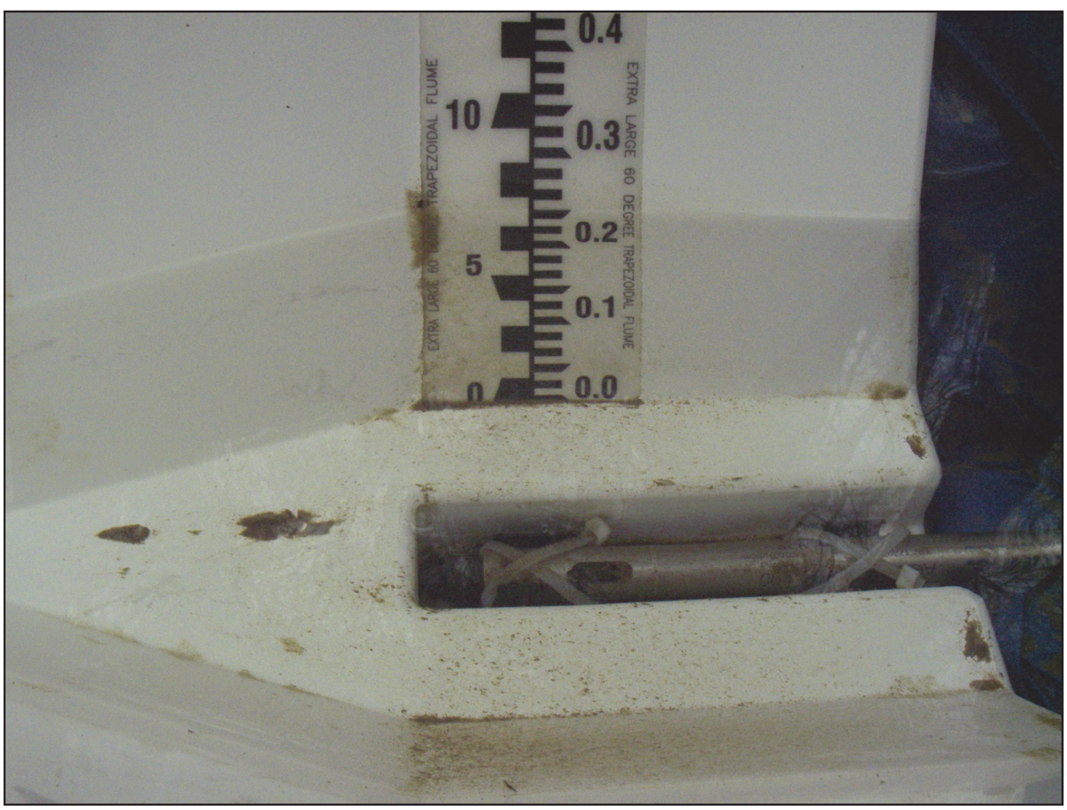
High-gradient ephemeral and intermittent streams in the Appalachian region that are forested, with a natural channel and mature, stable riparian/buffer zone represent the least-altered study sites (sites 2, 3, 5, 8, and 9; Table 5.1.2). These study sites would be expected to have similar discharge patterns. Discharge for altered streams would be expected to have discharge patterns that deviate related to the type and extent of alterations to the channel and surrounding watershed (sites 1, 4, 6, 7, and 10; Figure 5.1.2).

5.2.2 Methods

Trapezodidal flumes were used to collect stream discharge data. Flumes are a constructed, calibrated device placed in a stream channel for the purpose of measuring stream discharge. The size and shape of the flume are known and calibrated, and discharge can be easily determined by measuring the depth of the water in the flume. Discharge measurements utilized a Plasti-Fab, extra-large, 60-degree, trapezoidal flume (Tualatin, OR); (Figure 5.2.1). The flume's construction included a recessed notch for the purpose of supporting an Aqua Troll 200 (Ft. Collins, CO) pressure transducer for measuring water depth. Pressure transducers were vented to compensate for changes in barometric pressure. Direct observations to calibrate water levels occurred during each site visit using the permanent gauge installed on the flume by the manufacturer (Figure 5.2.2). The difference in the observed water level and pressure transducer data for 20 or more gauge readings was used to determine a calibration factor between each data logger and the water level in the flume. The data logger recorded water levels every 15 min. Discharge calculations utilized the water depth measurements collected by the pressure transducers and depth-discharge equations supplied by flume

Figure 5.2.1. Flume for measuring stream discharge.

Figure 5.2.2. Gauge and pressure transducer in flume for recording water level.



manufacturers. The flumes design accommodated for discharges of 0.0054 to 42.92 L/sec (Walkowiak 2008). As a result, stream discharge measurements below 0.0054 L/sec (approx. 1.0 teaspoon per second) were recorded as zero discharge (Croker et al. 2003; Gordon et al. 2004; Smakhtin 2001). Periods with zero discharge were removed from the data set before analysis since without measurable discharge, water and sediment are not being transported and channel forming processes cannot take place. The remaining periods when discharge was ≥0.0054 L/sec are defined as "discharge-day" (Table 5.2.1). Maximum water levels (42.92 L/sec) were recorded in six flumes during the study period.

Table 5.2.1 Number and percent of days with measurable discharge (discharge-day).

Site	Number of discharge-day	% discharge-day
1	270.2	53.3
2	197.6	39.8
3	193.2	38.0
4	165.4	32.6
5	478.2	95.3
6	309.3	61.7
7	204.5	48.9
8	380.2	98.3
9	380.6	97.7
10	203.3	56.5

Rainfall was measured for each site utilizing a tipping bucket HOBO Data Logging Rain Gauge (RG3) (Pocasset, MA: Figure 5.2.3). Rain gauges were located in a nearby clearing with an adequate canopy opening to reduce canopy interception of rainfall. Rainfall was recorded each time the rainfall reached 0.254 mm (0.01 inches). Evaluations of rainfall normality determined whether the validation study occurred during a period of drought, excessive rainfall, or average climatic conditions. Rainfall (examined monthly) is considered normal if the observed rainfall amount falls within the 30th and 70th percentiles established by a 30-year rainfall average generated at the nearest National Oceanic and Atmospheric Administration (NOAA) weather station (modified from Sprecher and Warne 2000).

Figure 5.2.3. Tipping bucket rain gauge used to record rainfall.

5.2.3 Results

All of the streams exhibited one or more periods with no surface water discharge during the monitoring period. This provides evidence that all of the study sites were ephemeral or intermittent and matched the stream subclass definition described within the HGM guidebook. Compared to the local National Oceanic and Atmospheric Administration (NOAA) weather station, rainfall at study sites exhibited dry, normal, and wetter-than-normal months during the study period (Figure 5.2.4). Normal rainfall occurred most frequently and accounted for 42 percent of the total months examined for all 10 sample sites. Wetter-than-normal rainfall months (35 percent) and drier-than-normal (23 percent) months occurred less often (Figure 5.2.5). Appendix B contains rainfall data for each study site.

Validation of the HGM Hydrology function utilized a number of comparisons. Discharge frequency curves illustrate the distribution and occurrence of various discharge events. Discharge curves were developed by sorting all discharge data from each for the study sites from most frequent to least frequent. Discharge measurements were adjusted for differences in watershed size (L/sec/ha). The least-altered study sites exhibited forested watersheds characterized by stable channels and received high FCI scores. These sites displayed a similar pattern of discharge frequency curves that group together across the range of discharges observed (shaded in gray; Figure 5.2.6). Study sites exhibiting alterations within the channel or the surrounding watershed deviated from the pattern observed in the least-altered sites and received lower FCI scores.

Figure 5.2.4. An example of a comparison of monthy rainfall data for one validation site. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

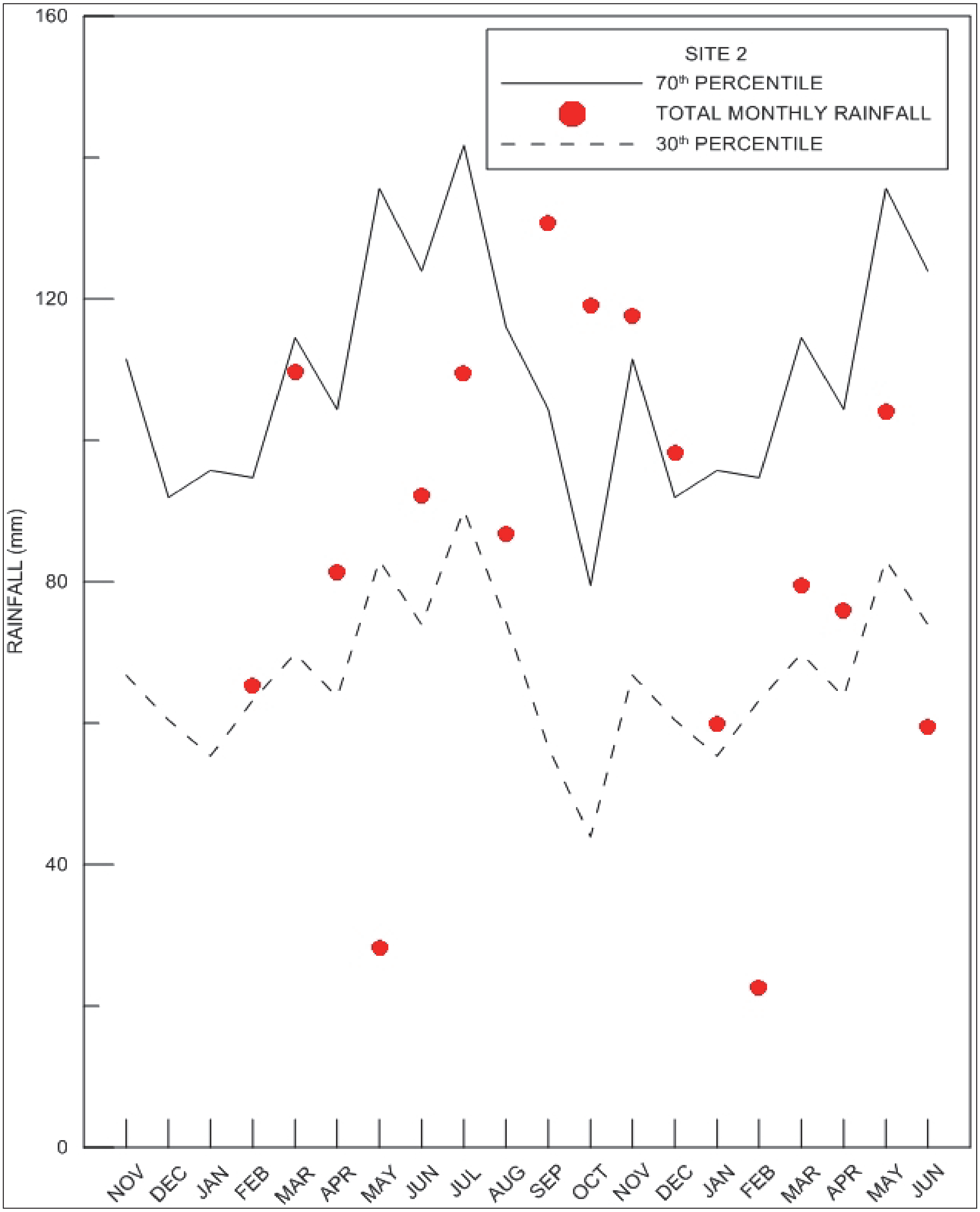


Figure 5.2.5. Months of drier-than-normal, normal, and wetter–than-normal rainfall observed at each study site.

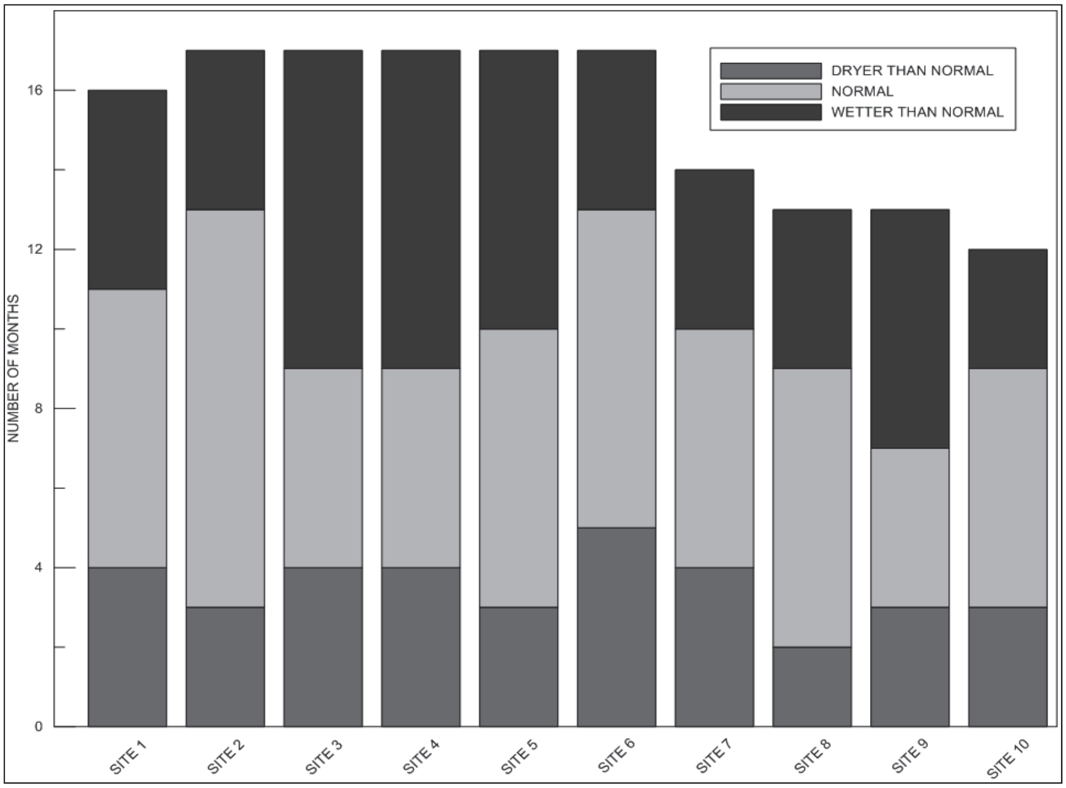
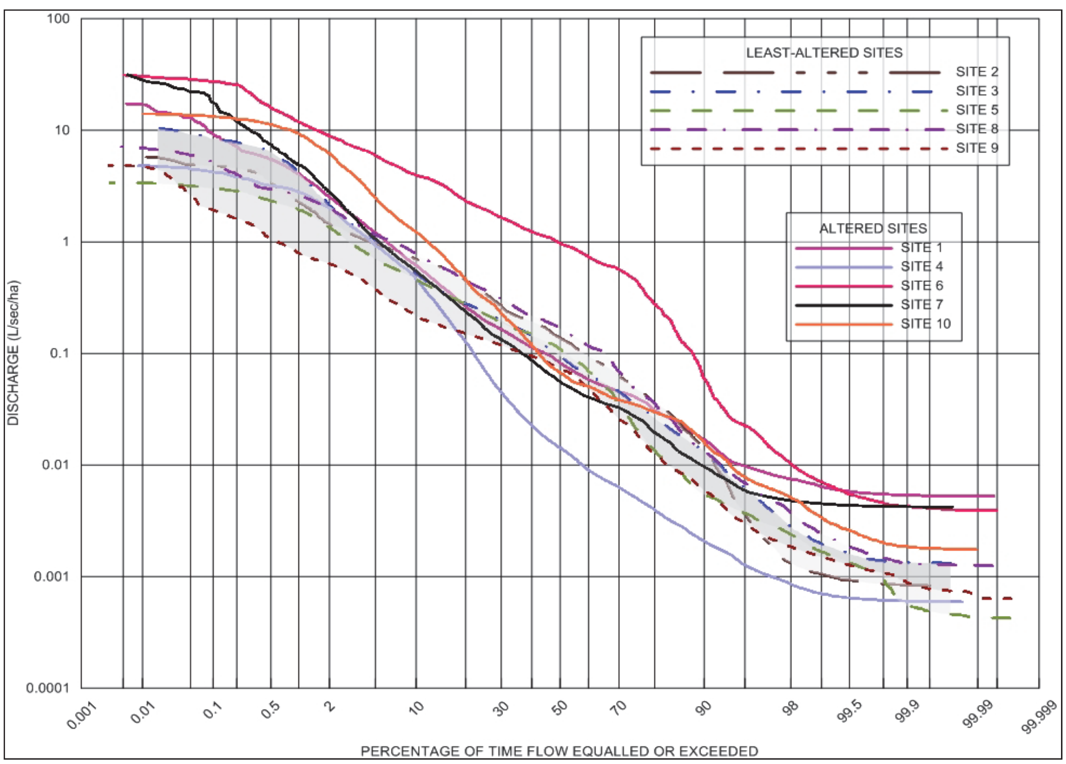


Figure 5.2.6. Stream discharge frequency curve. Note the shaded area indicating the distinct pattern observed within the least-altered sites and the deviation of the altered sites.



For example altered sites 1, 6, 7, and 10 exhibited higher discharge compared to least-altered study sites at a frequency of 98 percent. This is consistent with results reported by Patric and Reinhart (1971) and Chang (2006). The discharge frequency curve for Site 6 shows a higher rate of discharge across the entire range when compared to the group of least-altered sites. Site 6 has a watershed impacted by mining and valley filling (see Section 2). In contrast, Site 4 exhibits a lower discharge rate across a large portion of this discharge curve when compared to the least-altered sites (Figure 5.2.6). All altered sites showed a deviation in discharge from least-altered sites along a portion of the discharge frequency curve. Differences in the discharge frequency curves for altered sites in relation to least-altered sites demonstrate that the Hydrology model differentiates between a range and variety of alterations (Table 5.1.2 and Figure 5.1.2).

Total stream flow volume (L) ranged from 7438 L to 129819 L during the study period. Due to the variation in study site watershed area and flow regime, cumulative volume provides limited utility in evaluating the HGM hydrology function. In order to account for periods when no discharge occurred, discharge calculations were converted to a per-discharge-day basis by dividing total discharge by number of days when discharge equaled or exceeded the minimum measurable level of 0.0054 L/sec (Table 5.2.1). Discharge ranged from 27.5 to 271.5 L/sec/day (Figure 5.2.7).

Comparing watershed area and discharge is one of the most widely used parameters in hydrologic analysis (Gingras, Adamowski, and Pilon 1994). Many studies have shown a relationship between watershed area and discharge (Leopold 1994; Leopold 1992; Chang 2006; Adams 2005; Whiting et al. 1999).

Comparing total discharge and watershed area at the least-altered study sites produces the expected relationship (Figure 5.2.8). These findings agree with data provided by Leopold et al. (1992), Leopold (1994) and Chang (2006). A number of the altered sites deviate from the relationship observed within least-altered sites. Several studies link the amount of alteration within a stream channel and the surrounding watershed to changes in discharge patterns. Specifically, increasing alteration and decreasing forest cover often increase discharge (Chang 2006; Douglass 1983; Adams 2005; Patric 1973; Patric and Reinhart 1971). The deviation of altered sites observed in Figure 5.2.8 demonstrates this effect, as all

Figure 5.2.7. Total discharge for each validation study site. Discharge calculations only include days when discharge equaled or exceeded the minimum measureable level of 0.0054 L/sec.

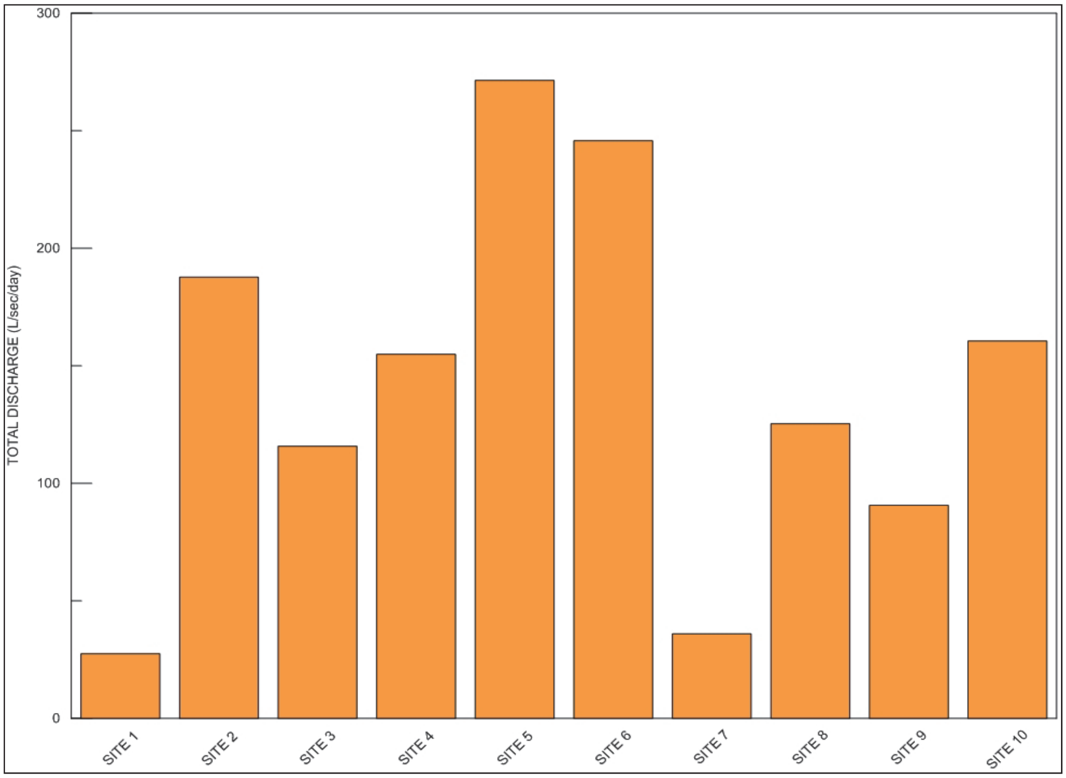
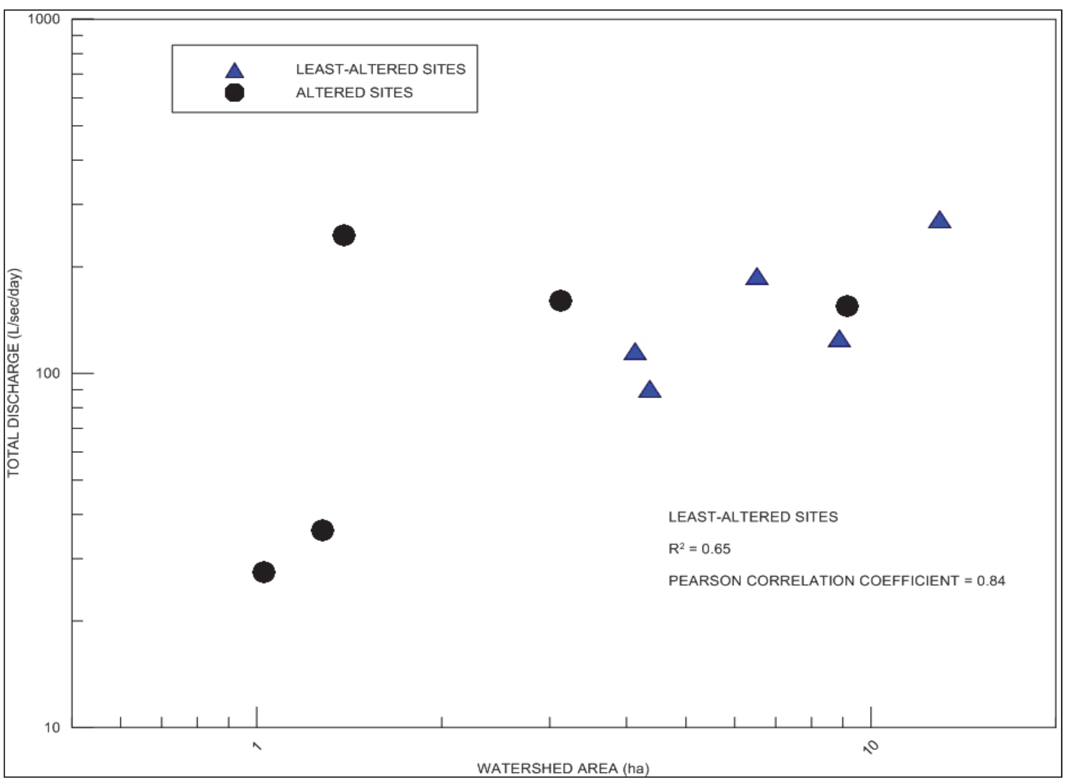


Figure 5.2.8. Total discharge compared to watershed area. Discharge calculations only include days when discharge equaled or exceeded the minimum measureable level of 0.0054 L/sec.



deviating sites include areas where silviculture, mining, and/or agricultural activities decrease forest cover and alter hydrologic regimes. These results support the configuration of the HGM Hydrology model. Findings also support the weighting of watershed land use as the most critical variable in the model (Equation 5.5.1).

Although some altered study sites show similar relationships as observed in least-altered sites with respect to discharge and watershed area, they deviate when compared to other validation metrics (Table 5.1.2; Figure 5.1.2). As indicated, the variability and complexity of hydrologic regimes makes establishing a single independent measure for validating hydrology function impracticable.

5.2.4 Summary

Stream discharge frequency curves show the relationships between stream discharge and frequency of occurrence. The least-altered study sites had forested watersheds characterized by stable channels displaying discharge frequency curves that grouped together forming a distinct pattern. A subset of study sites exhibiting alterations within the channel or the surrounding watershed deviated from the pattern observed in the least-altered sites and received FCI scores that reflected the type and amount of alteration (Table 5.1.2 and Figure 5.1.2). Stream discharge and watershed area results showed similar results, with least-altered locations forming a distinct relationship, while a subset of altered locations deviated from the observed pattern. The HGM Hydrology model displayed the ability to differentiate between altered and least-altered locations.

5.3 Sediment Transport

5.3.1 Rational for selecting the independent measures

All streams transport sediment. Forested sites with a natural channel and mature, riparian/buffer zone as observed at the least-altered study sites (Table 5.1.2) are considered to be stable and transport sediment that is in balance with hydrology. The amount of sediment is controlled by many factors, but is directly related to the hydrology of the system. Several studies have shown that alteration to the stream channel, riparian/buffer zone, or surrounding watershed can alter the amount of sediment transported by the streams discharge (Cretaz and Barten 2007). By comparing the amount of sediment transported by altered study sites to least-altered study sites we can test the Hydrology function (Figure 5.1.2).

Stream sediment is commonly divided into bedload and suspended sediment. For the purpose of this study bedload is all mineral particles that settled and were trapped in the settling pool constructed directly in front of each flume. Suspended sediment was collected in the water column with water samples examining water quality.

5.3.2 Methods

Bedload and suspended sediment were the two types of sediment examined to validate the Hydrology function.

Bedload measurements utilized a settling pool constructed directly in front of each flume. All sediments deposited within the settling pool were routinely cleared, weighed and sieved for particle size. Sediment measurements included weighing and sieving using 9.5 mm (0.38 in.), 2 mm (0.08 in.) and 0.6 mm (0.02 in.) sieves (Figure 5.3.1). Sediment was air dried in the field before being sieved and weighed. In addition, the three largest particles observed within each settling pool were weighed and measured (Figure 5.3.2). If the total weight of sediment exceeded 45 kg (100 lbs), a 45 kg subsample was weighed and sieved. In order to account for periods when no discharge occurred and sediment could not be transported, bedload calculations were converted to a per-discharge-day basis by dividing total bedload by number of days when discharge equaled or exceeded the minimum measurable level of 0.0054 L/sec (Table 5.2.1).

Site 3 experienced three storm events resulting in uncharacteristic amounts of sedimentation based on the surrounding forested watershed. These events resulted in bedload approximately six times greater than any other observed event at the sample site. Visual evidence suggests that debris flows produced the observed sedimentation (Figure 5.3.3; Pierson 2005). As a result, the three sediment events were removed from the dataset as outliers prior to analysis. Exclusion of the three data points is supported by the fact that no other high discharge events (>10 L/sec/day) moved large quantities of sediment at Site 3.

Figure 5.3.1. Cleaning and sieving sediment from settling pool.

Figure 5.3.2. Measurement of one of the three largest particles found in the sediment pool at each observation.

Figure 5.3.3. Evidence of debris flow at Site 3.

Suspended solid measurements and the calculation of sediment loading rates at each study site followed the approaches outlined in Section 4. Statistical procedures included normality testing applying the Shapiro-Wilk test ($p>0.05$ indicates a normal distribution) followed by Pearson Product Moment Correlation analysis (SPSS IBM, Inc. Version 20).

5.3.3 Results

Total bedload transport results ranged from 39 to 2,193 kg (Figure 5.3.4). Due to the variation in watershed area and flow regime among study sites, cumulative sedimentation data provides limited utility in evaluating the HGM model. The amount of bedload ranged from 0.15 to 8.7 kg per discharge-day.

A common hydrological comparison examines the relationship between transported sediment and watershed size. Comparing bedload and watershed size at the least-altered study sites produces the expected relationship (Figure 5.3.5). Two of the altered sites displayed increased bedload, deviating from the relationship observed within the least-altered sites. This is consistent with the findings of Chang (2006), Whiting et al. (1999), and Leopold et al. (1992) who link watershed and stream alteration

Figure 5.3.4. Total bedload for each sample location. Three outliers were later removed from Site 3 prior to analysis.

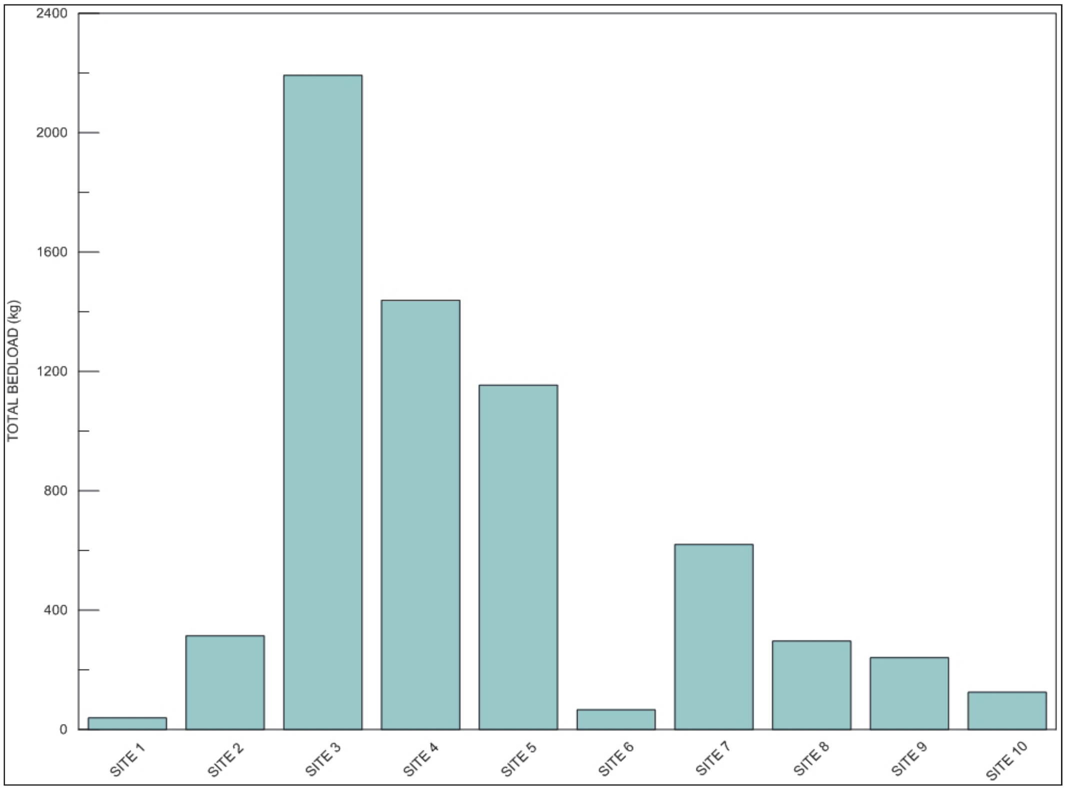
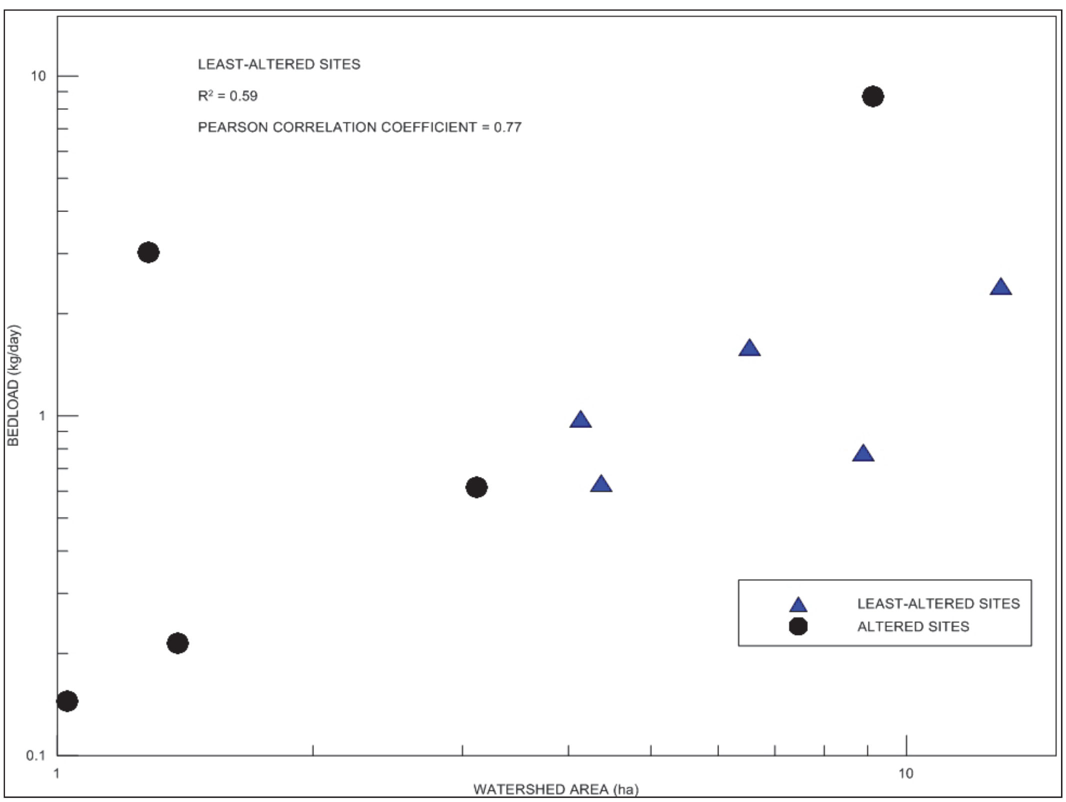


Figure 5.3.5. Bedload transport compared to watershed area.



with increased bedload transport. The departure of a subset of the altered sites from the least-altered sites provides evidence that the current HGM Hydrology model and FCI scores distinguish between site conditions across a gradient of stream channel and watershed disturbance.

The comparison of discharge and bedload represents another common metric addressing stream properties and hydrology. Available literature links increases in discharge with increasing bedload transport (Patric 1973; Chang 2006; Patric 1976; Whiting et al. 1999). Comparing discharge and bedload at the least-altered study sites produces the expected association (Figure 5.3.6). A number of the altered sites deviate from the relationship observed within least-altered sites. Several altered study sites exhibit bedload transport rates higher than the range observed at least-altered study sites. However, lower-than-expected bedload occurred in study sites where flow was artificially restricted, including a rock-lined channel where flow was constricted by culverts and where sediment basins had been constructed upstream. The relationship between discharge and bedload transport for least-altered sites shows significant correlations ($r = 0.98$, $n = 5$, $P = 0.0013$). Sites characterized by expected sediment-discharge relationships included study sites with unaltered stream channels and forested watersheds.

Section 4 describes methods used to collect data for suspended solids and discusses the use of suspended solids loading as an independent measure of the Biogeochemical Cycling function in streams. However, the transport of suspended sediments also represents an independent measure of hydrologic function. The findings of this study and others suggest that alteration increases sediment transport. As a result, suspended sediment is also expected to increase within altered study sites. Total suspended solids loading ranged from 22.2 – 375 kg/ha, with an average of 167 kg/ha. As seen in Figure 5.3.7, study sites receiving high Hydrology FCI scores exhibited low suspended sediment yields (<120 kg/ha). These sites were characterized by stable stream channels and forested watersheds. In contrast, sites displaying high suspended sediment yields received Hydrology FCI scores reflecting the alterations within the watershed (Table 5.1.2 and Figure 5.1.2). Comparisons of total suspended solid loading with HGM Hydrology FCI scores show significant negative correlations ($r = -0.81$, $n = 10$, $P = 0.0047$). These findings support the current HGM model.

Figure 5.3.6. Bedload transport compared with total discharge.

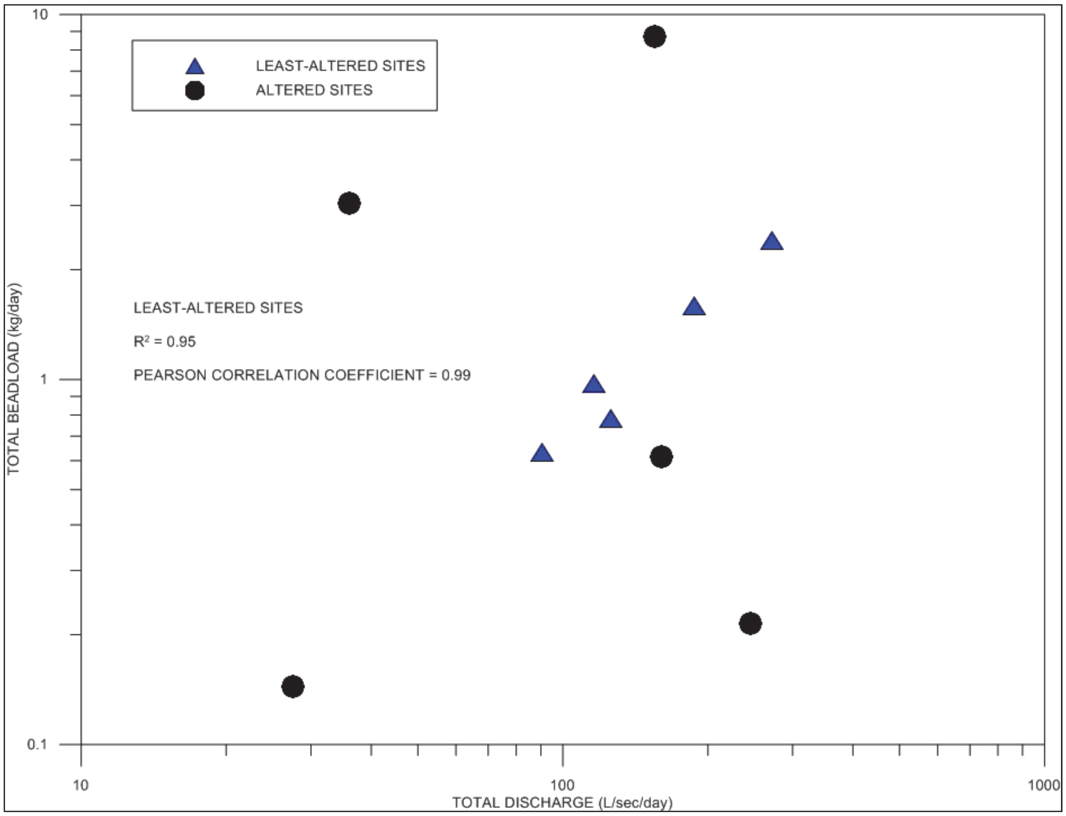
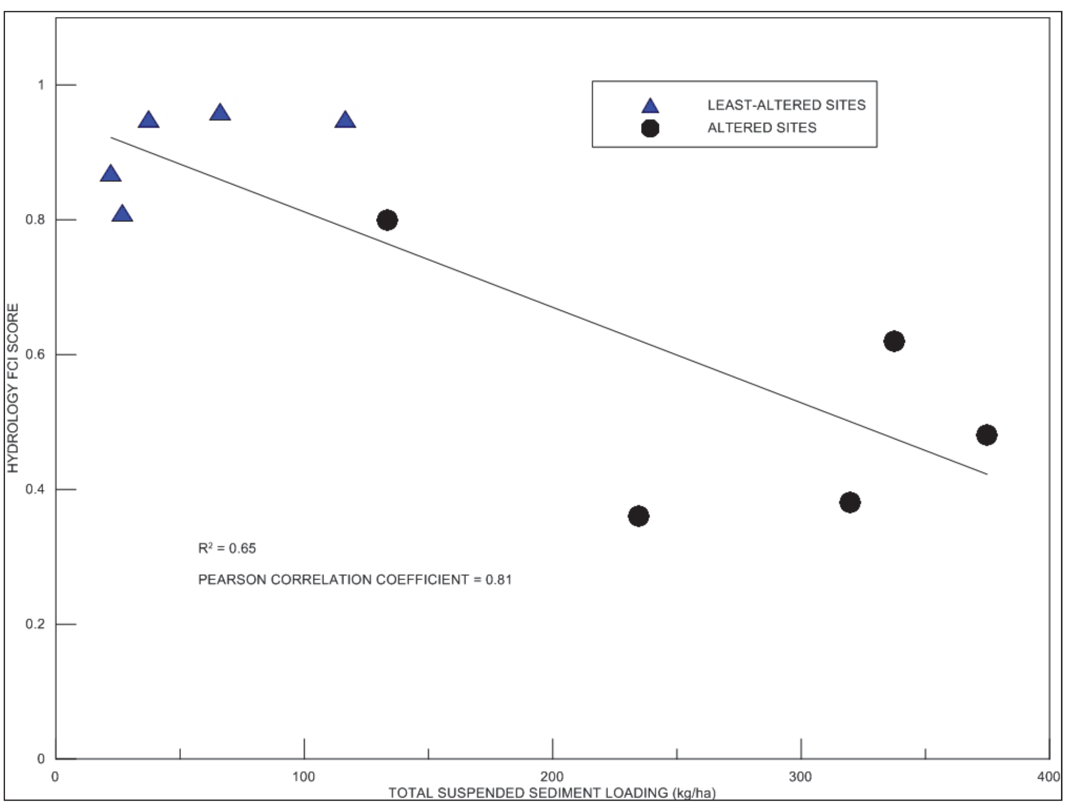


Figure 5.3.7. Comparison of hydrology FCI scores to suspended sediment loading.



5.3.4 Summary

Sediment transport evaluations displayed relationships with the least-altered study sites in terms of both discharge and watershed area. Further, significant correlations between FCI scores and suspended sediment loading support the ability of the HGM model to distinguish between sites exhibiting a variety of alterations.

5.4 Characteristic stream channel geometry

5.4.1 Rational for selecting the independent measures

The ratio of stream width to channel depth represents a common measure of stream stability and geometry (Whiting et al. 1999; Leopold 1994; Leopold 1992; Adams 2005; Rosgen 1996). Forested sites with a natural channel and mature, stable riparian/buffer zone as observed at the least-altered study sites (Table 5.1.2; Figure 5.1.2) are considered to be stable and can be expected to have channel geometry (width:depth ratio) in balance with the hydrology of the ecosystem. Alterations to the stream channel or surrounding watershed can alter the balance of hydrology and sediment transport, changing the width:depth ratio.

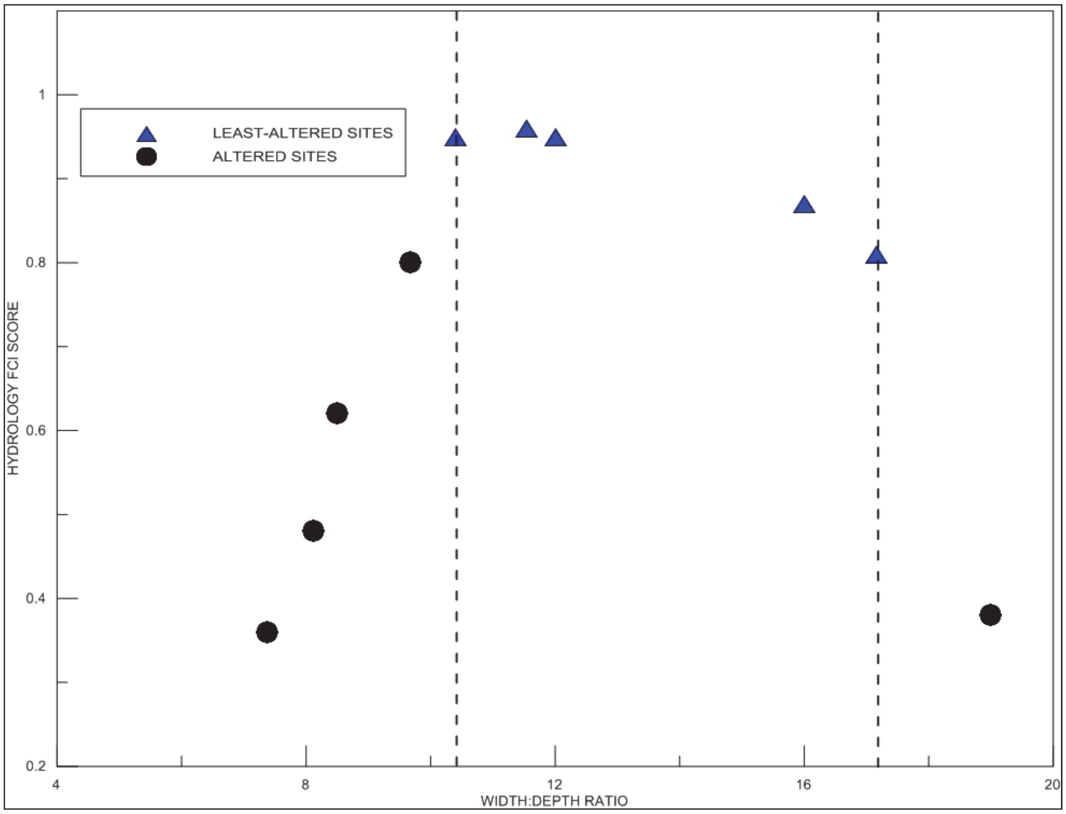
5.4.2 Methods

Stream cross sections were developed from ground-based LiDAR data taken during the summer of 2011. Using HECRAS (Brunner 1995), cross sections were placed approximately every 2.0 m along the entire stream reach. Representative cross sections were selected based on proximity to the flume and knowledge of the stream reach. The width for the width:depth ratio was based on the channel width at the 50% discharge interval and the depth was based on the maximum channel depth determined from the LiDAR data.

5.4.3 Results

Width:depth ratios among all study sites ranged from 7.4 to 19.0 (Figure 5.4.1). The range for least-altered sites, which were forested with stable channels and high FCI scores, was between 10.4 and 17.2. The average width:depth ratio for least-altered sites was 13.4 and is similar to the approximate average width:depth ratio of 14 reported by Adams and Spotila (2005) in the Appalachian Region for streams with watersheds <200 ha. A subset of altered study sites displayed width:depth ratios

Figure 5.4.1. Comparison of width:depth ratio and Hydrology FCI score.

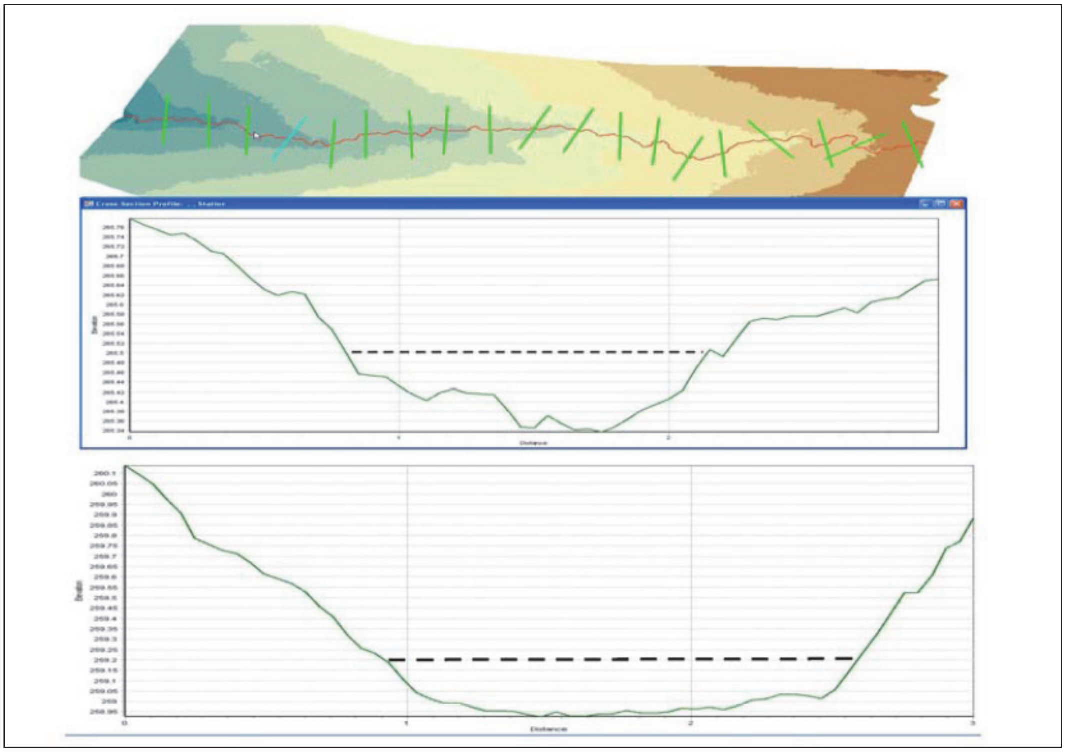


outside of this range (i.e. <10.4 and >17.2) and received lower Hydrologic FCI scores than least-altered sites. The relationship illustrated in Figure 5.4.2 shows that alterations to the watershed can cause either 1) channel down-cutting, producing a channel that is narrow and deep or 2) bankside erosion and deposition producing a channel that is wide and shallow in relation to least-altered sites. The examination of width:depth ratios in conjunction with the other metrics outlined above support the current HGM model.

5.4.4 Summary

Stream channel geometry investigations utilized width:depth ratios extracted from stream cross sections. Altered sites exhibited either down-cutting channels with low width:depth ratios (<10), or widening channels with high width:depth ratios (>18). Both low and high width:depth ratios reflect unstable channels, which is often related to hydrologic discharges that exceed channel capacity (Gordon et al. 2004).

Figure 5.4.2. From top: 1) example of cross section locations within a validation study reach, 2) representative cross sections within least-altered, and 3) altered study sites. Dashed lines display the estimated discharge frequency occurrence of 50 percent (Figure 5.2.4). Note that the width:depth ratio would be lower in the altered sites.



5.5 Hydrology function sensitivity analysis

Sensitivity analysis results demonstrate that all variables incorporated into the Hydrology model affect rapid assessment outcomes and interact as intended by model developers (Equation 5.5.1). Appendix A presents sensitivity testing results for all variables, with only a subset examined in this section. Examples and data analysis are provided for one variable displaying a strong influence over FCI scores and one variable displaying a weaker impact on FCI scores. Watershed land use (V_{WLUSE}) exhibits the strongest effect on model components (Figure 5.5.1, Table 5.5.1). When V_{WLUSE} values remain at the lowest allowable subindex score (0.0), the maximum attainable model FCI output equals 0.5 (Figure 5.5.2). When V_{WLUSE} values reach a maximum value of 1.0, the range of potential values equals 0.5 - 1.0. Increasing the V_{WLUSE} score incrementally from 0.0 to 1.0 while holding all other variables constant results in as much as a 10-fold increase in FCI scores. The selection and scaling of V_{WLUSE} as a controlling factor for the model is based on the potential of the land use from the surrounding watershed to be the driving source for water runoff and sediment dynamics within the stream (Chang 2006; Adams and Spotila 2005).

$$FCI = \left\{ \frac{V_{WLUSE} + \left[\frac{V_{LWD} + \min\left(V_{SUBSTRATE}, V_{EMBED}, V_{BERO}\right)}{2} \right]}{2} \right\} \tag{5.5.1}$$

Large, woody debris (V_{LWD}) also displays a strong impact on model outcomes, with a maximum attainable FCI score of 0.75 when V_{LWD} values remain minimized at 0.0. Under optimal conditions (i.e. V_{LWD} = 1.0), the observed output range equals 0.25 – 1.0. Large, woody debris in the stream channel and within the riparian buffer zone provide roughness that slows the flow of water to the stream channel as well as in the channel while stabilizing sediments; therefore, V_{LWD} represents another key variable within the Hydrology function.

Figure 5.5.1. Relative distribution of variable influence on Hydrology FCI scores.

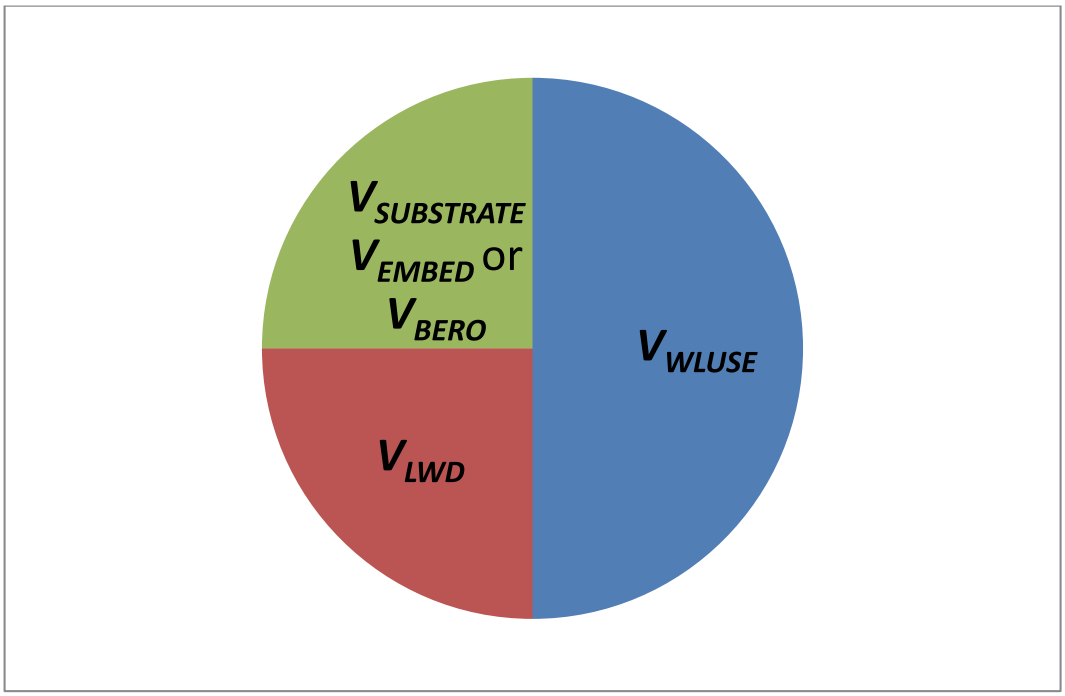


Table 5.5.1. Range of FCI scores attainable based on Hydrology sensitivity analysis (≥20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.1 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{WLUSE}	0.5	0	0.5	0.5	0.05	0.55	0.5	0.125	0.625
V_{EMBED}	0	0	0	0	0.1	0.1	0.0375	0.2125	0.25
V_{LWD}	0.25	0	0.25	0.25	0.075	0.325	0.25	0.1875	0.4375
$V_{SUBSTRATE}$	0	0	0	0.025	0.075	0.1	0.0625	0.1875	0.25
V_{BERO}	0	0	0	0.025	0.075	0.1	0.0625	0.1875	0.25
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{WLUSE}	0.5	0.25	0.75	0.5	0.375	0.875	0.5	0.5	1
V_{EMBED}	0.1	0.4	0.5	0.1625	0.5875	0.75	0.225	0.775	1
V_{LWD}	0.25	0.375	0.625	0.25	0.5625	0.8125	0.25	0.75	1
$V_{SUBSTRATE}$	0.125	0.375	0.5	0.1875	0.5625	0.75	0.25	0.75	1
V_{BERO}	0.125	0.375	0.5	0.1875	0.5625	0.75	0.25	0.75	1

The remaining variables, channel substrate size ($V_{SUBSTRATE}$), channel substrate embeddedness (V_{EMBED}), and potential channel bank erosion (V_{BERO}) exhibit the same effect of FCI scores as V_{LWD}. However, only the lowest observed score of these three variables is applied. As a result, $V_{SUBSTRATE}$, V_{BERO}, and V_{EMBED} work in concert, limiting the FCI score by the minimum value observed. This approach prevents streams with particle substrate, particle embeddedness, or erosion measurements diverging from the reference standard range from receiving high FCI scores. For example, when $V_{SUBSTRATE}$ is minimized at 0.0, the range of potential FCI score equals 0.0 – 0.75 (Figure 5.5.3). When maximized at 1.0, the observed range equals between 0.0 and 1.0. Increasing the $V_{SUBSTRATE}$ score incrementally from 0.0 to 1.0 while holding all other variables constant results in an FCI score increase of up to 33%. An FCI score of zero requires that V_{WLUSE}, V_{LWD}, and one of either $V_{SUBSTRATE}$ or V_{BERO} all equal 0.0. Achieving an FCI score of 1.0 requires all five variables exhibit values equaling 1.0.

Figure 5.5.2. Results of sensitivity test of V_{WLUSE}. Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

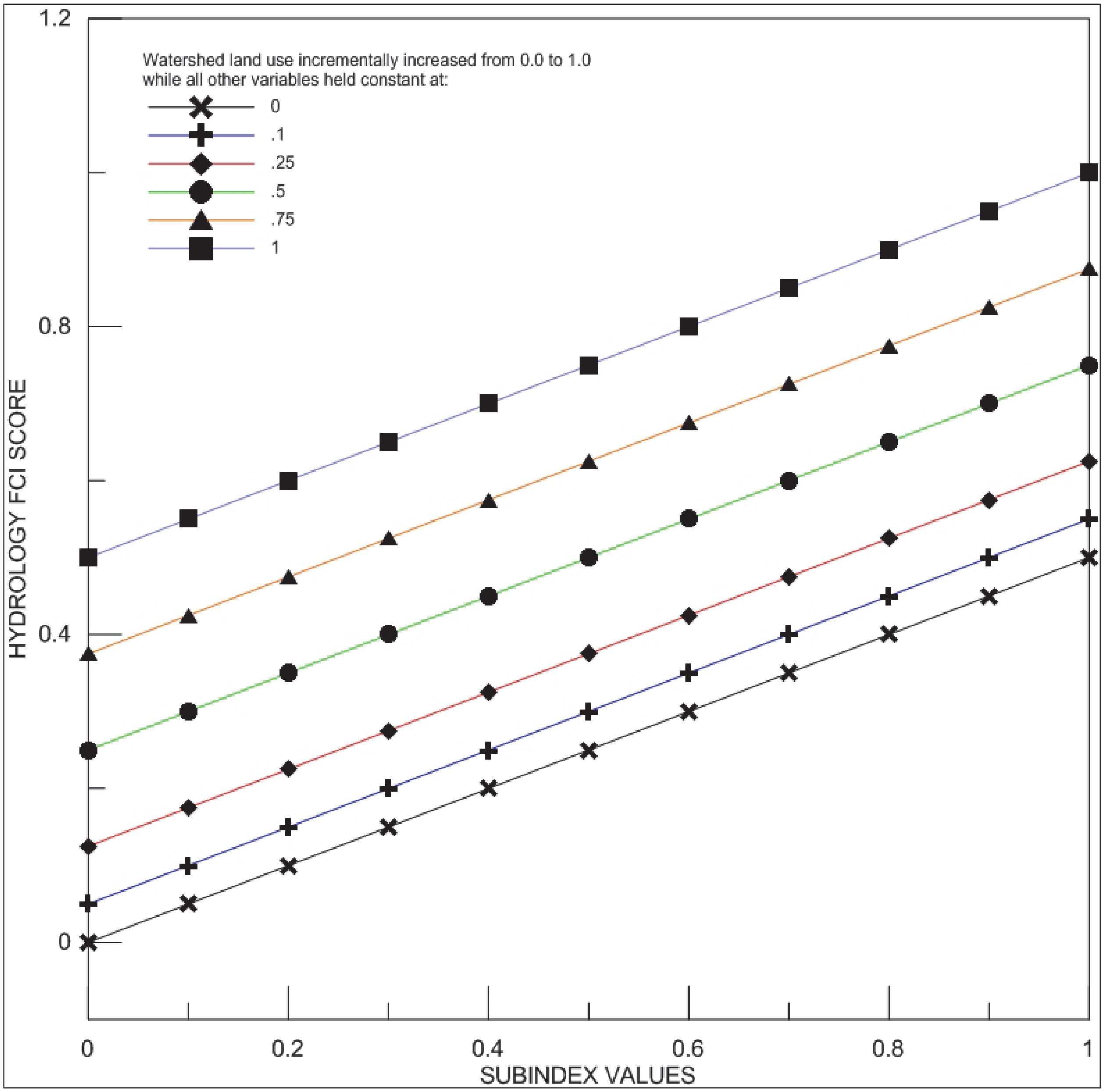
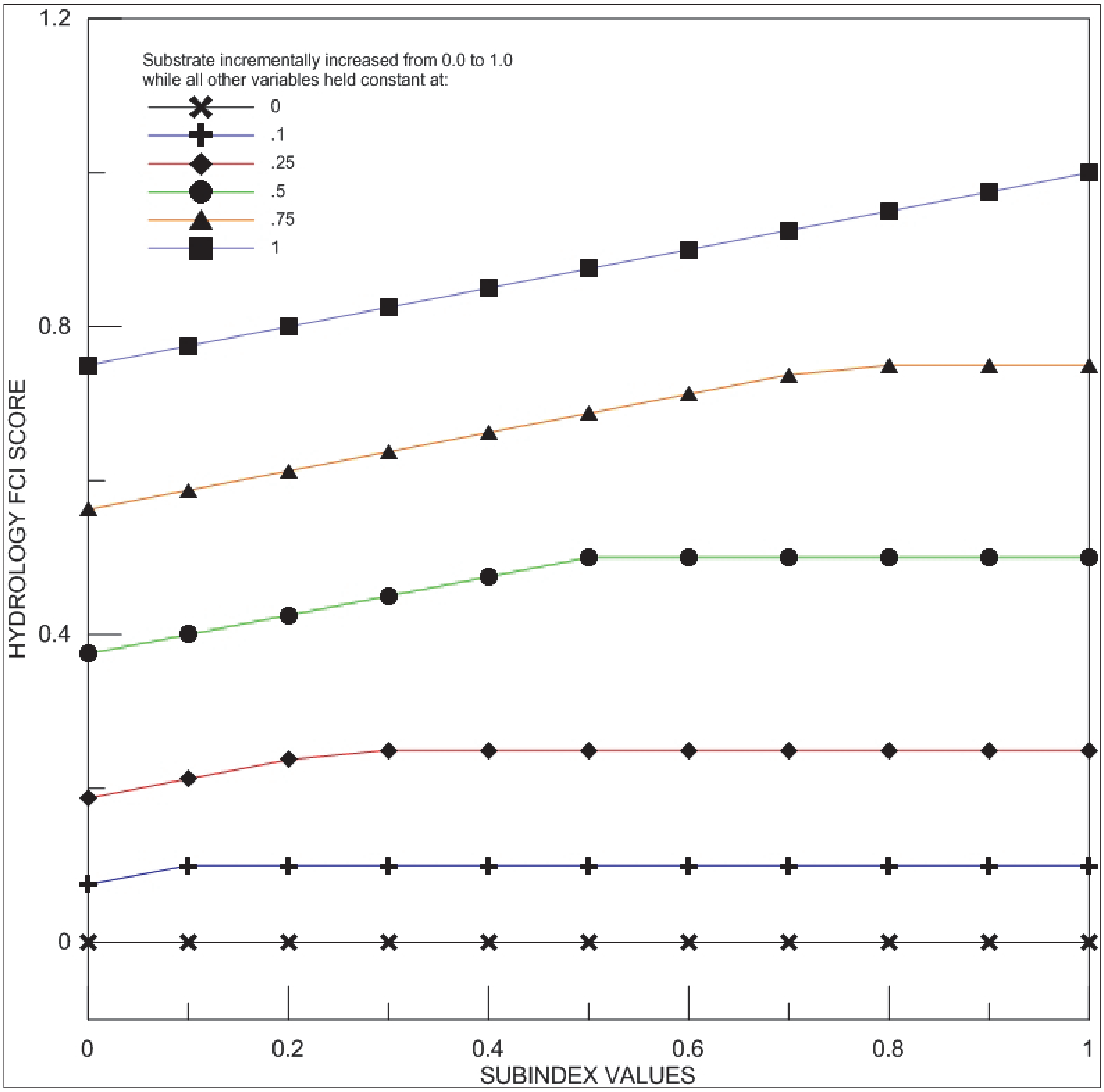


Figure 5.5.3. Results of sensitivity test of $V_{SUBSTRATE}$. Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



6 Summary

Results of the study demonstrate that the HGM models examining Habitat, Biogeochemical Cycling, and Hydrology functions differentiate between sites exhibiting a gradient of alteration. Sensitivity analysis indicates that model outputs behave as intended. The relationship between the HGM assessment models and independent measures is clear. Sites with high HGM scores perform Habitat, Biogeochemical Cycling, and Hydrology functions at increased levels compared to sites with lower scores. The HGM models effectively differentiated between high and low functioning headwater stream ecosystems and validation data supports the continued use of these models.

Validation of the Habitat function examined salamander, benthic macroinvertebrate, and riparian vegetation communities. Results indicate that the current HGM model responded appropriately. HGM Habitat scores corresponded to measures of salamander species richness and total abundance. Altered study sites exhibited lower HGM scores and contained few or no salamanders as well as decreased aquatic invertebrate fauna. HGM scores also proved an effective proxy measurement for Floristic Quality Index values.

Validation results for the Biogeochemical Cycling function support the HGM model configuration based on findings investigating nutrient inputs, processing, and stream loading of nutrients and materials. The highest HGM scores occurred at sites displaying low levels of alteration. These sites contained increased rates of nutrient inputs and material processing (e.g., decomposition, carbon release, nitrogen release). Mature forested study sites receiving high HGM scores maintained lower loading rates and demonstrated that additional biogeochemical cycling occurs within least-altered headwater stream catchments.

The hydrology validation study further supports the current HGM model with measures of surface hydrology, sediment transport and stream channel geomorphology corresponding to the gradient of site alteration examined. Flow frequency curves and the transport of sediments displayed consistent and predicted relationships at study sites with high HGM scores. Study sites with least-altered channel and watershed attributes received high HGM scores, while altered study sites displayed low HGM scores and hydrologic characteristics outside of the expected range.

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Appendix A: Sensitivity testing results

A.1 Habitat function sensitivity testing results

Table A.1.1 provides sensitivity analysis results, and Figures A.1.1 through A.1.7 provide graphical representations of sensitivity analysis results for the Habitat function for canopy cover $\geq$20 percent (Equation A.1.1). A detailed discussion of sensitivity analysis results is provided in section 3.5.

Table A.1.2 provides sensitivity analysis results, and Figures A.1.8 through A.1.14 provide graphical representations of sensitivity analysis results for the Habitat function for canopy cover <20 percent (Equation A.1.2). A detailed discussion of sensitivity analysis results is provided in section 3.5.

$$FCI = \left\langle \left[\frac{V_{CCANOPY} + \min\left(V_{EMBED}, V_{SUBSTRATE}\right)}{2}\right] \times \left\{\frac{\left(\frac{V_{LWD} + V_{DETRITUS}}{2}\right) + \left[\frac{\left(\frac{V_{SNAG} + V_{TDBH} + V_{SRICH}}{3}\right) + V_{WLUSE}}{2}\right]}{2}\right\} \right\rangle^{1/2} \quad \text{(A.1.1)}$$

Table A.1.1. Range of FCI scores attainable based on Habitat sensitivity analysis (≥20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.1 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
$V_{CCANOPY}$	0.06	0.03	0.09	0.13	0.10	0.23	0.19	0.21	0.40
V_{EMBED}	0.00	0.03	0.03	0.00	0.10	0.10	0.04	0.21	0.25
$V_{SUBSTRATE}$	0.01	0.03	0.04	0.03	0.07	0.10	0.07	0.18	0.25
V_{LWD} / $V_{DETRITUS}$ / V_{WLUSE}	0.09	0.03	0.12	0.09	0.09	0.18	0.11	0.22	0.33
V_{TDBH} / V_{SNAG}	0.04	0.03	0.07	0.03	0.10	0.13	0.04	0.24	0.28
V_{SRICH}	0.04	0.03	0.07	0.04	0.10	0.13	0.04	0.24	0.28
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
$V_{CCANOPY}$	0.23	0.39	0.61	0.25	0.56	0.81	0.26	0.74	1.00

V_{EMBED}	0.11	0.39	0.50	0.19	0.56	0.75	0.26	0.74	1.00
$V_{SUBSTRATE}$	0.15	0.35	0.50	0.22	0.53	0.75	0.29	0.71	1.00
V_{LWD} / $V_{DETRITUS}$ / V_{WLUSE}	0.13	0.43	0.56	0.13	0.65	0.78	0.13	0.87	1.00
V_{TDBH} / V_{SNAG}	0.04	0.48	0.52	0.04	0.72	0.76	0.04	0.96	1.00
V_{SRICH}	0.04	0.48	0.52	0.04	0.72	0.76	0.04	0.96	1.00

Figure A.1.1. Relative distribution of variable influence on Habitat FCI values in areas displaying ≥20 percent canopy cover.

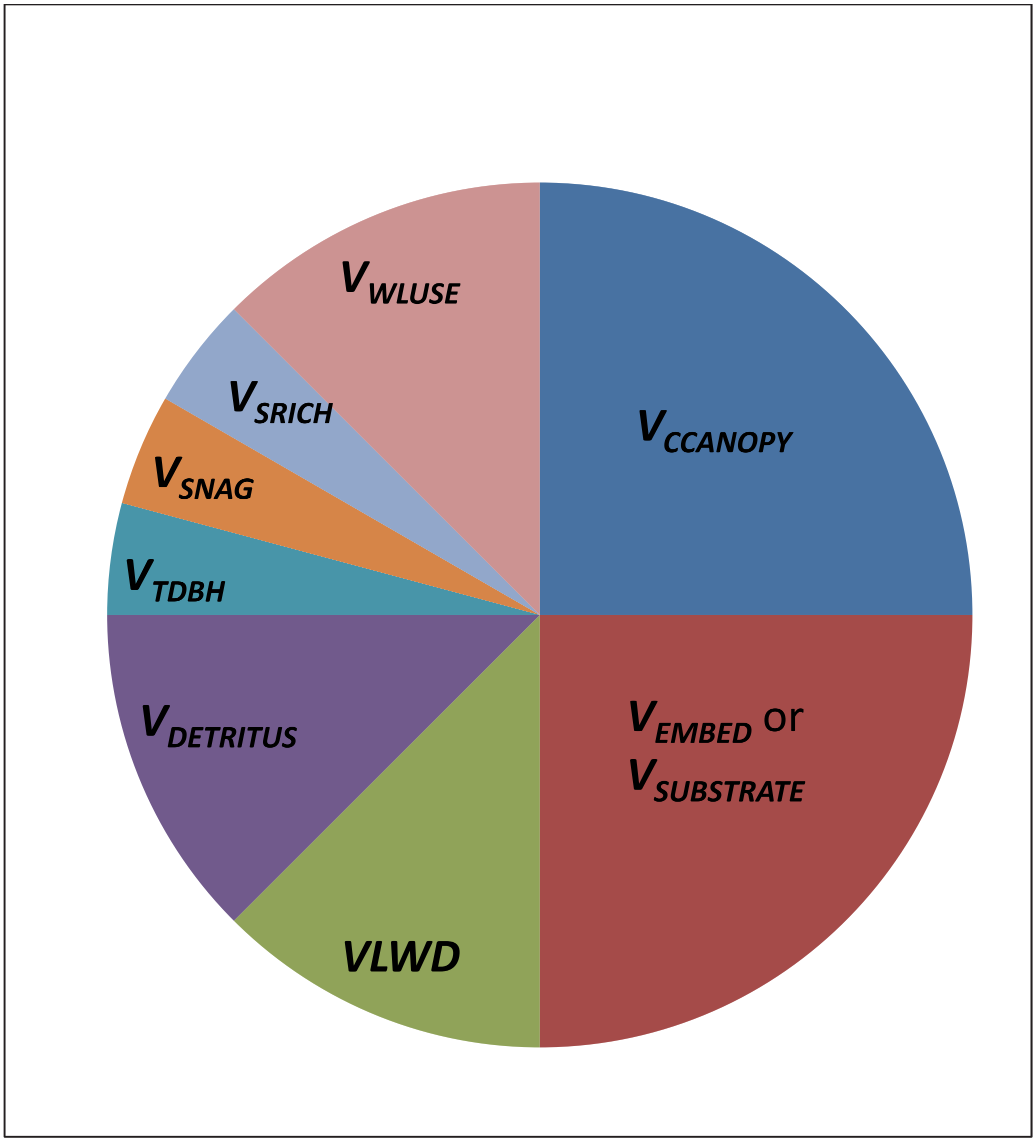


Figure A.1.2. Results of sensitivity test of V_{CANOPY} (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

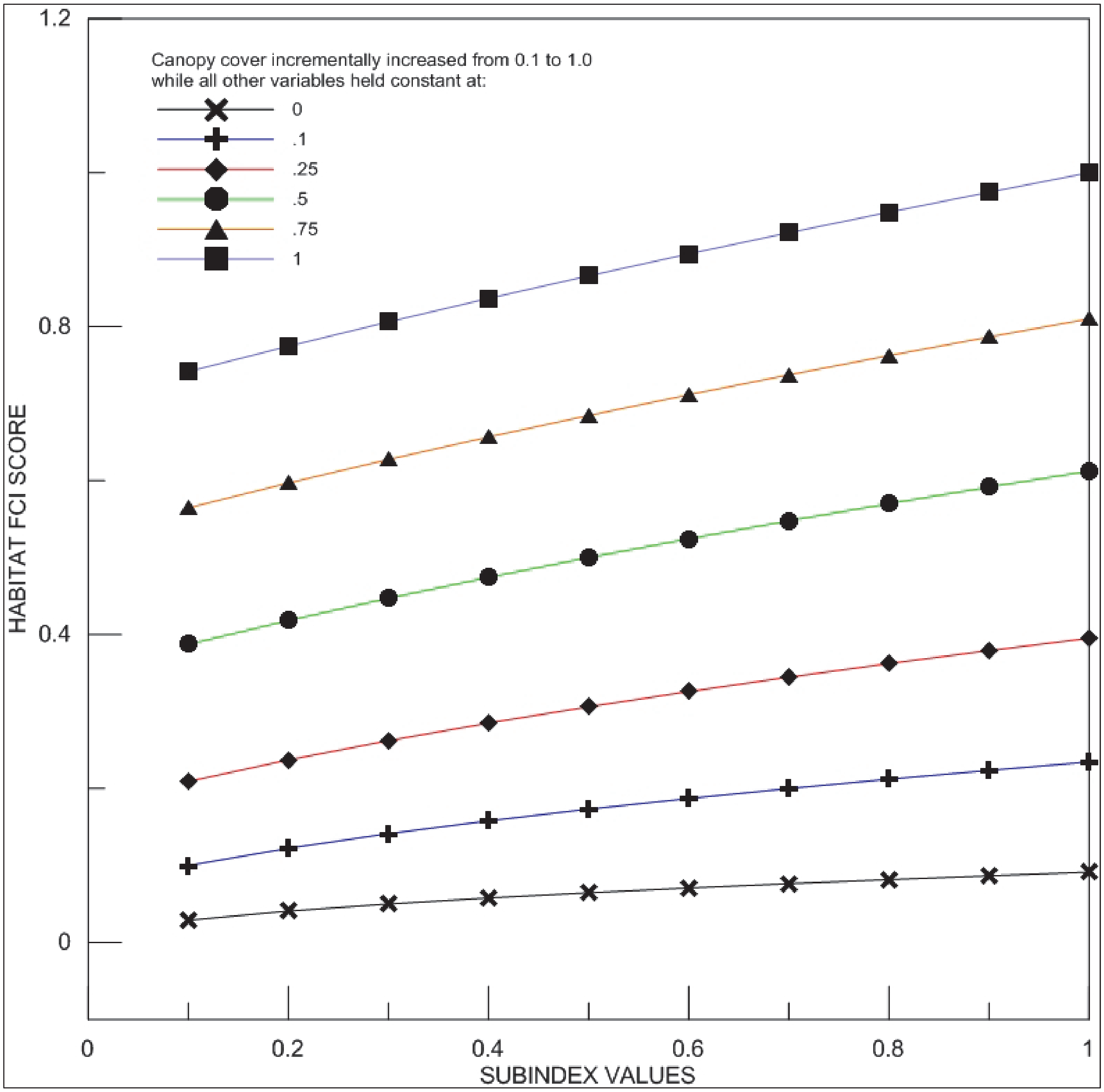


Figure A.1.3. Results of sensitivity test of V_{EMBED} (canopy cover $\geq$20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

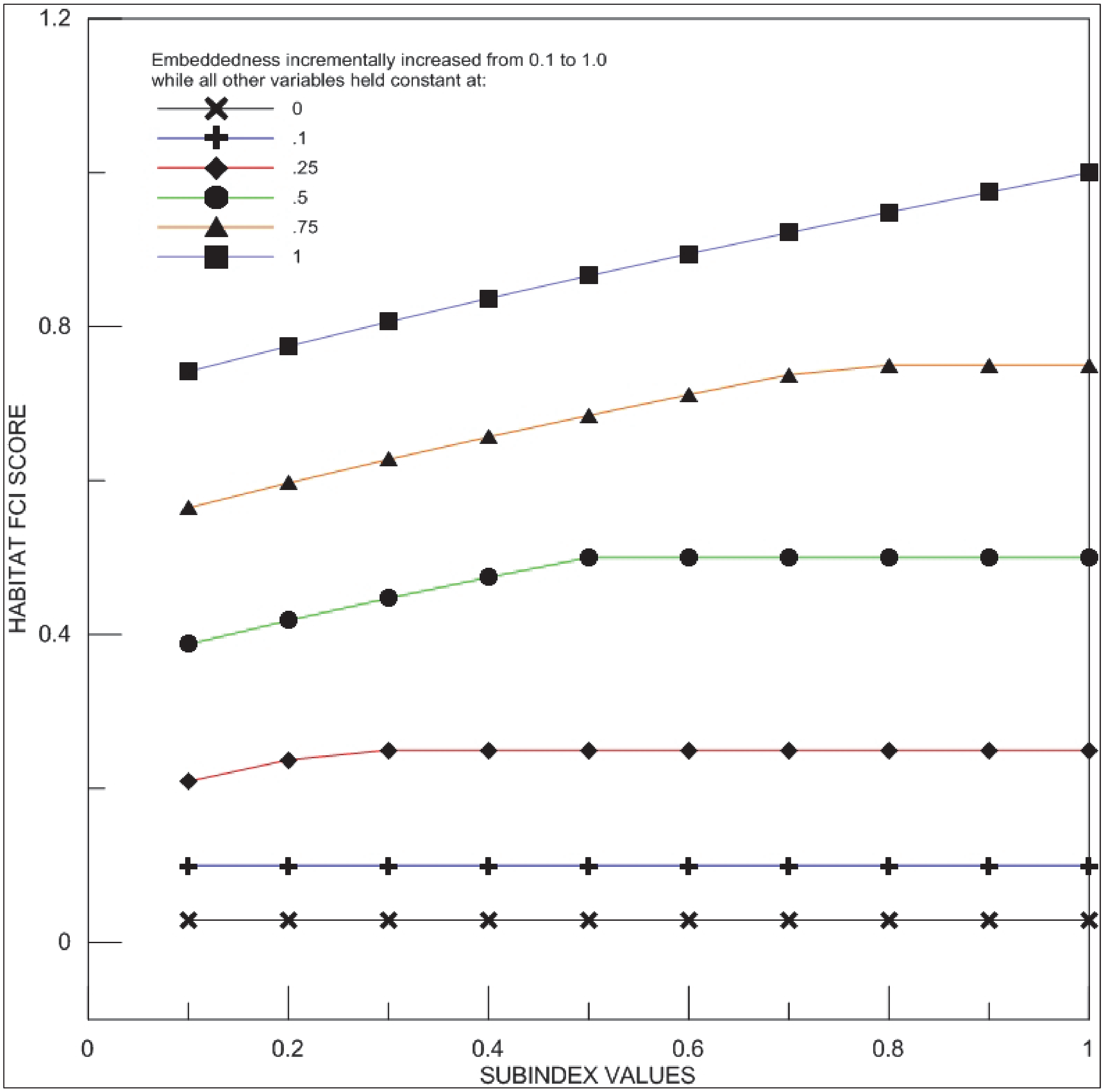


Figure A.1.4. Results of sensitivity test of $V_{SUBSTRATE}$ (canopy cover >20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

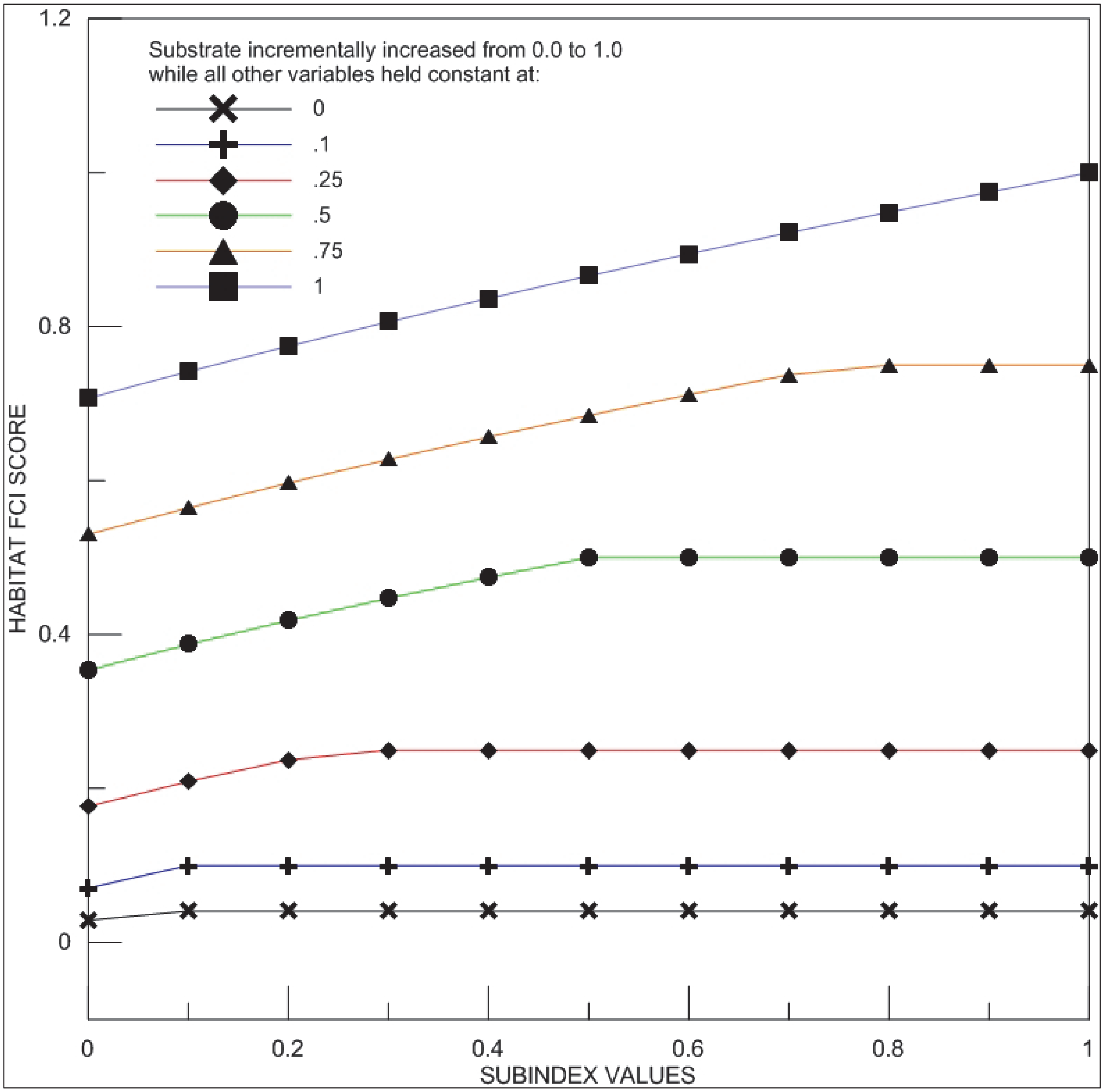


Figure A.1.5. Results of sensitivity test of V_{LWD} (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0. Results for $V_{DETRITUS}$ and V_{WLUSE} are identical.

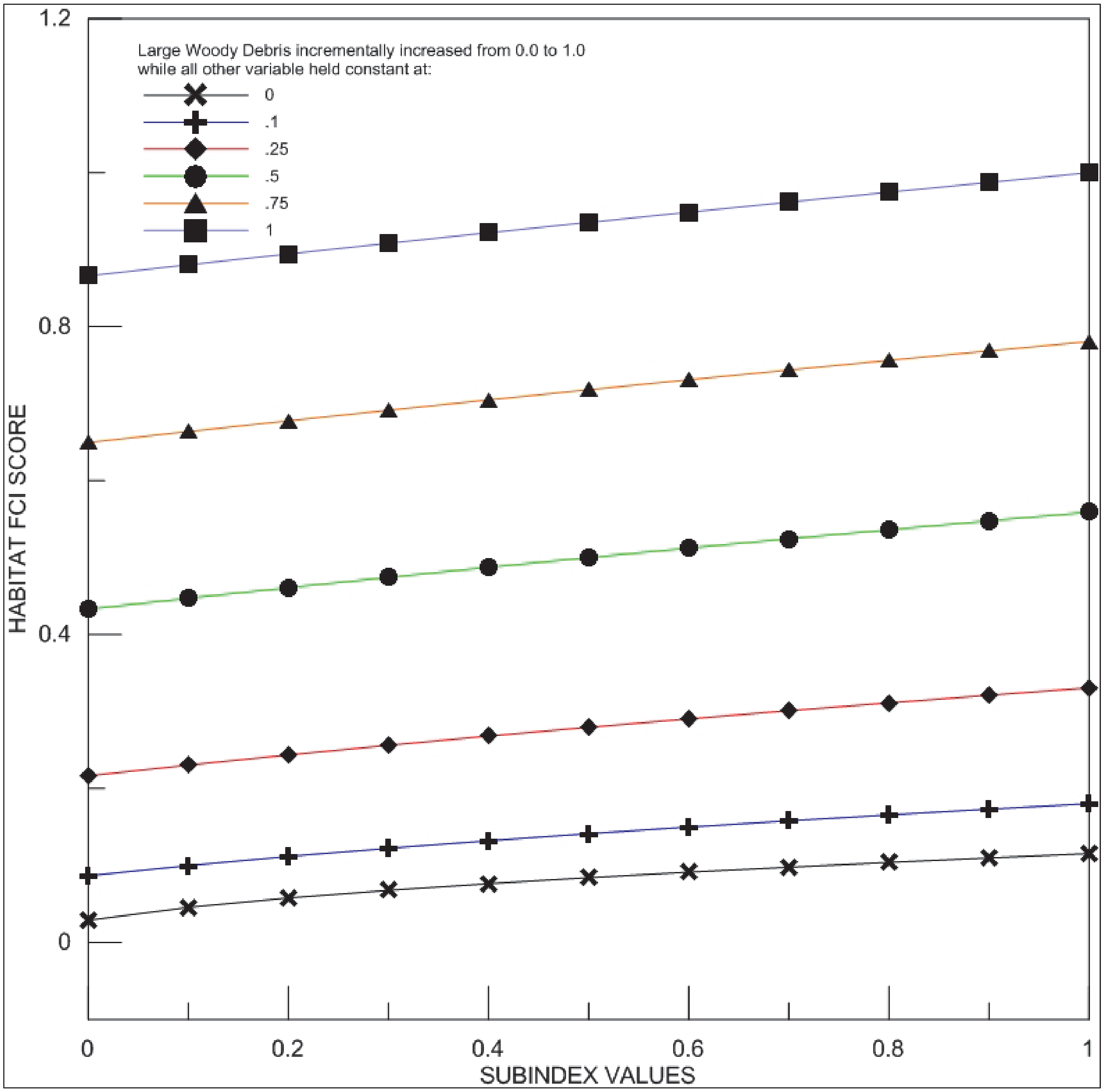


Figure A.1.6. Results of sensitivity test of V_{TDBH} (canopy cover $\geq$20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0. Results for V_{SNAG} are identical.

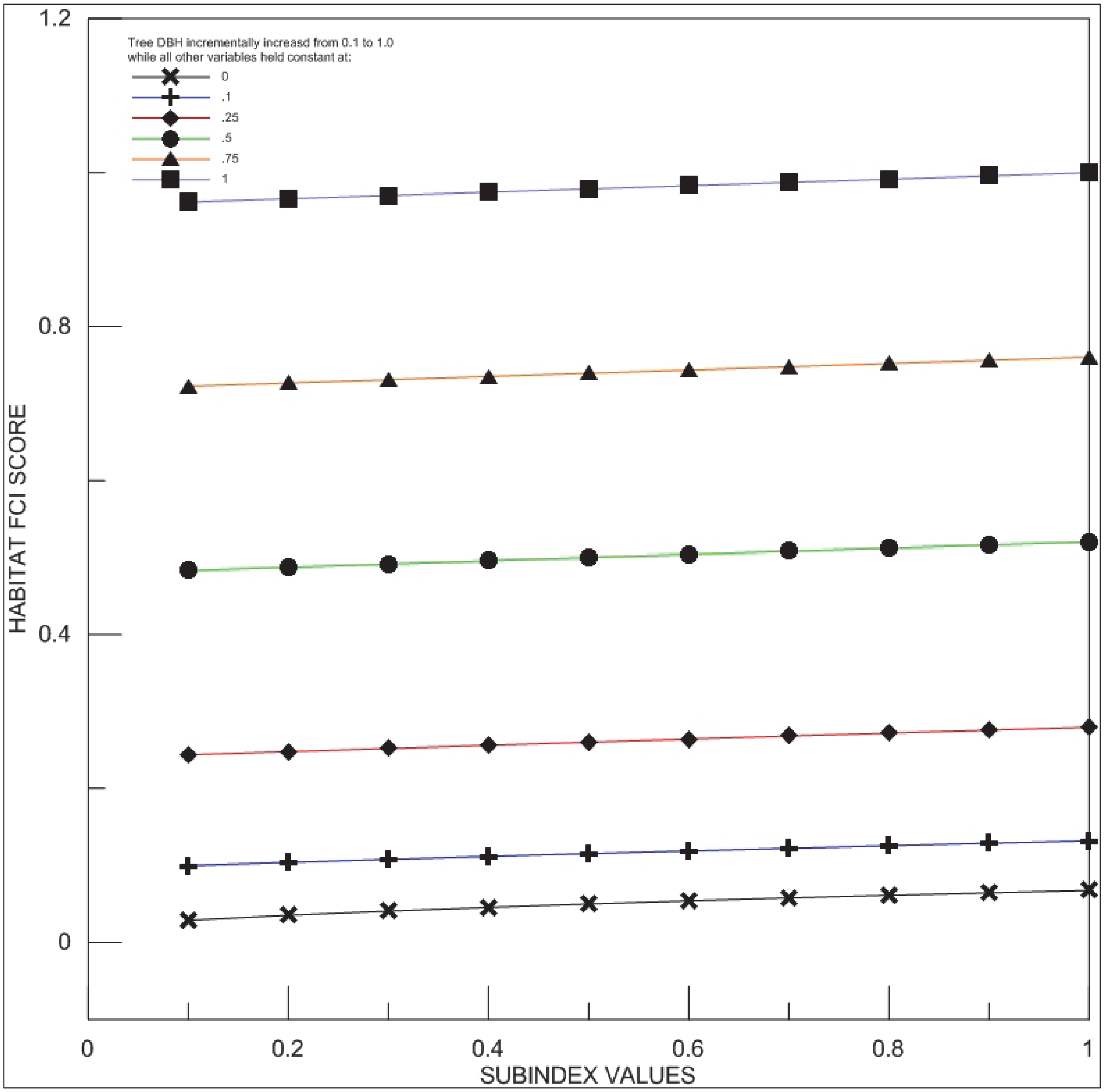
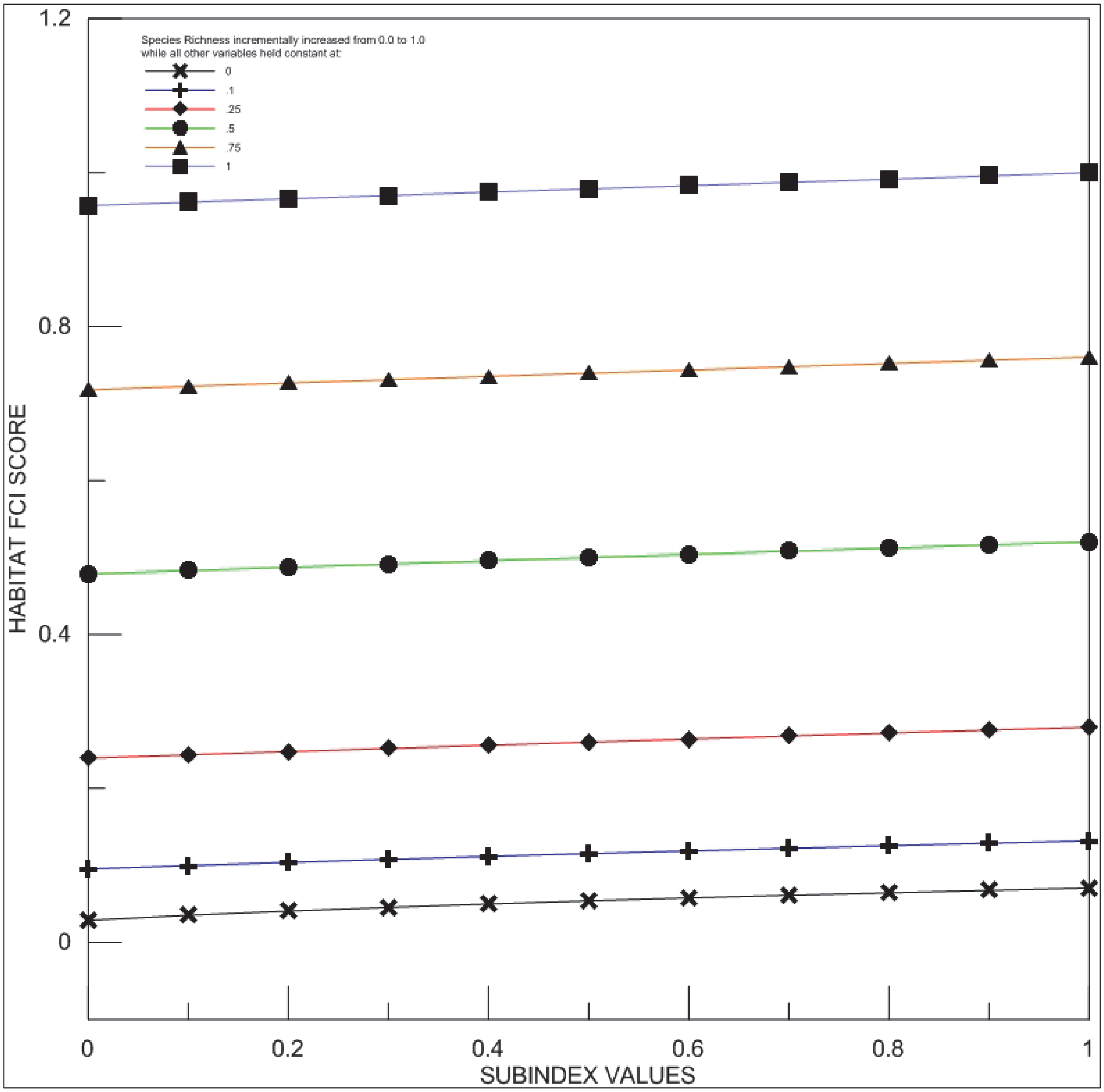


Figure A.1.7. Results of sensitivity test of V_{SRICH} (canopy cover >20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



$$FCI = \left\{ \min\left(V_{EMBED}, V_{SUBSTRATE}\right) \times \left\{ \frac{\left(\frac{V_{LWD} + V_{DETRITUS}}{2}\right) + \left[\frac{\left(\frac{V_{SNAG} + V_{SSD} + V_{HERB} + V_{SRICH}}{6}\right) + V_{WLUSE}}{4}\right]}{2} \right\} \right\}^{1/2} \quad (A.1.2)$$

Table A.1.2. Range of FCI scores attainable based on habitat sensitivity analysis (<20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.1 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.00	0.00	0.00	0.00	0.08	0.08	0.08	0.13	0.21
$V_{SUBSTRATE}$	0.01	0.00	0.01	0.08	0.00	0.08	0.21	0.00	0.21
V_{LWD} / $V_{DETRITUS}$	0.00	0.00	0.00	0.10	0.07	0.17	0.13	0.17	0.30
V_{WLUSE}	0.00	0.00	0.00	0.06	0.08	0.14	0.07	0.19	0.26
V_{SNAG} / V_{SSD} / V_{HERB} / V_{SRICH}	0.00	0.00	0.00	0.01	0.08	0.09	0.01	0.21	0.22

Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.23	0.19	0.42	0.40	0.23	0.63	0.58	0.27	0.84
$V_{SUBSTRATE}$	0.42	0.00	0.42	0.63	0.00	0.63	0.84	0.00	0.84
V_{LWD} / $V_{DETRITUS}$	0.15	0.34	0.49	0.16	0.51	0.67	0.16	0.68	0.84
V_{WLUSE}	0.07	0.38	0.46	0.08	0.57	0.65	0.08	0.76	0.84
V_{SNAG} / V_{SSD} / V_{HERB} / V_{SRICH}	0.01	0.42	0.43	0.01	0.62	0.63	0.01	0.83	0.84

Figure A.1.8. Relative distribution of variable influence on habitat FCI values in areas displaying <20 percent canopy cover.

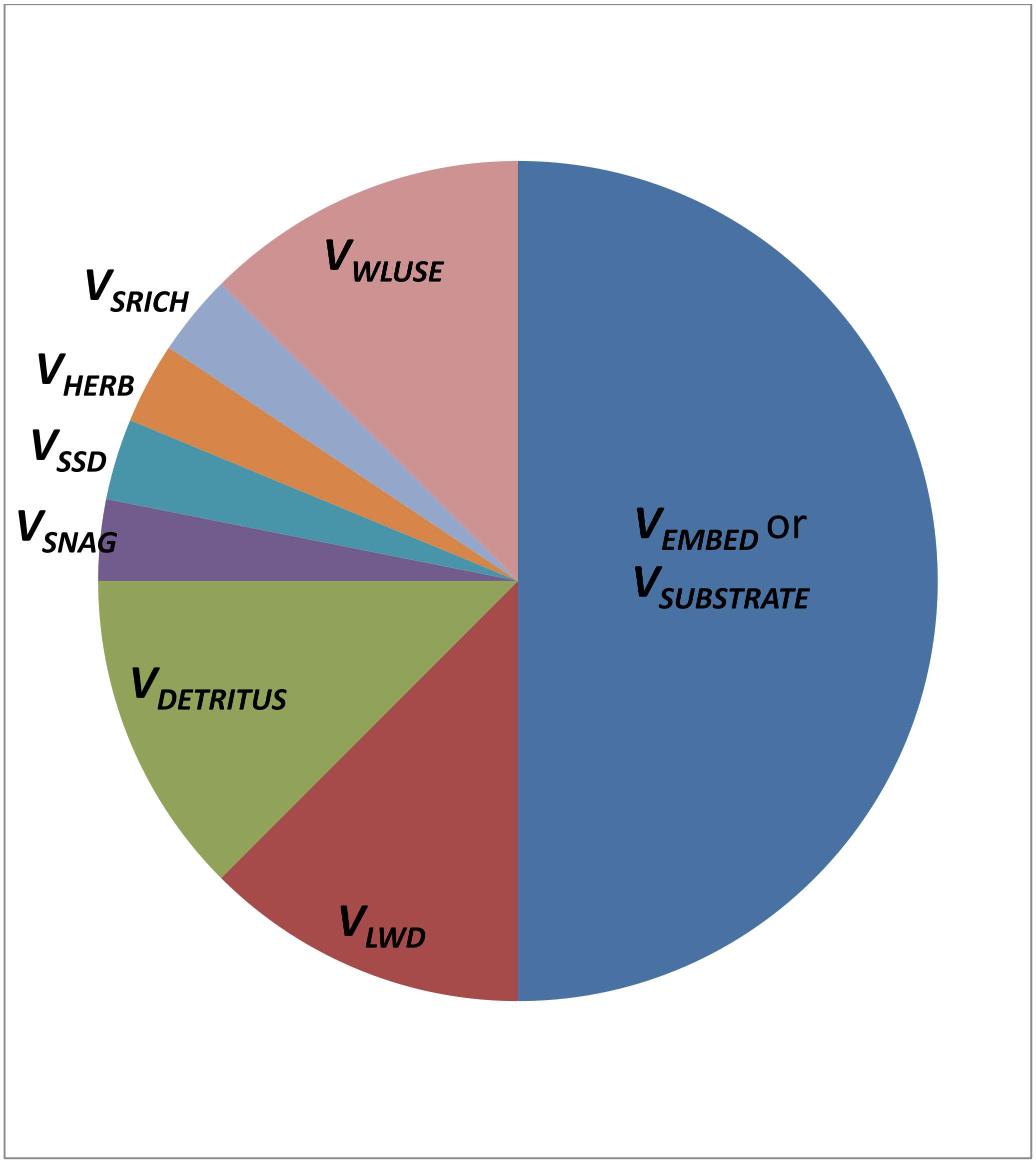


Figure A.1.9. Results of sensitivity test of V_{EMBED} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

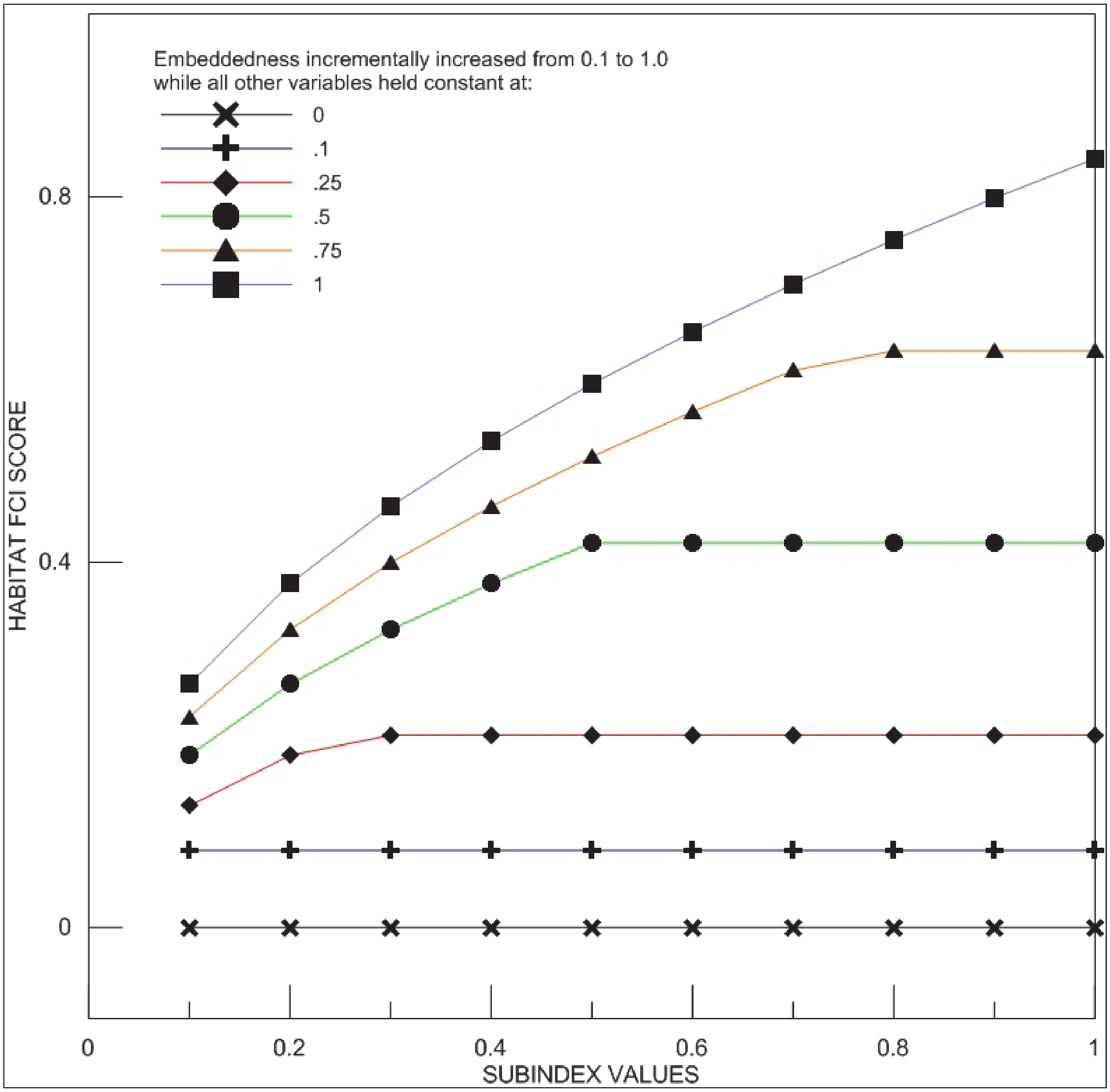


Figure A.1.10. Results of sensitivity test of $V_{SUBSTRATE}$ (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

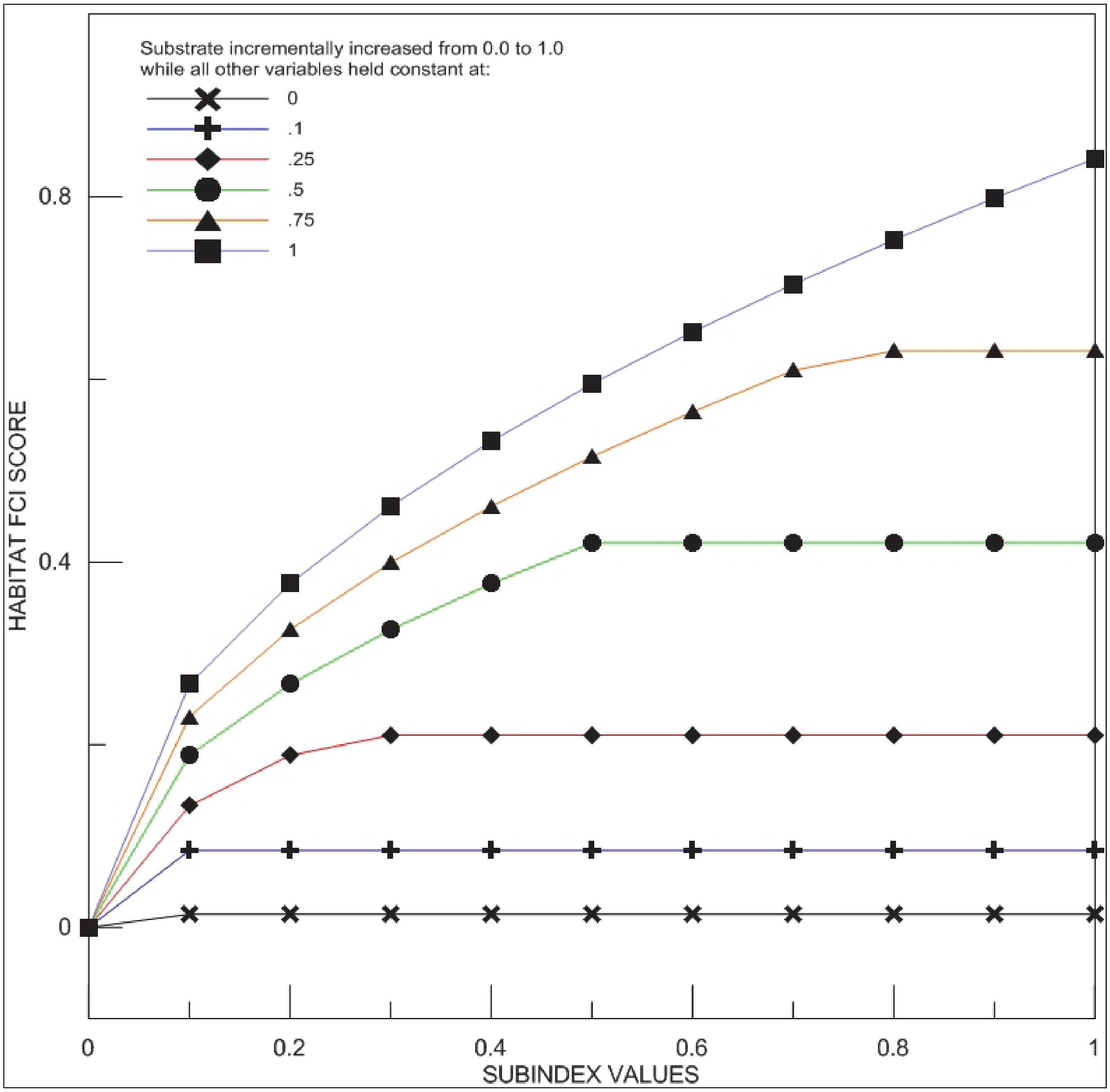


Figure A.1.11. Results of sensitivity test of V_{LWD} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0. Results for $V_{DETRITUS}$ are identical.

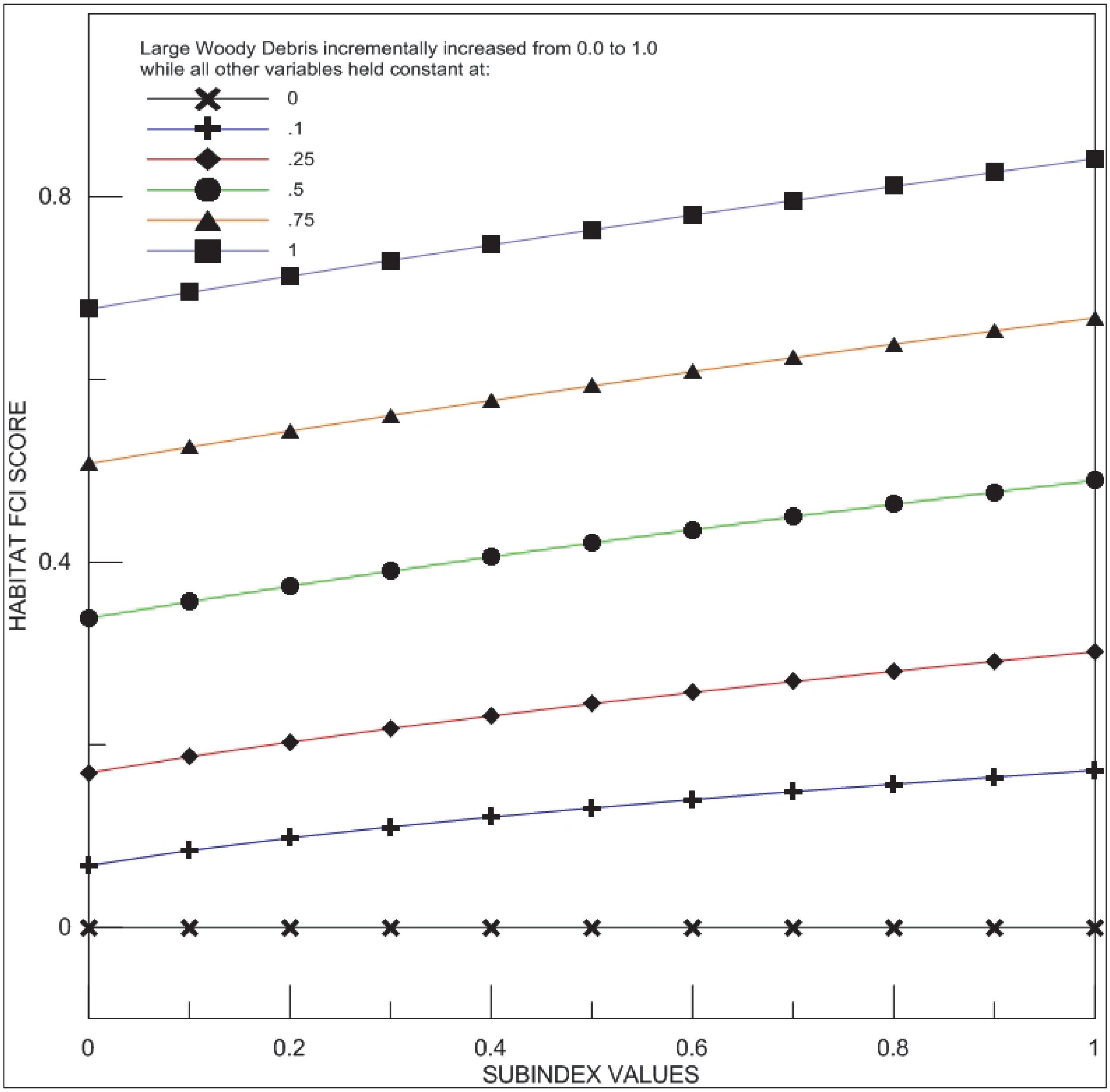


Figure A.1.12. Results of sensitivity test of V_{WLUSE} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

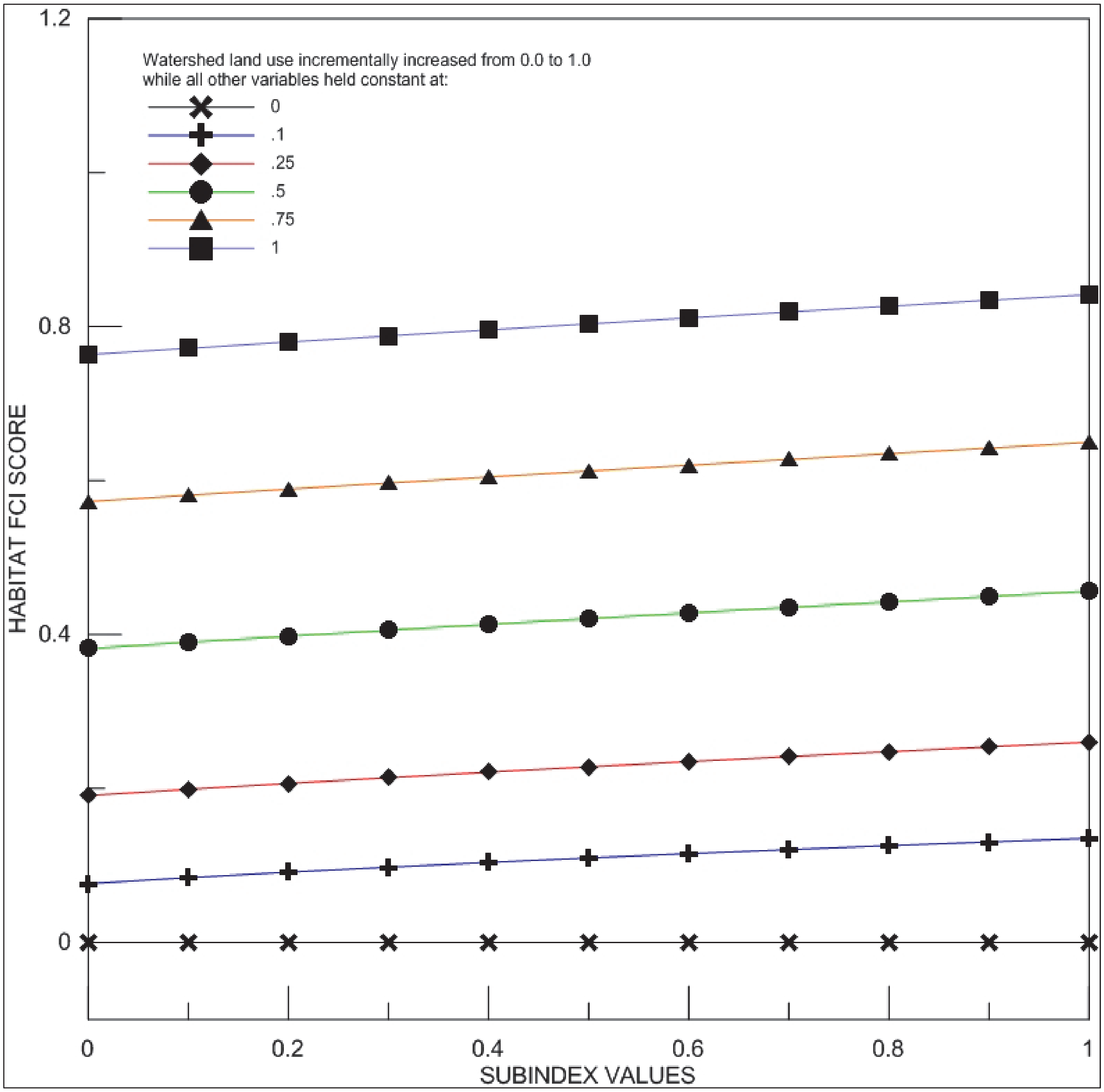


Figure A.1.13. Results of sensitivity test of V_{SNAG} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

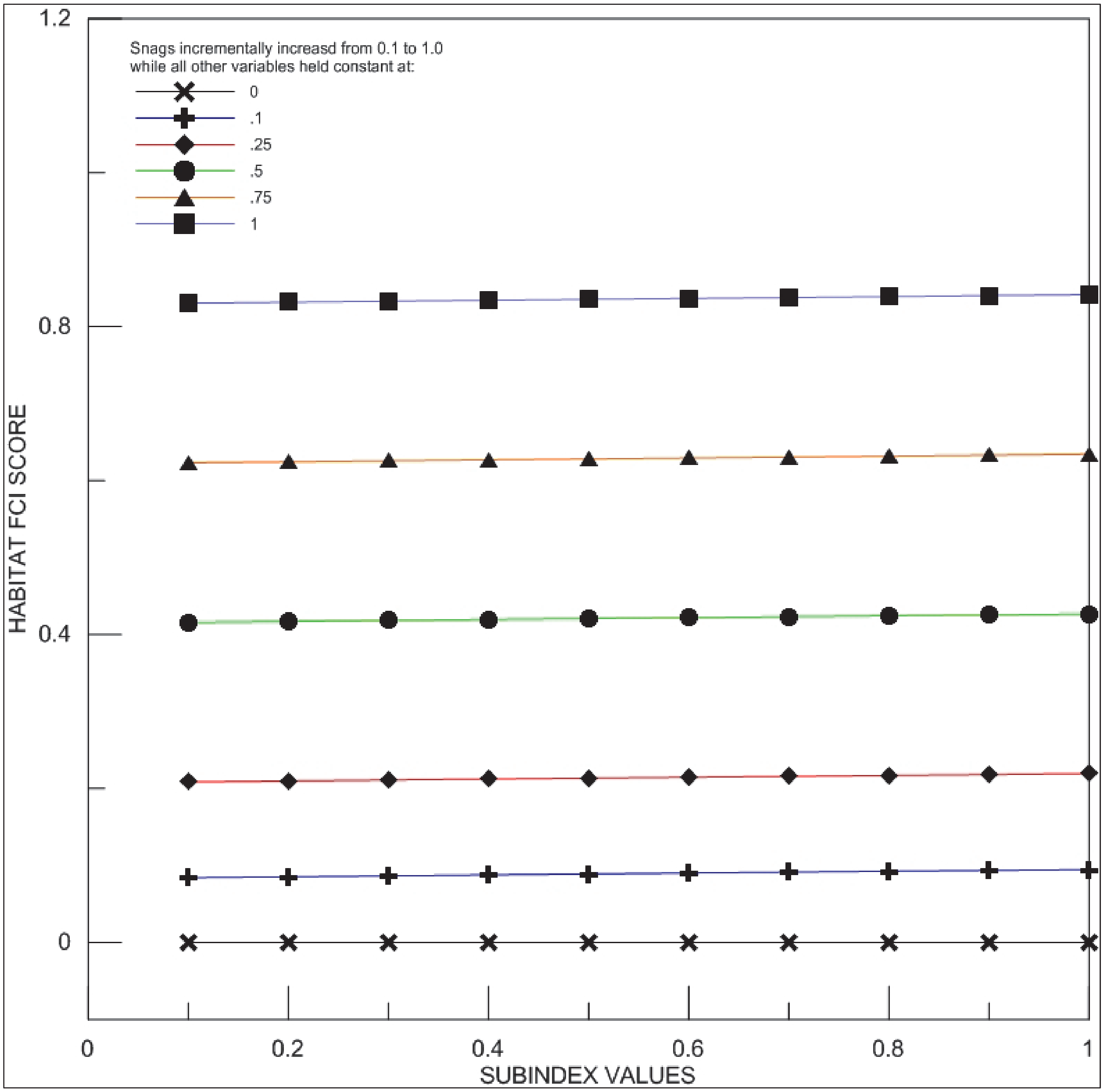
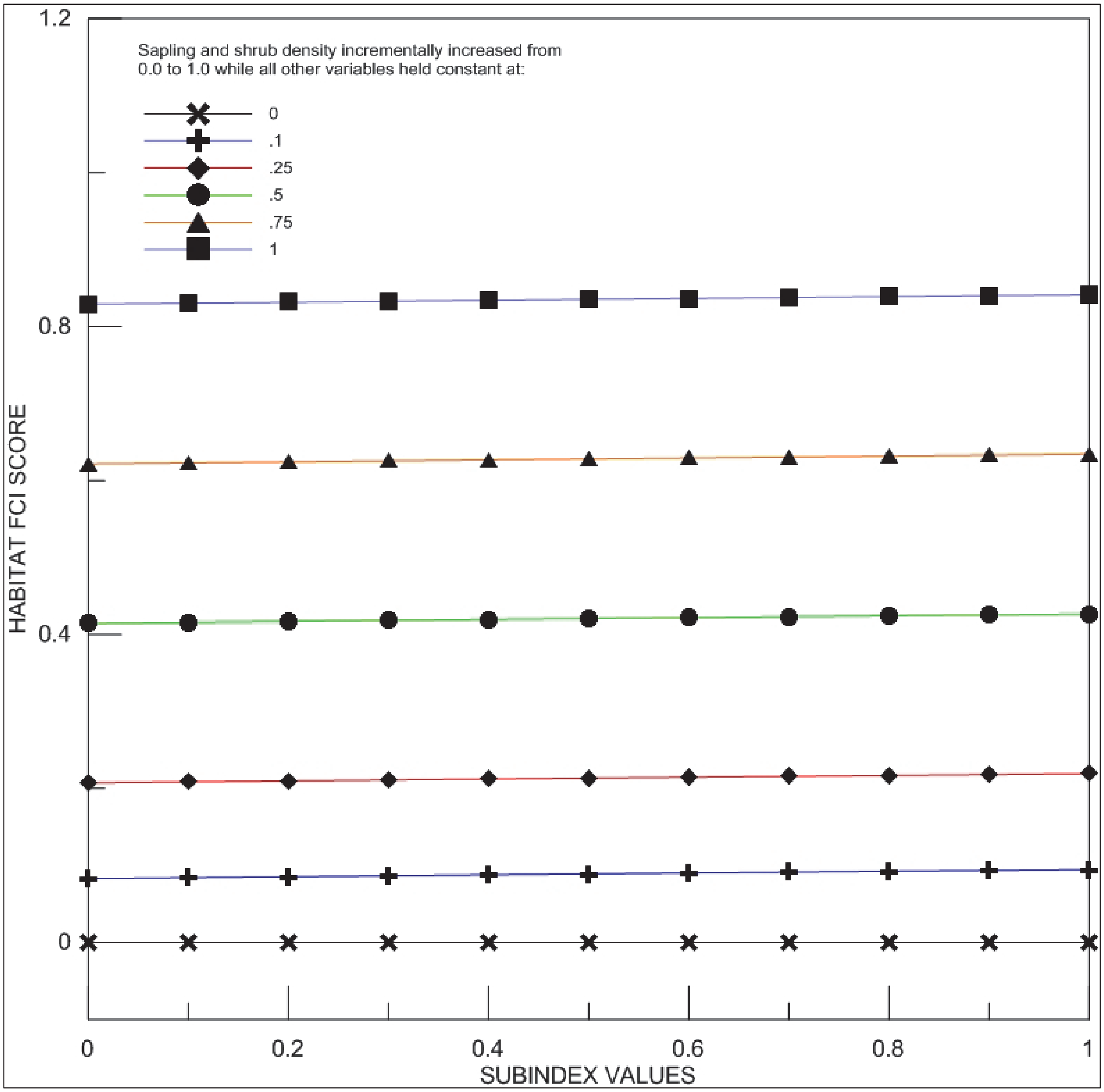


Figure A.1.14. Results of sensitivity test of V_{SSD} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0. Results for V_{HERB} and V_{RICH} are identical.



A.2 Biogeochemical Cycling function sensitivity testing results

Table A.2.1 Provides sensitivity analysis results, and Figures A.2.1 through A.2.6 provide graphical representations of sensitivity analysis results for the Biogeochemical Cycling function for canopy cover ≥20 percent (Equation A.2.1). A detailed discussion of sensitivity analysis results is provided in section 4.5.

Table A.2.2 Provides sensitivity analysis results, and Figures A.2.7 through A.2.13 provide graphical representations of sensitivity analysis results for the Biogeochemical Cycling function for canopy cover <20 percent (Equation A.2.2). A detailed discussion of sensitivity analysis results is provided in section 3.6.

$$FCI = \left\{ V_{EMBED} \times \left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{TDBH}}{3} \right) + V_{WLUSE}}{2} \right] \right\}^{1/2} \qquad (A.2.1)$$

Table A.2.1. Range of FCI scores attainable based on Biogeochemistry Cycling sensitivity analysis (≥20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.10, 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.09	0.04	0.13	0.22	0.10	0.32	0.34	0.16	0.50
V_{LWD}	0.09	0.04	0.14	0.07	0.09	0.16	0.08	0.23	0.31
$V_{DETRITUS}$	0.09	0.04	0.14	0.07	0.09	0.16	0.08	0.23	0.31
V_{TDBH}	0.09	0.04	0.13	0.07	0.09	0.16	0.08	0.23	0.31
V_{WLUSE}	0.19	0.04	0.23	0.16	0.07	0.23	0.22	0.18	0.40

Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.48	0.22	0.71	0.59	0.27	0.87	0.68	0.32	1.00
V_{LWD}	0.08	0.46	0.54	0.09	0.68	0.77	0.09	0.91	1.00
$V_{DETRITUS}$	0.08	0.46	0.54	0.09	0.68	0.77	0.09	0.91	1.00
V_{TDBH}	0.08	0.46	0.54	0.09	0.68	0.77	0.09	0.91	1.00
V_{WLUSE}	0.26	0.35	0.61	0.28	0.53	0.81	0.29	0.71	1.00

Figure A.2.1. Relative distribution of variable influence on Biogeochemical Cycling FCI scores in areas displaying ≥20 percent canopy cover.

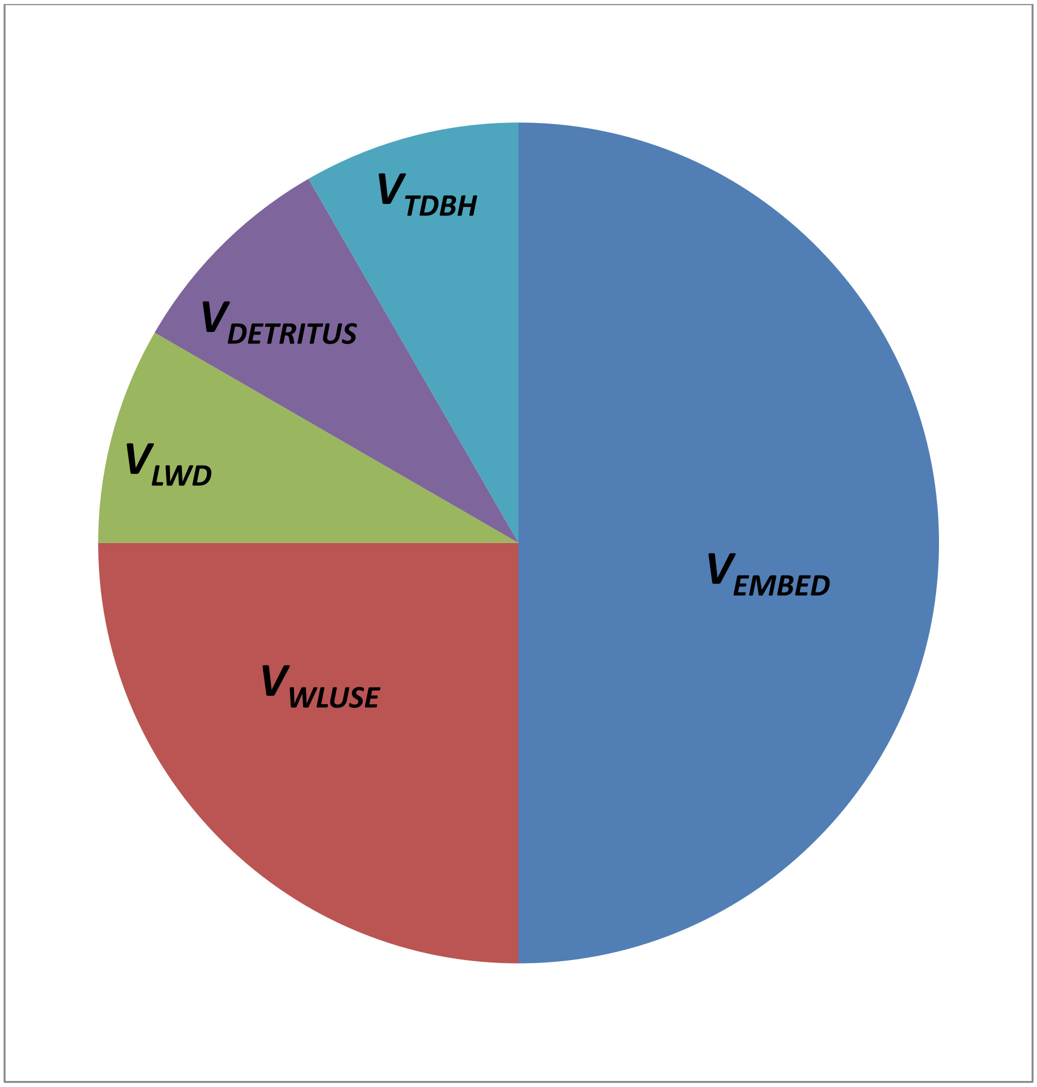


Figure A.2.2. Results of sensitivity test of V_{EMBED} (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

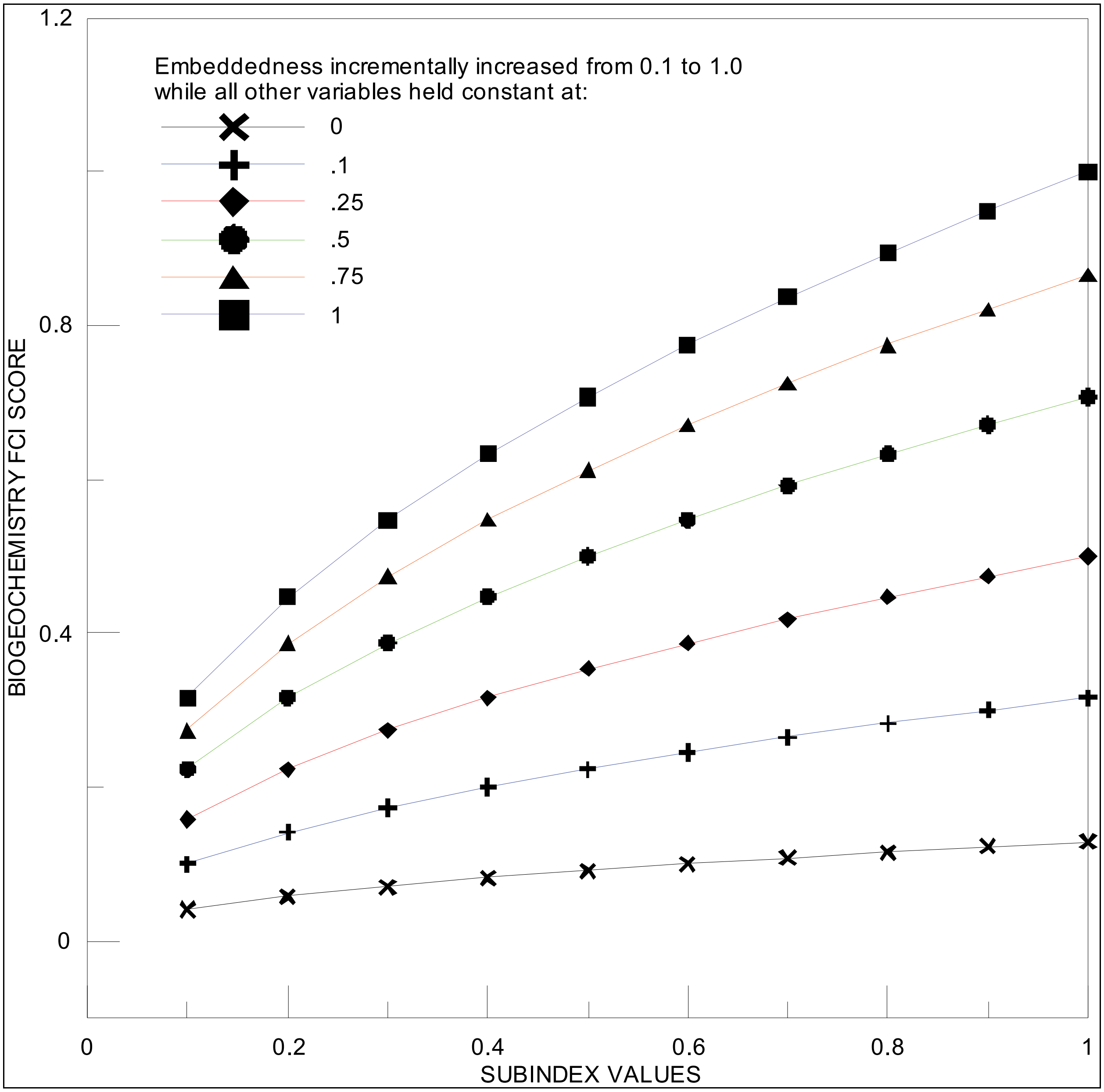


Figure A.2.3. Results of sensitivity test of V_{LWD} (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

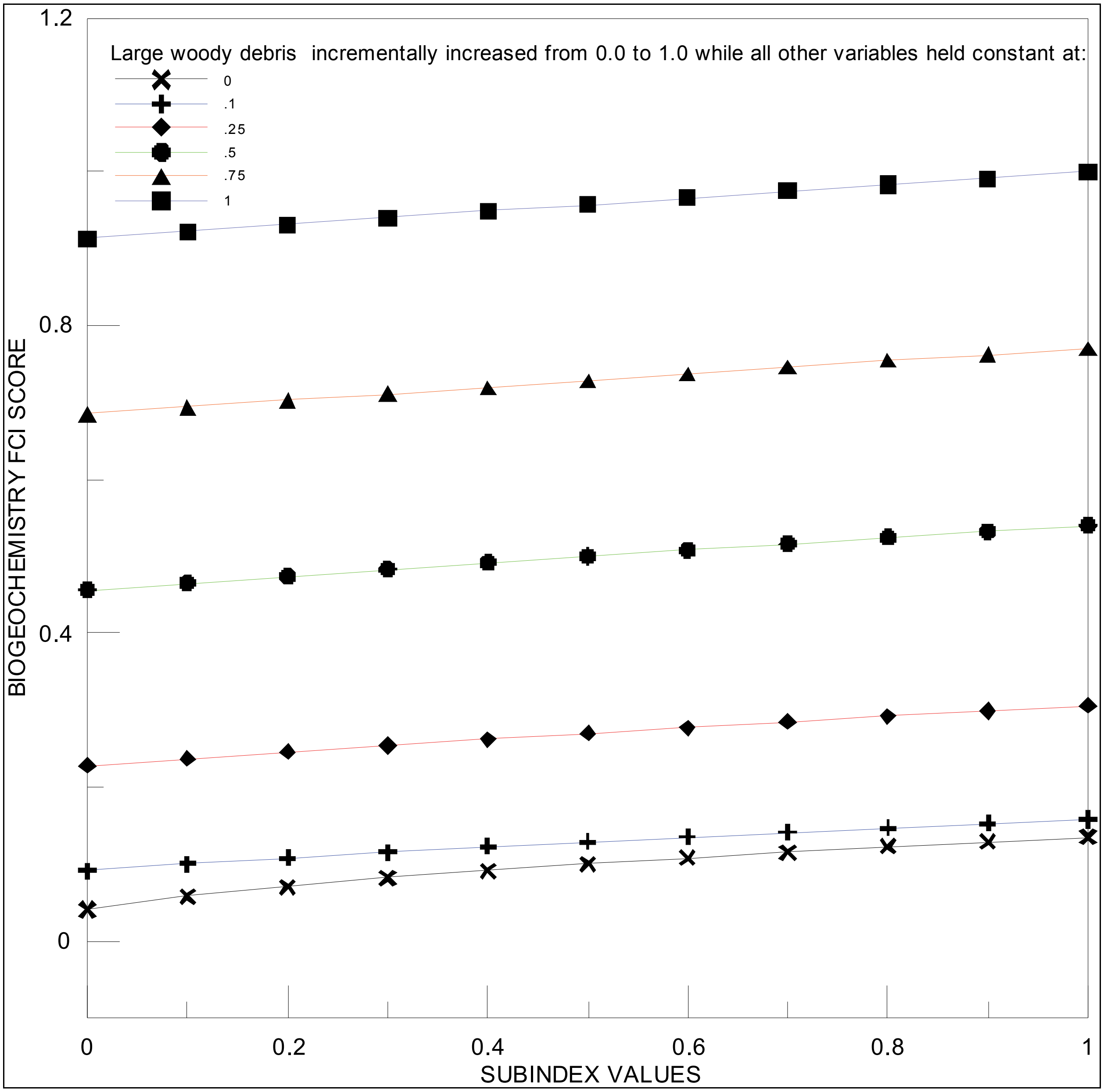


Figure A.2.4. Results of sensitivity test of $V_{DETRITUS}$ (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

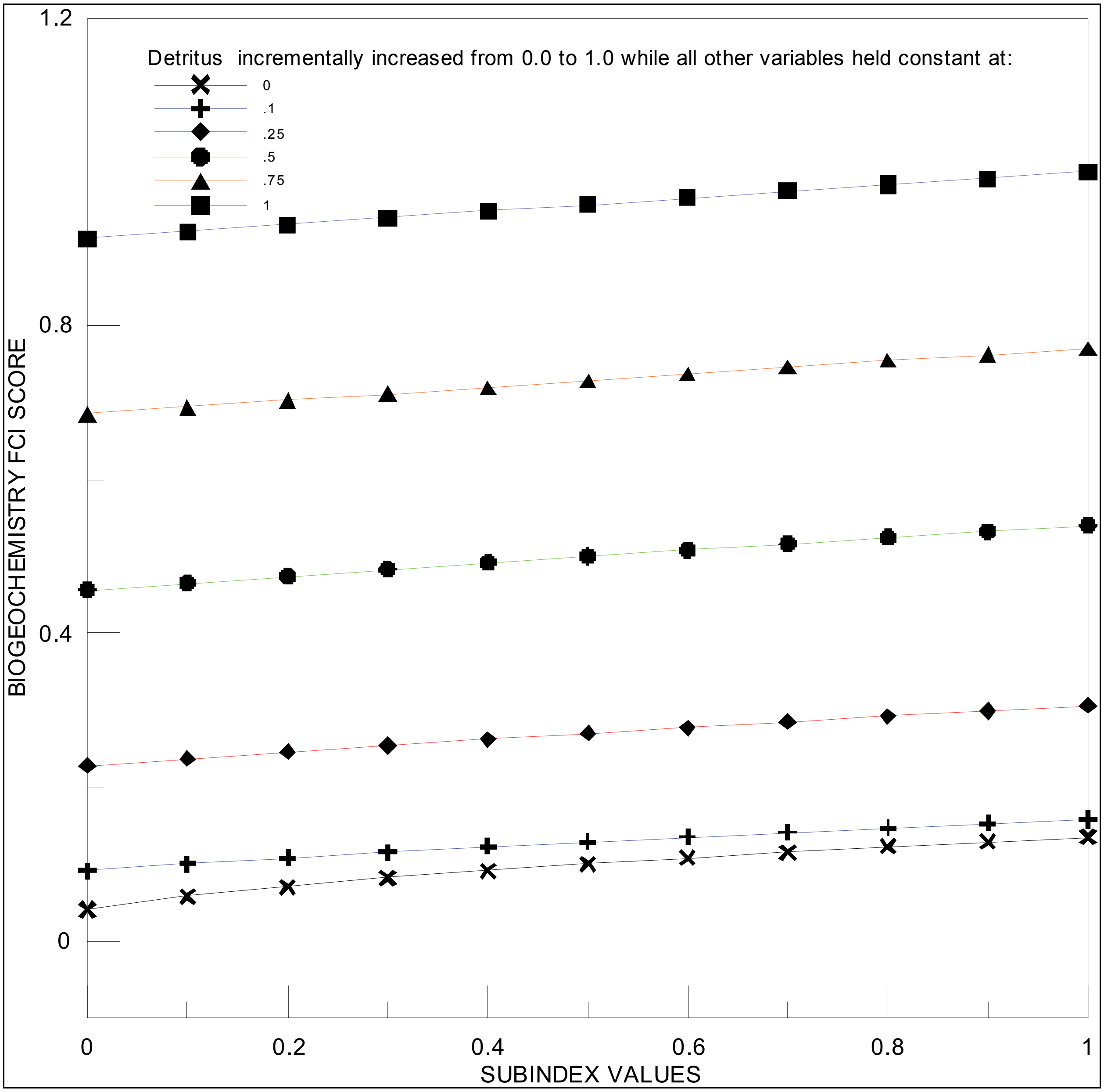


Figure A.2.5. Results of sensitivity test of V_{DBH} (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

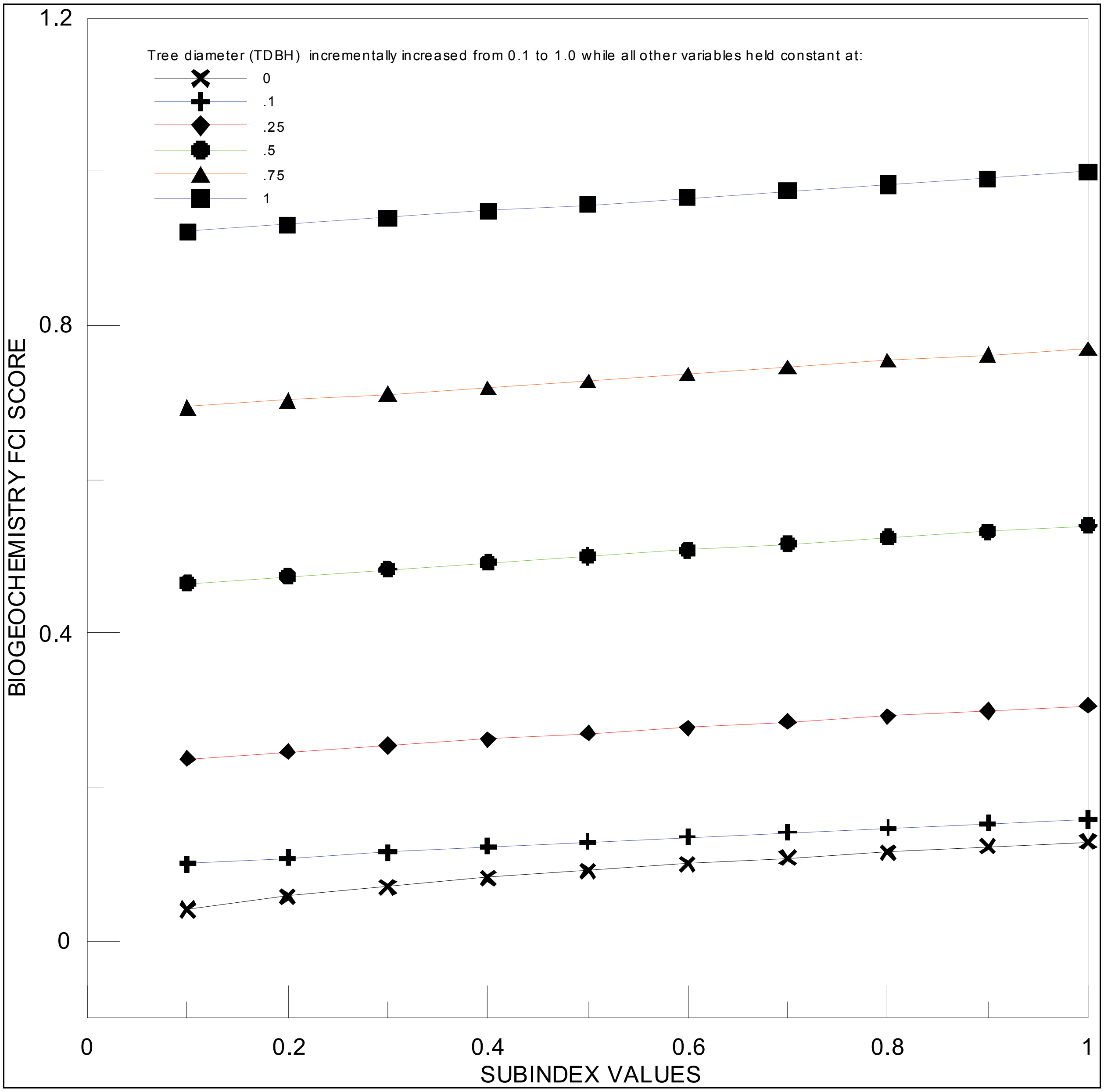
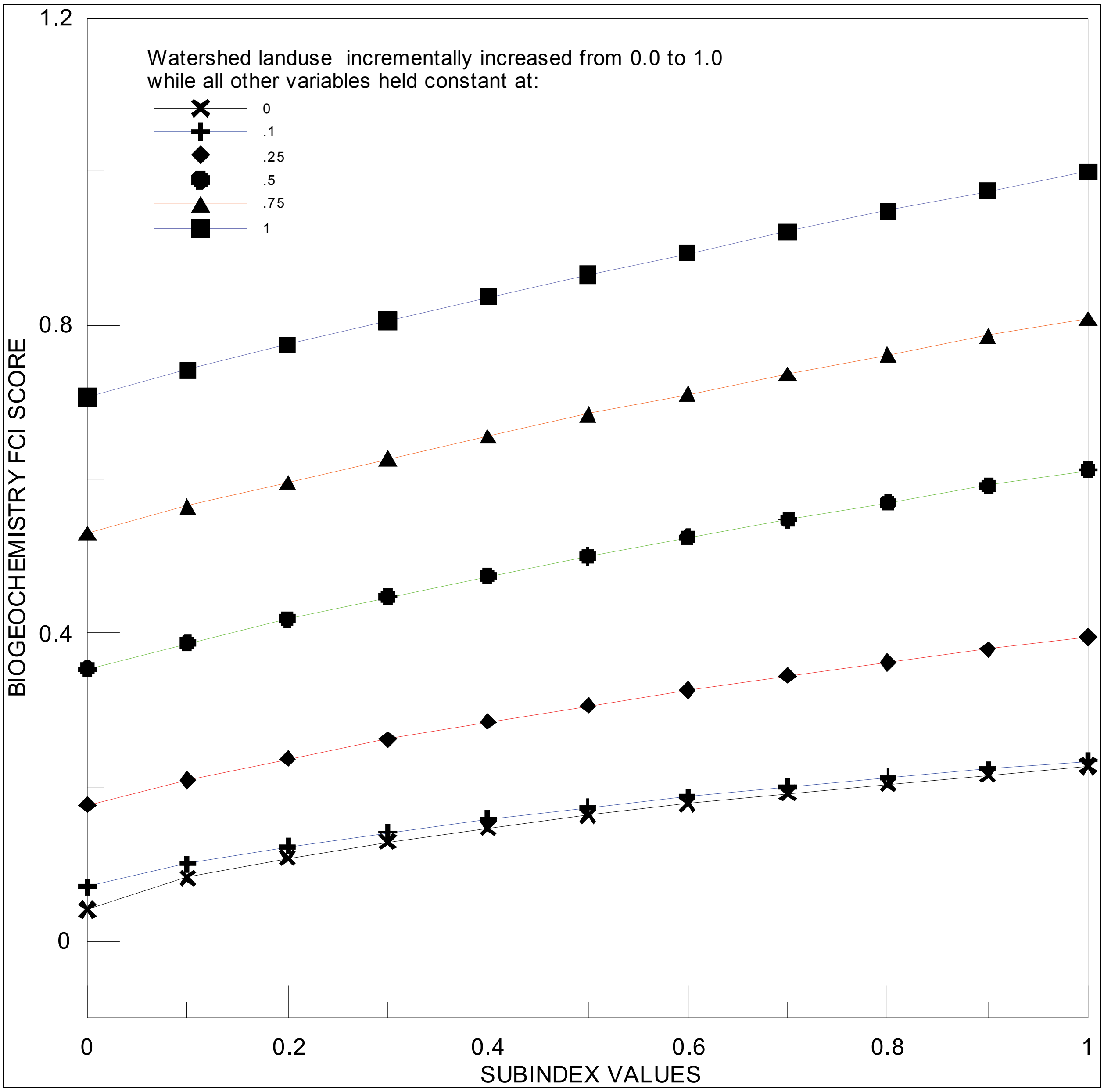


Figure A.2.6. Results of sensitivity test of V_{WLUSE} (canopy cover ≥20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



$$FCI = \left\{ V_{EMBED} \times \left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{SSD} + V_{HERB}}{4} \right) + V_{WLUSE}}{4} \right] \right\}^{1/2} \tag{A.2.2}$$

Table A.2.2. Range of FCI scores attainable based on Biogeochemical Cycling sensitivity analysis (≥20 percent canopy cover). Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.1, 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.00	0.00	0.00	0.15	0.07	0.22	0.24	0.11	0.35
V_{LWD}	0.08	0.00	0.08	0.04	0.07	0.10	0.04	0.17	0.21
$V_{DETRITUS}$	0.08	0.00	0.08	0.04	0.07	0.10	0.04	0.17	0.21
V_{DBH}	0.08	0.00	0.08	0.04	0.07	0.10	0.04	0.17	0.21
V_{WLUSE}	0.16	0.00	0.16	0.12	0.05	0.17	0.15	0.13	0.28
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{EMBED}	0.34	0.16	0.50	0.42	0.19	0.61	0.48	0.22	0.71
V_{LWD}	0.04	0.33	0.38	0.05	0.50	0.54	0.05	0.66	0.71
$V_{DETRITUS}$	0.04	0.33	0.38	0.05	0.50	0.54	0.05	0.66	0.71
V_{DBH}	0.04	0.33	0.38	0.05	0.50	0.54	0.05	0.66	0.71
V_{WLUSE}	0.18	0.25	0.43	0.20	0.38	0.57	0.21	0.50	0.71

Figure A.2.7. Relative distribution of variable influence on Biogeochemical Cycling FCI scores in sites displaying <20 percent canopy cover.

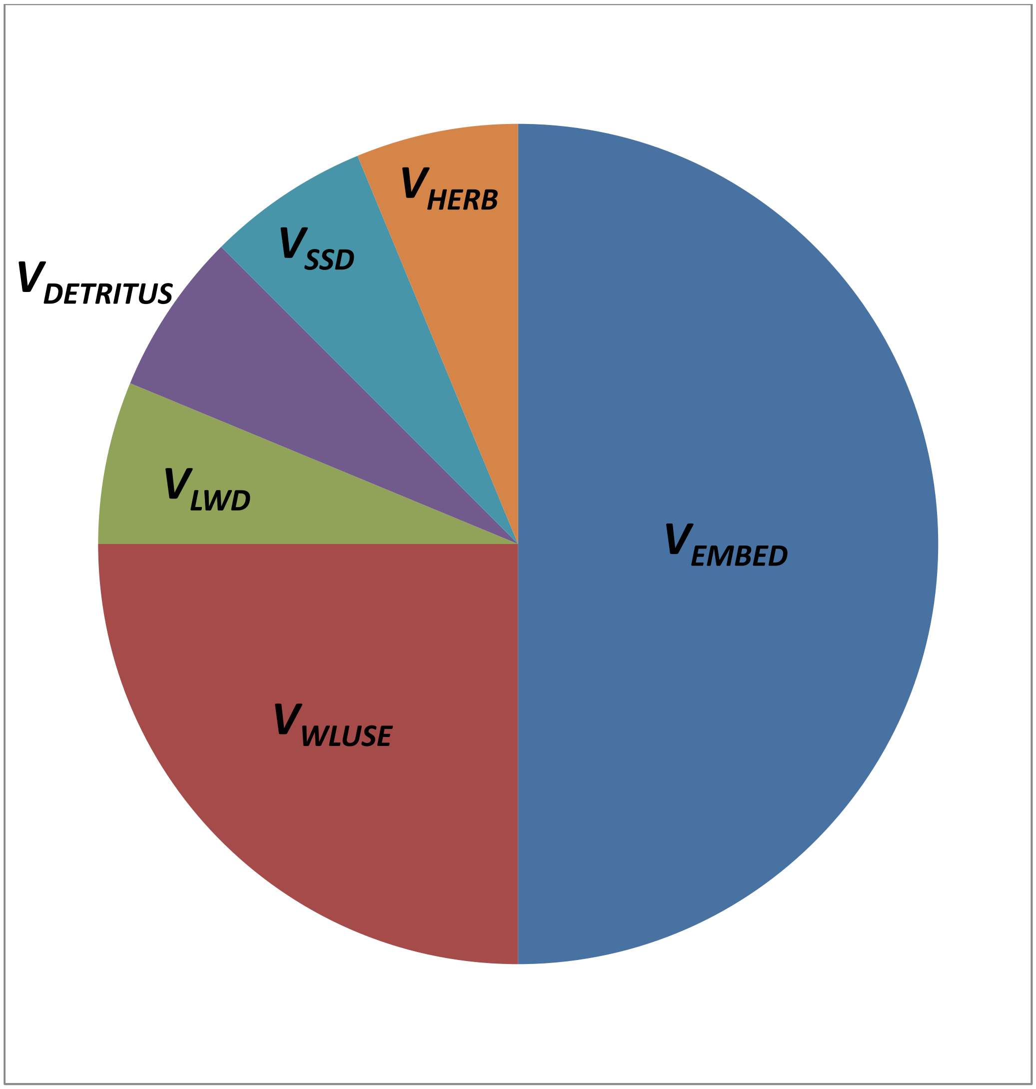


Figure A.2.8. Results of sensitivity test of V_{EMBED} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

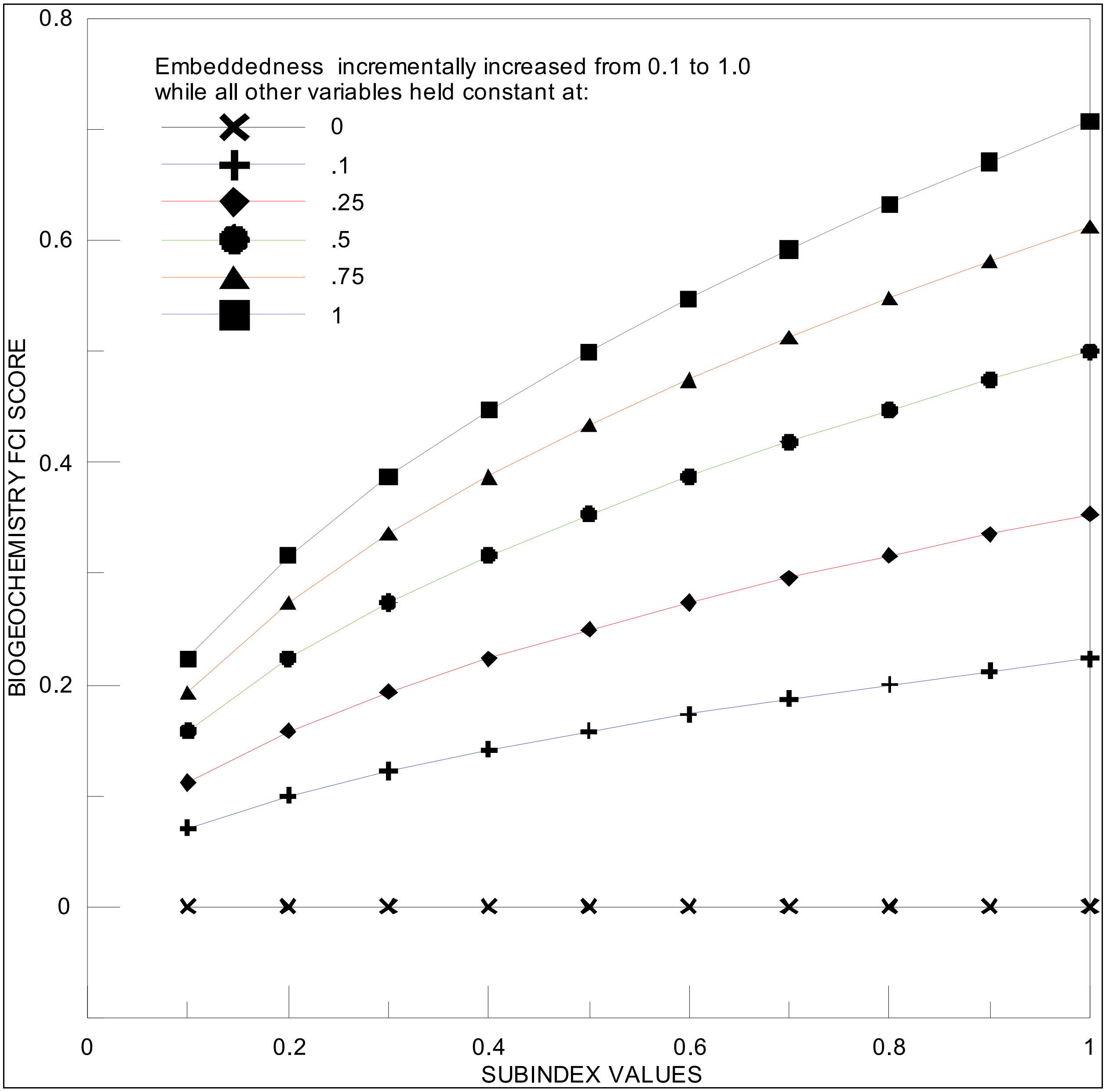


Figure A.2.9. Results of sensitivity test of V_{LWD} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

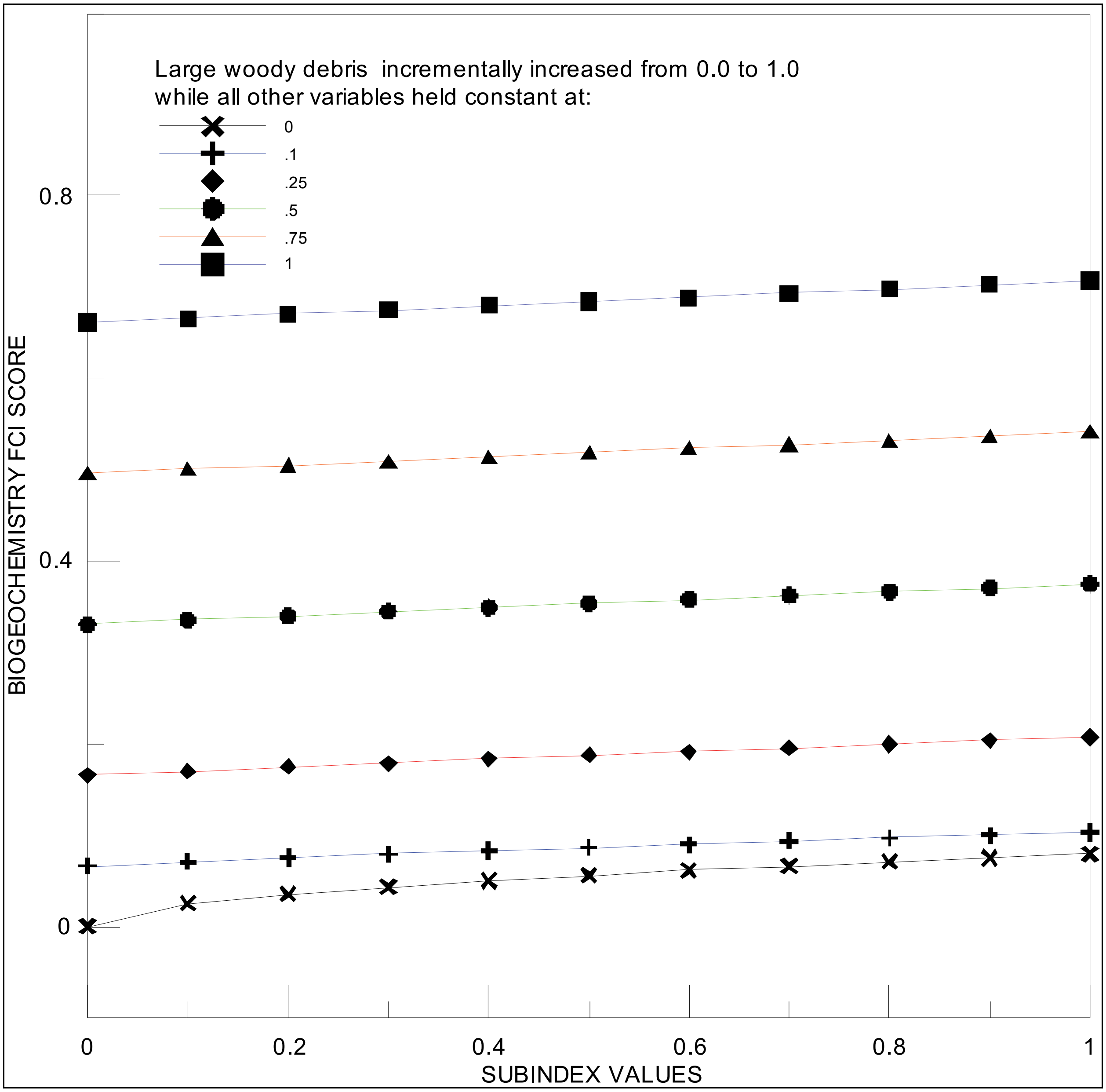


Figure A.2.10. Results of sensitivity test of $V_{DETRITUS}$ (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

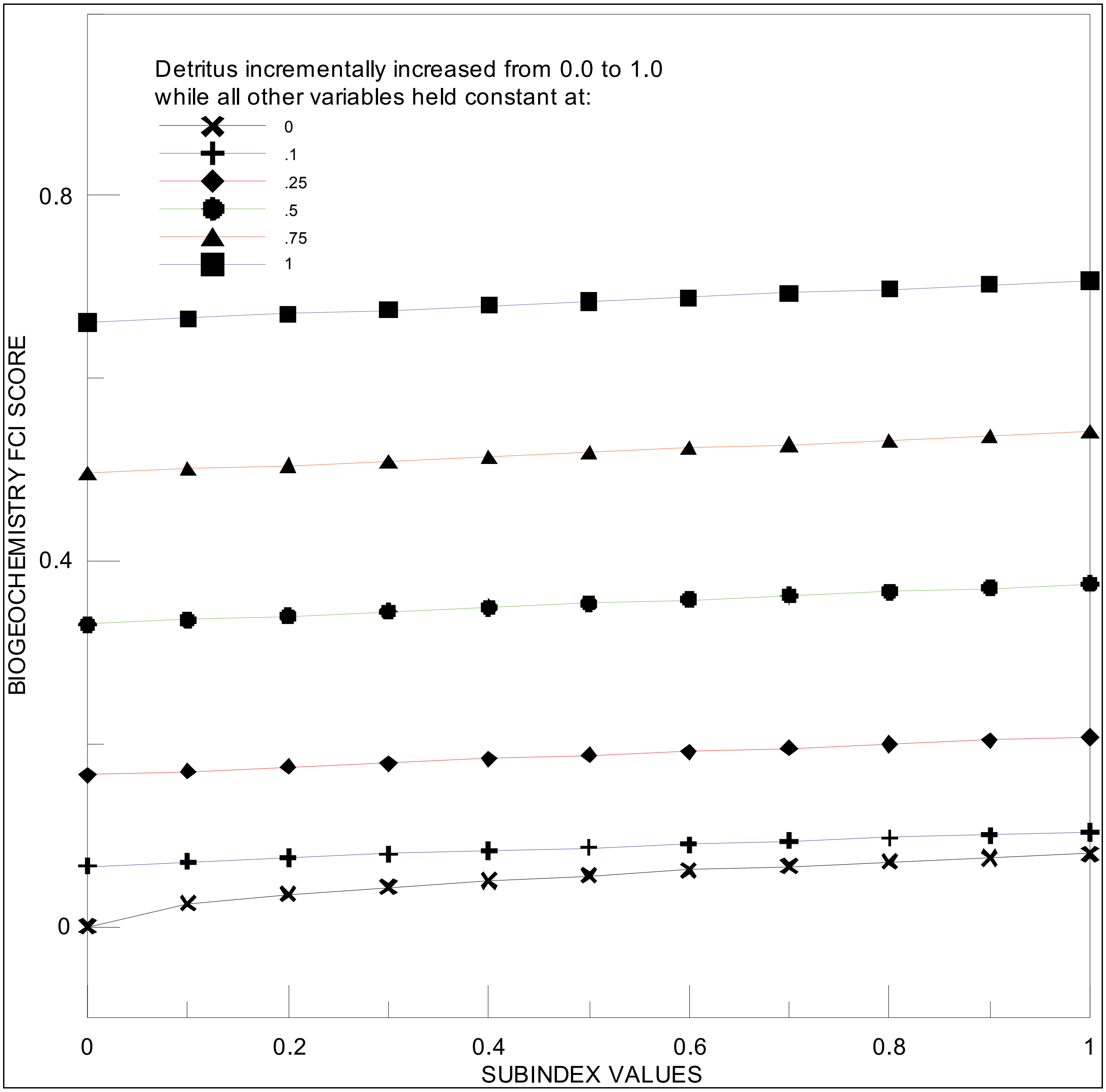


Figure A.2.11. Results of sensitivity test of V_{SSD} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

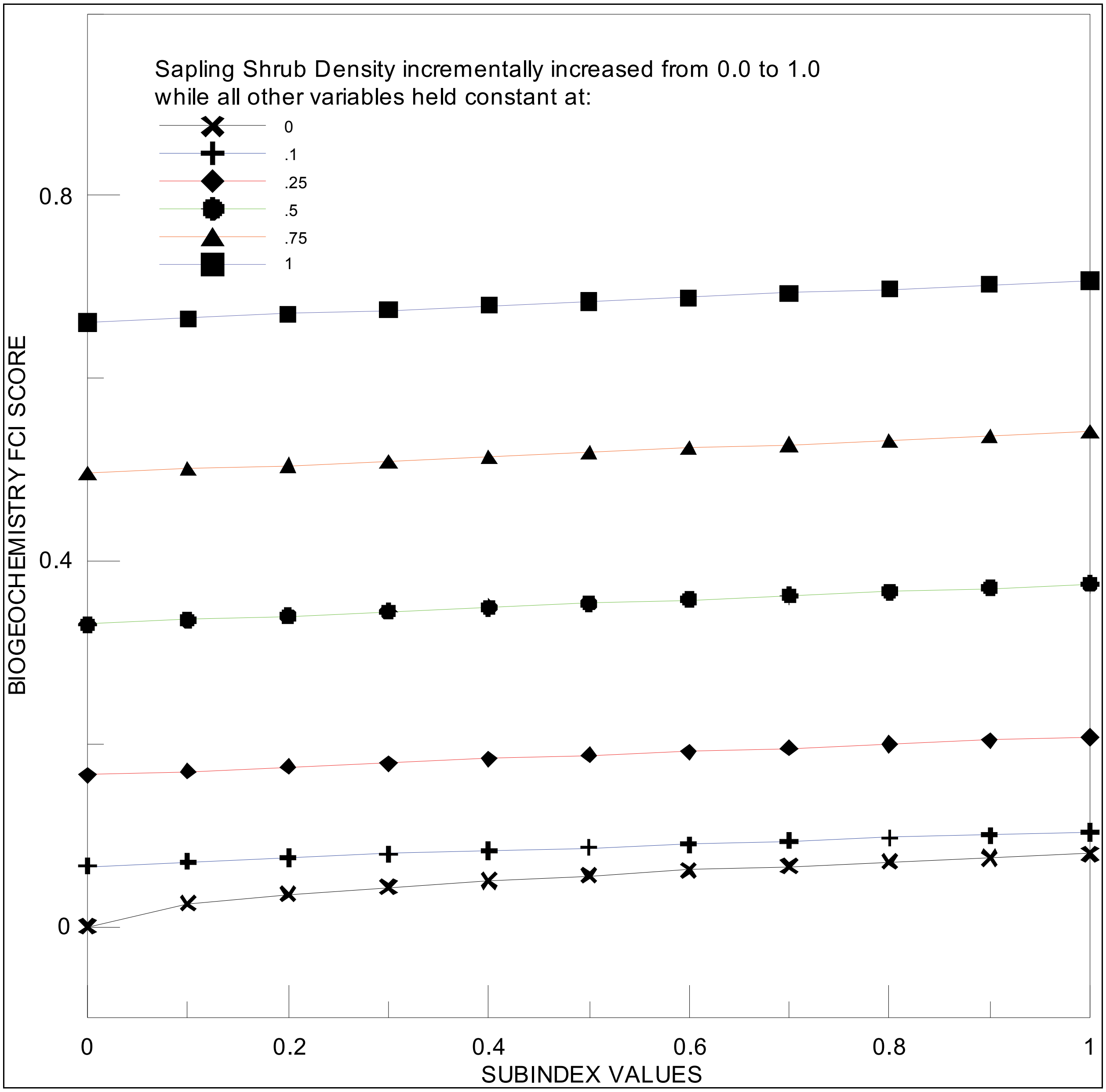


Figure A.2.12. Results of sensitivity test of V_{HERB} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

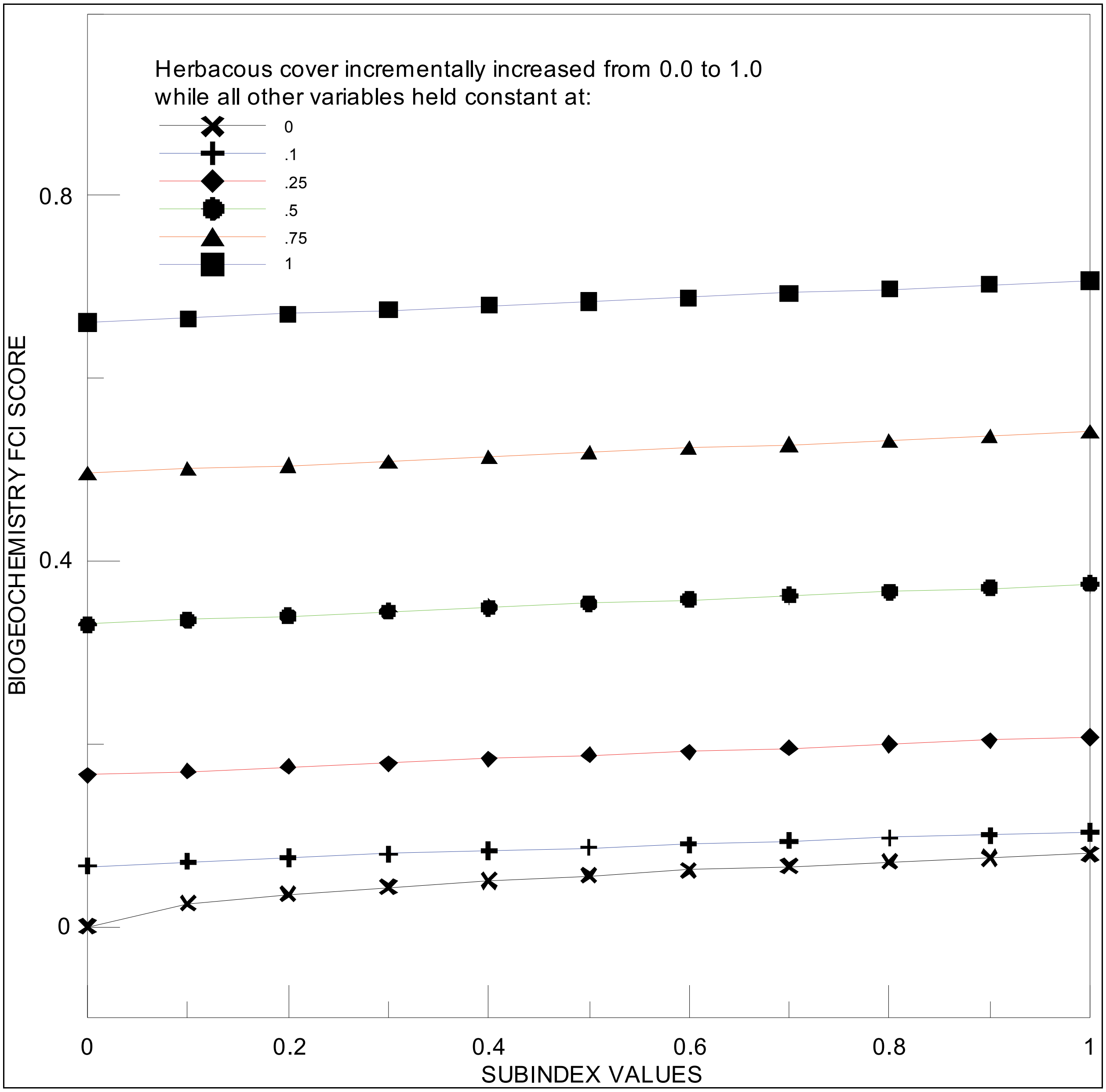
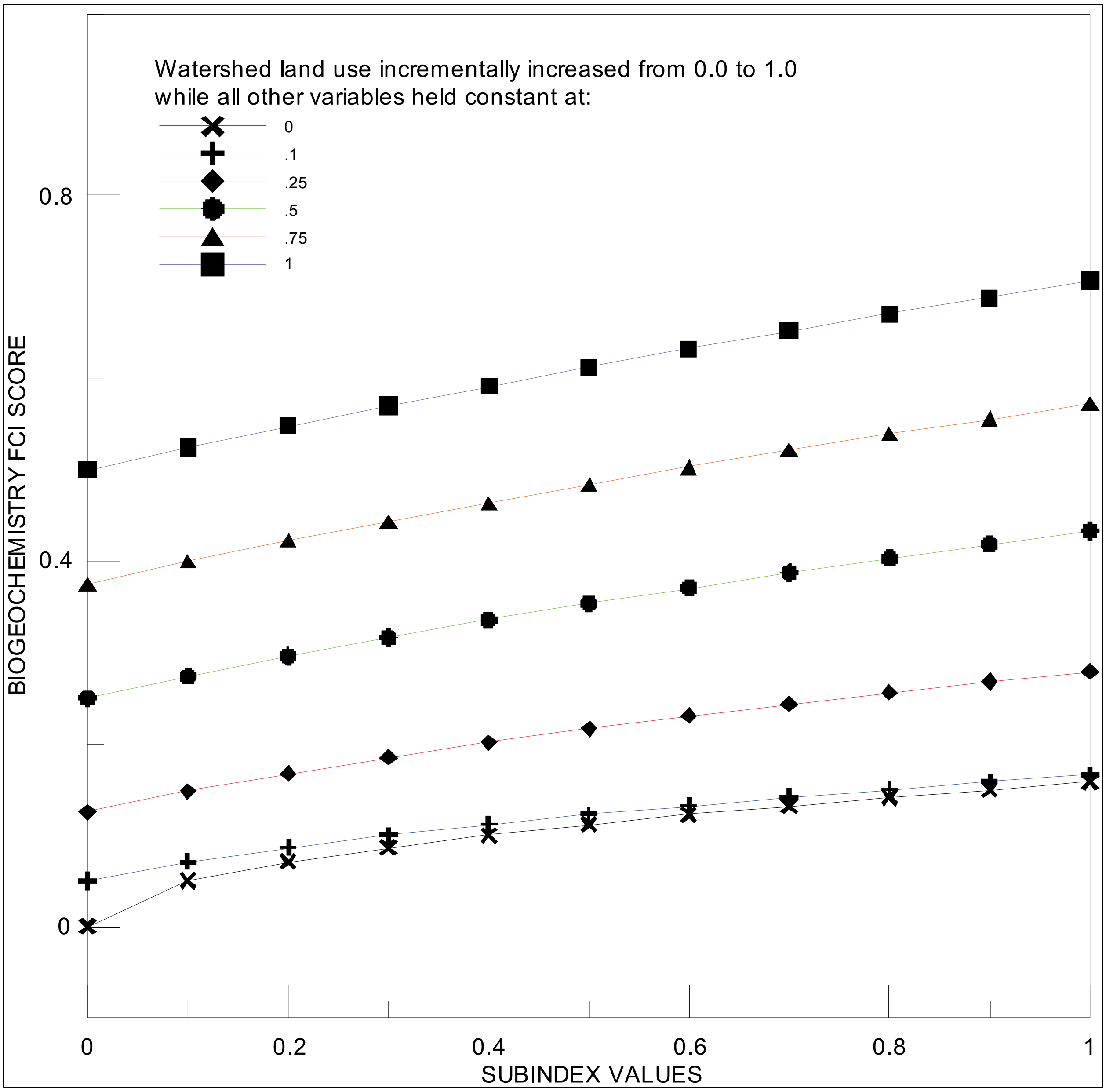


Figure A.2.13. Results of sensitivity test of V_{WLUSE} (canopy cover <20 percent). Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



A.3 Hydrology function sensitivity testing results

Table A.3.1 Provides sensitivity analysis results, and Figures A.3.1 through A.3.6 provide graphical representations of sensitivity analysis results for the Hydrology model (Equation A.3.1). A detailed discussion of sensitivity analysis results is provided in section 5.6.

$$FCI = \left\{ \frac{V_{WLUSE} + \left[\frac{V_{LWD} + \min\left(V_{SUBSTRATE}, V_{EMBED}, V_{BERO}\right)}{2} \right]}{2} \right\} \quad \text{A.3.1}$$

Table A.3.1. Range of FCI scores attainable based on Hydrology sensitivity analysis. Each variable is increased from 0.0 to 1.0 in increments of 0.1, while all other variables are held at the values presented below (minimum, 0.1, 0.25, 0.5, 0.75, and 1.0).

Variable	Subindex values								
	Minimum			0.10			0.25		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{WLUSE}	0.5	0	0.5	0.5	0.05	0.55	0.5	0.125	0.625
V_{EMBED}	0	0	0	0	0.1	0.1	0.0375	0.2125	0.25
V_{LWD}	0.25	0	0.25	0.25	0.075	0.325	0.25	0.1875	0.4375
$V_{SUBSTRATE}$	0	0	0	0.025	0.075	0.1	0.0625	0.1875	0.25
V_{BERO}	0	0	0	0.025	0.075	0.1	0.0625	0.1875	0.25
Variable	Subindex values								
	0.50			0.75			1.00		
	Range	Low	High	Range	Low	High	Range	Low	High
V_{WLUSE}	0.5	0.25	0.75	0.5	0.375	0.875	0.5	0.5	1
V_{EMBED}	0.1	0.4	0.5	0.1625	0.5875	0.75	0.225	0.775	1
V_{LWD}	0.25	0.375	0.625	0.25	0.5625	0.8125	0.25	0.75	1
$V_{SUBSTRATE}$	0.125	0.375	0.5	0.1875	0.5625	0.75	0.25	0.75	1
V_{BERO}	0.125	0.375	0.5	0.1875	0.5625	0.75	0.25	0.75	1

Figure A.3.1. Relative distribution of variable influence on Hydrology FCI scores.

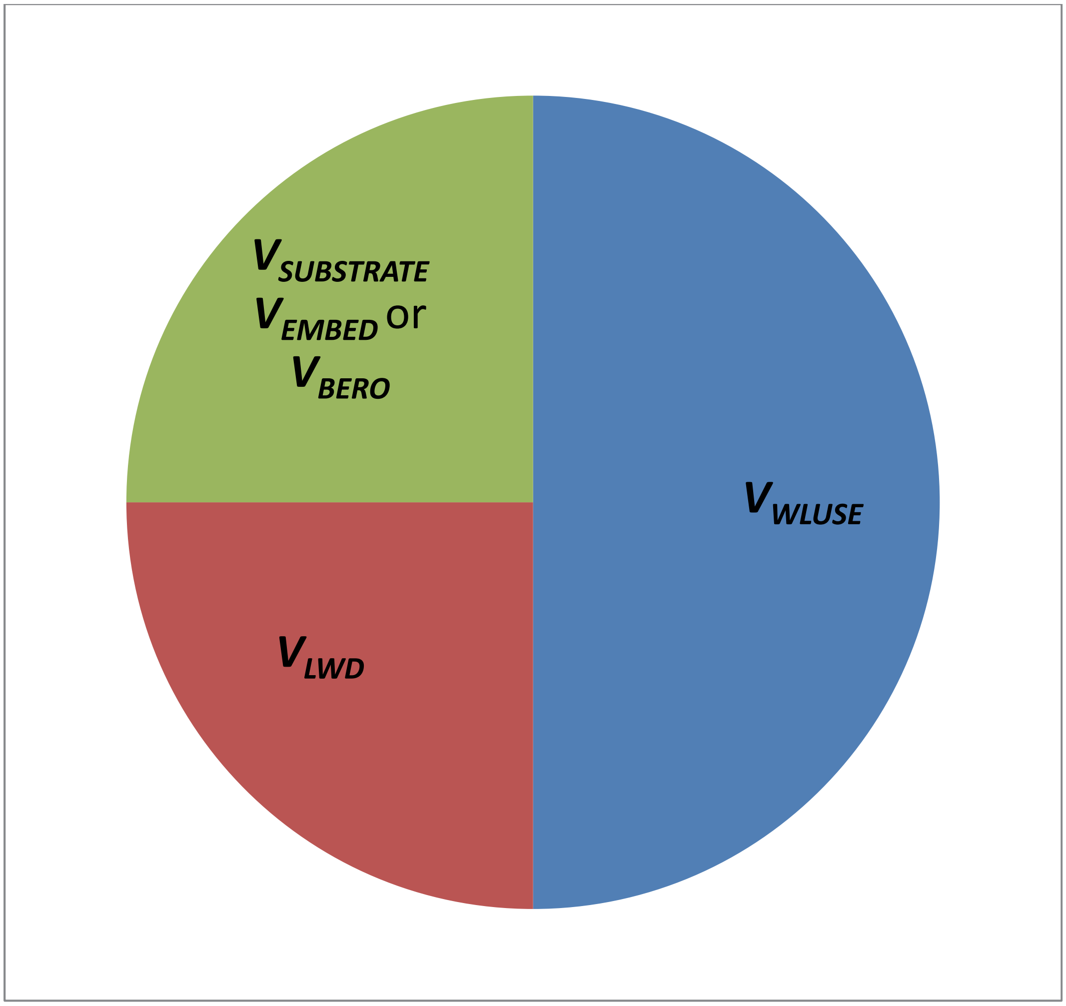


Figure A.3.2. Results of sensitivity test of V_{WLUSE}. Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

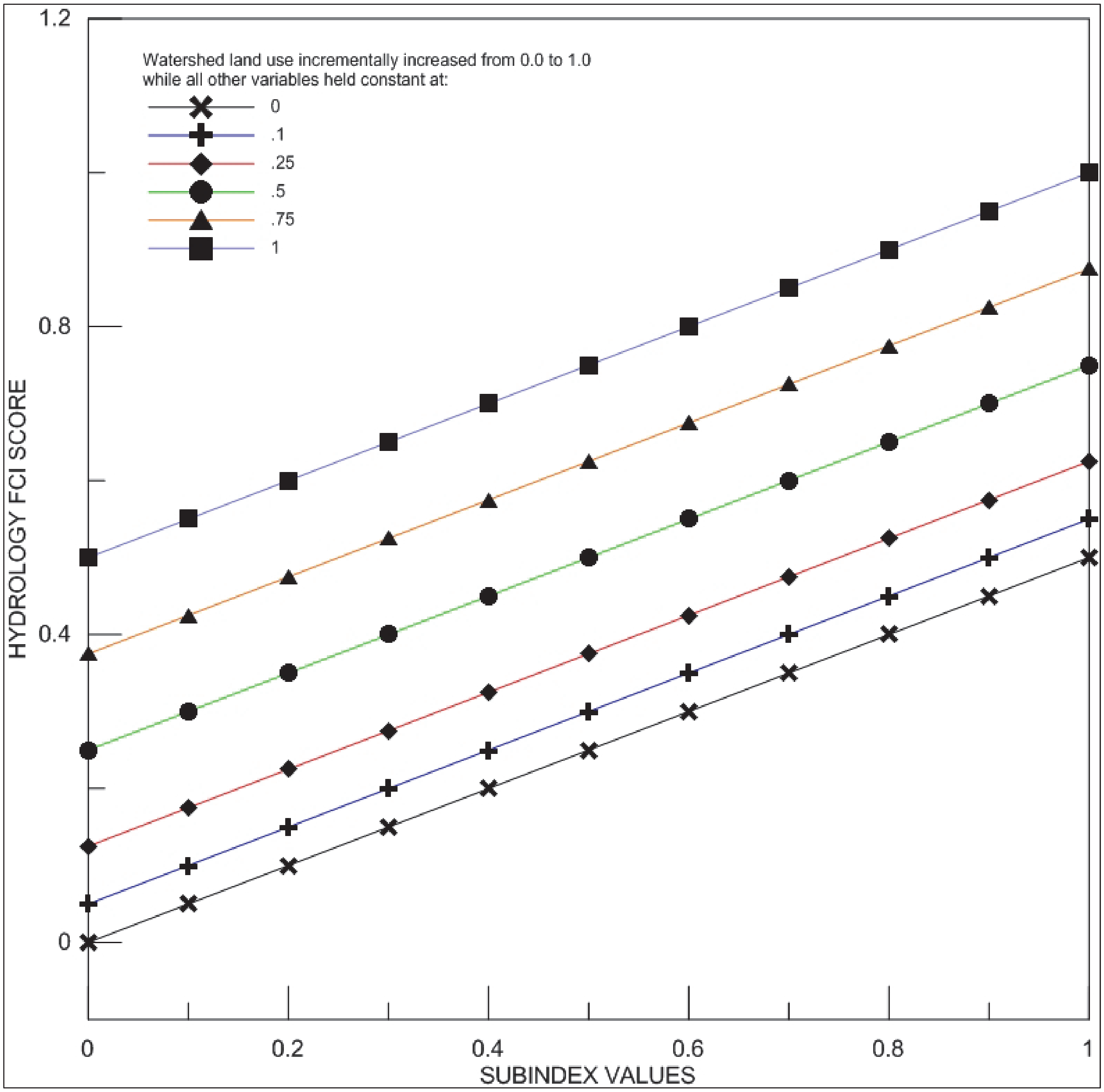


Figure A.3.3. Results of sensitivity test of V_{EMBED}. Each line in the figure represents a test scenario in which the target variable was increased from 0.1 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

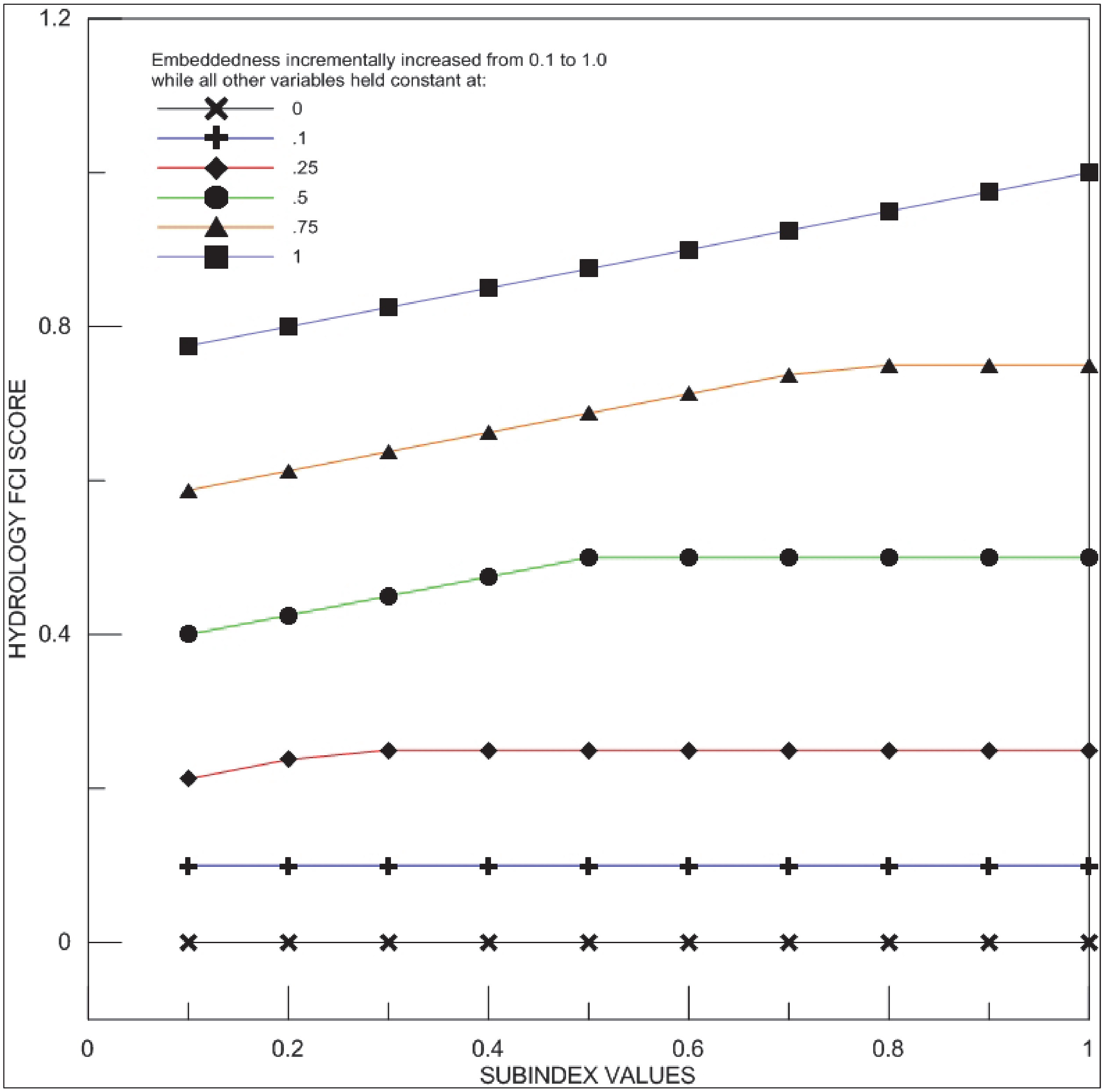


Figure A.3.4. Results of sensitivity test of V_{LWD}. Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.

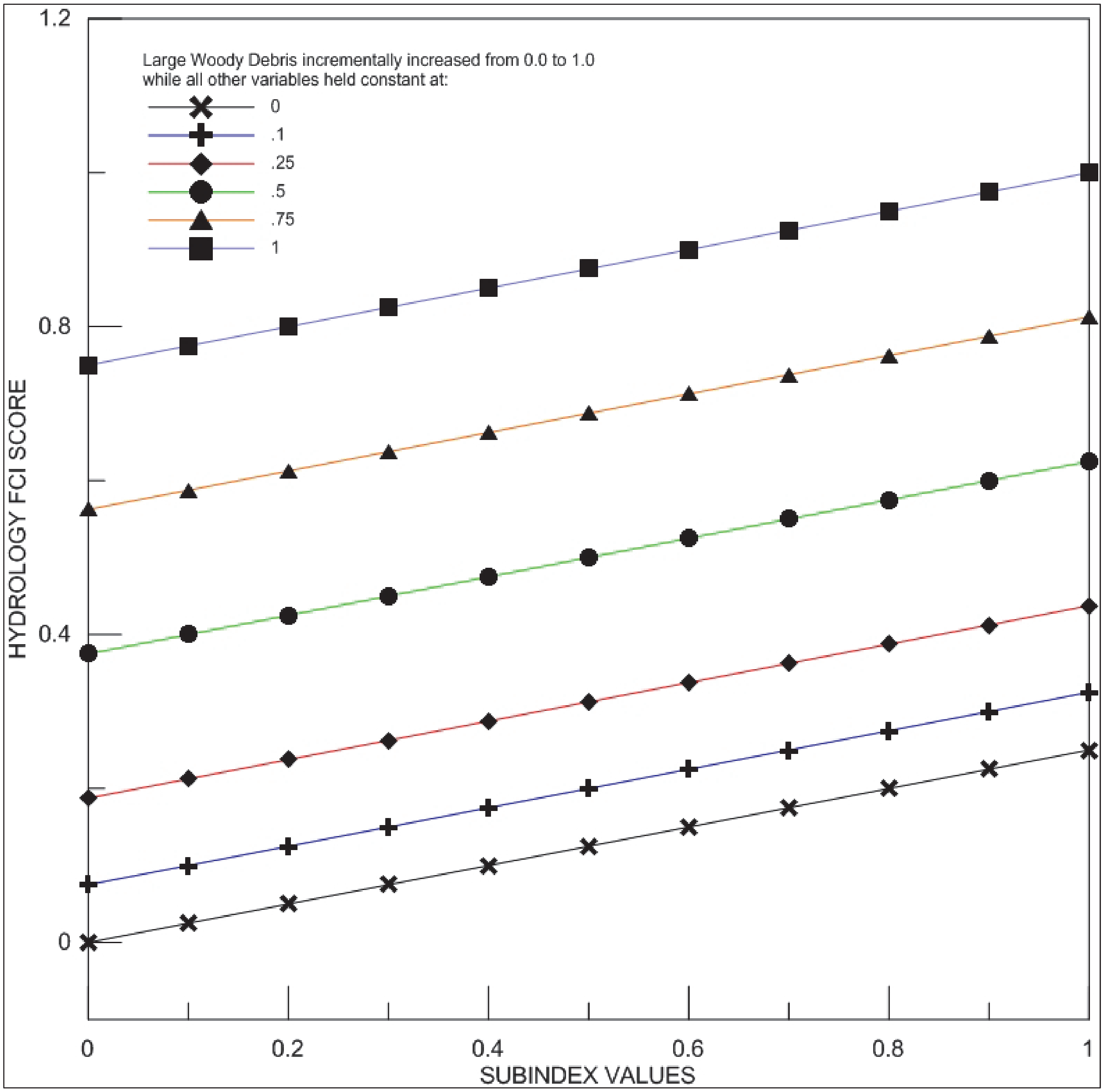


Figure A.3.5. Results of sensitivity test of $V_{SUBSTRATE}$. Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0

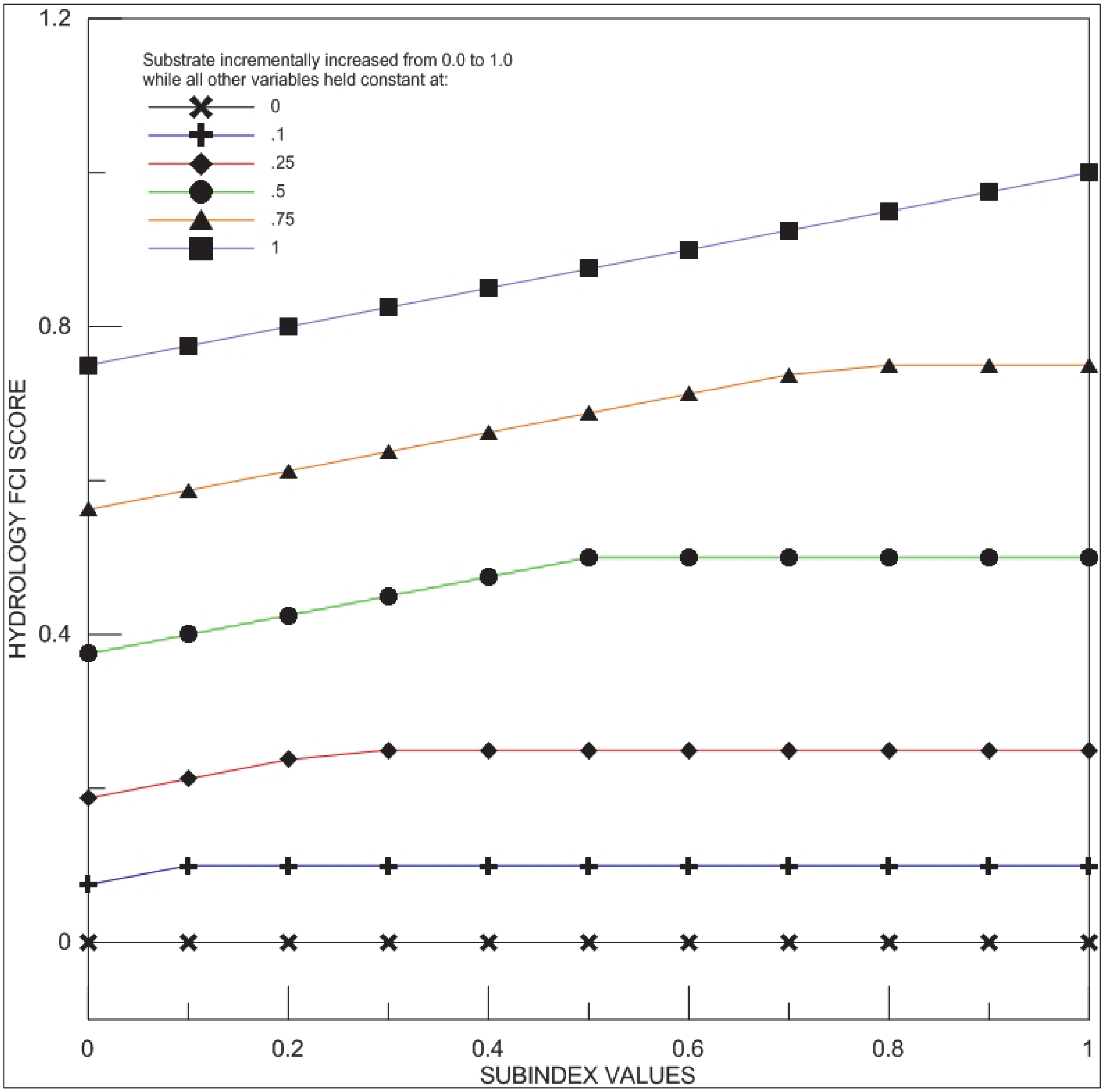
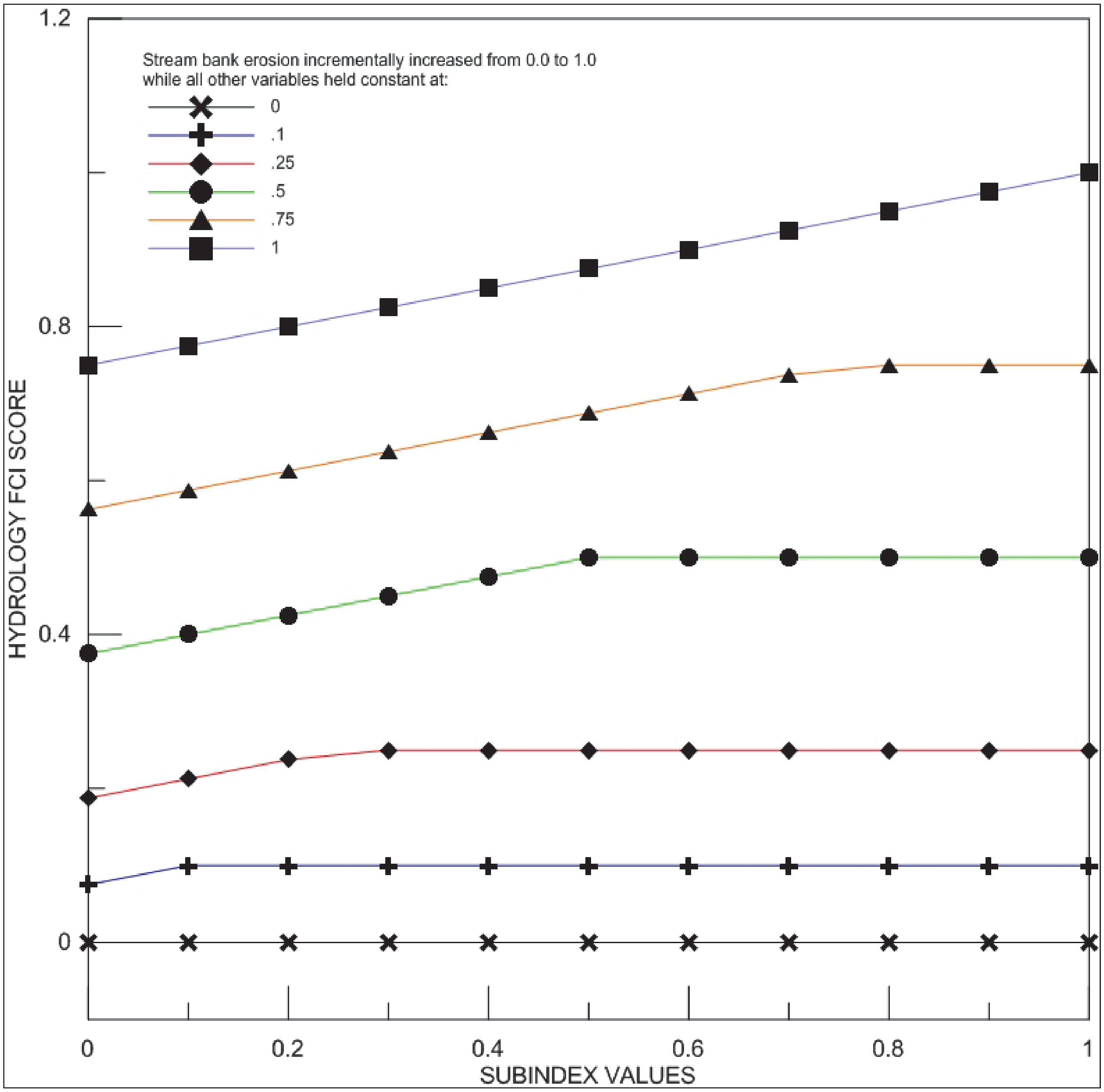


Figure A.3.6. Results of sensitivity test of V_{BERO}. Each line in the figure represents a test scenario in which the target variable was increased from 0.0 to 1.0 in increments of 0.1, while holding all other model variables stable at levels of 0.0, 0.1, 0.25, 0.5, 0.75, and 1.0.



Appendix B: Climatic data

Figure B.1. Monthly rainfall site 1. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

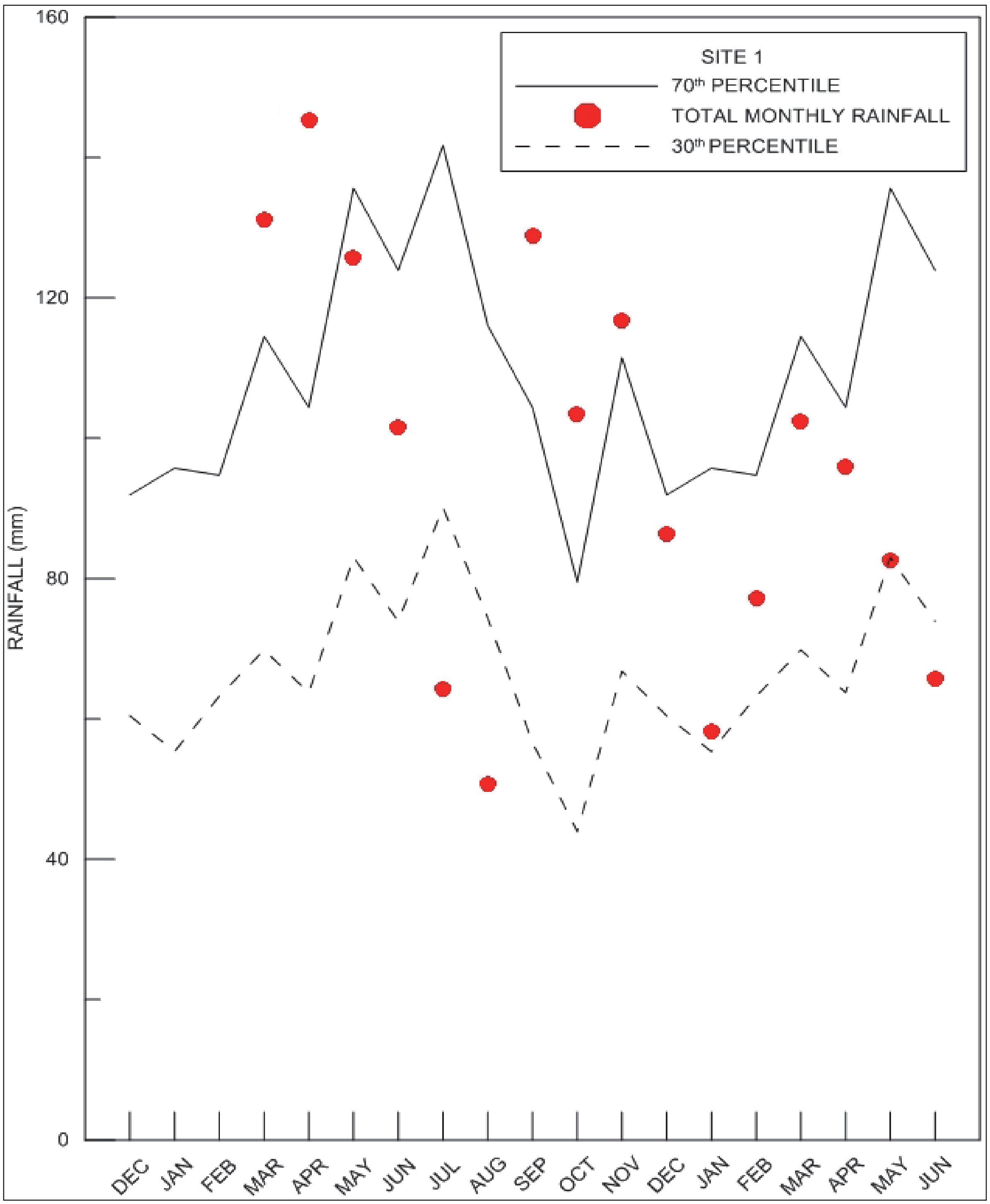


Figure B.2. Monthly rainfall site 2. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

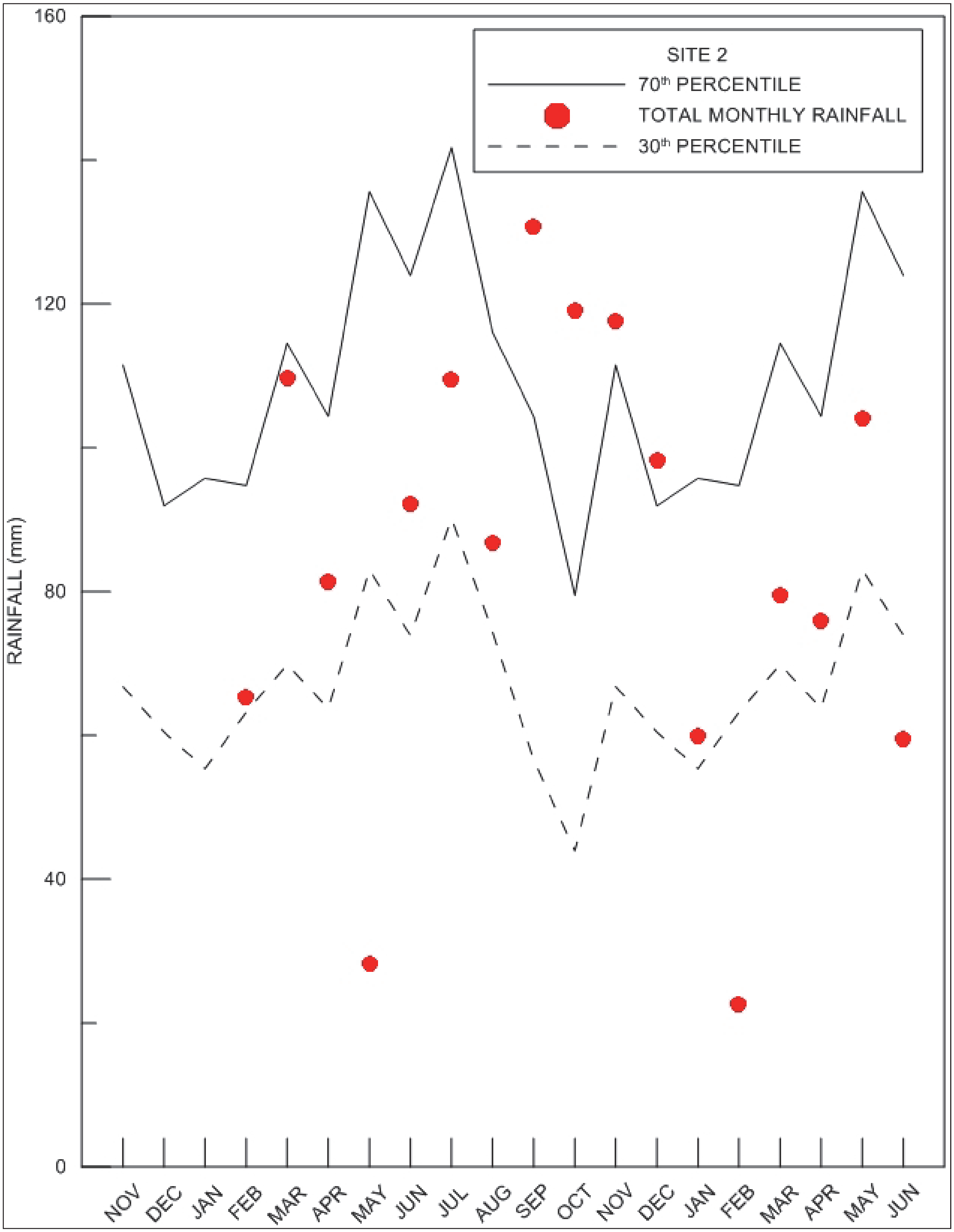


Figure B.3. Monthly rainfall site 3. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

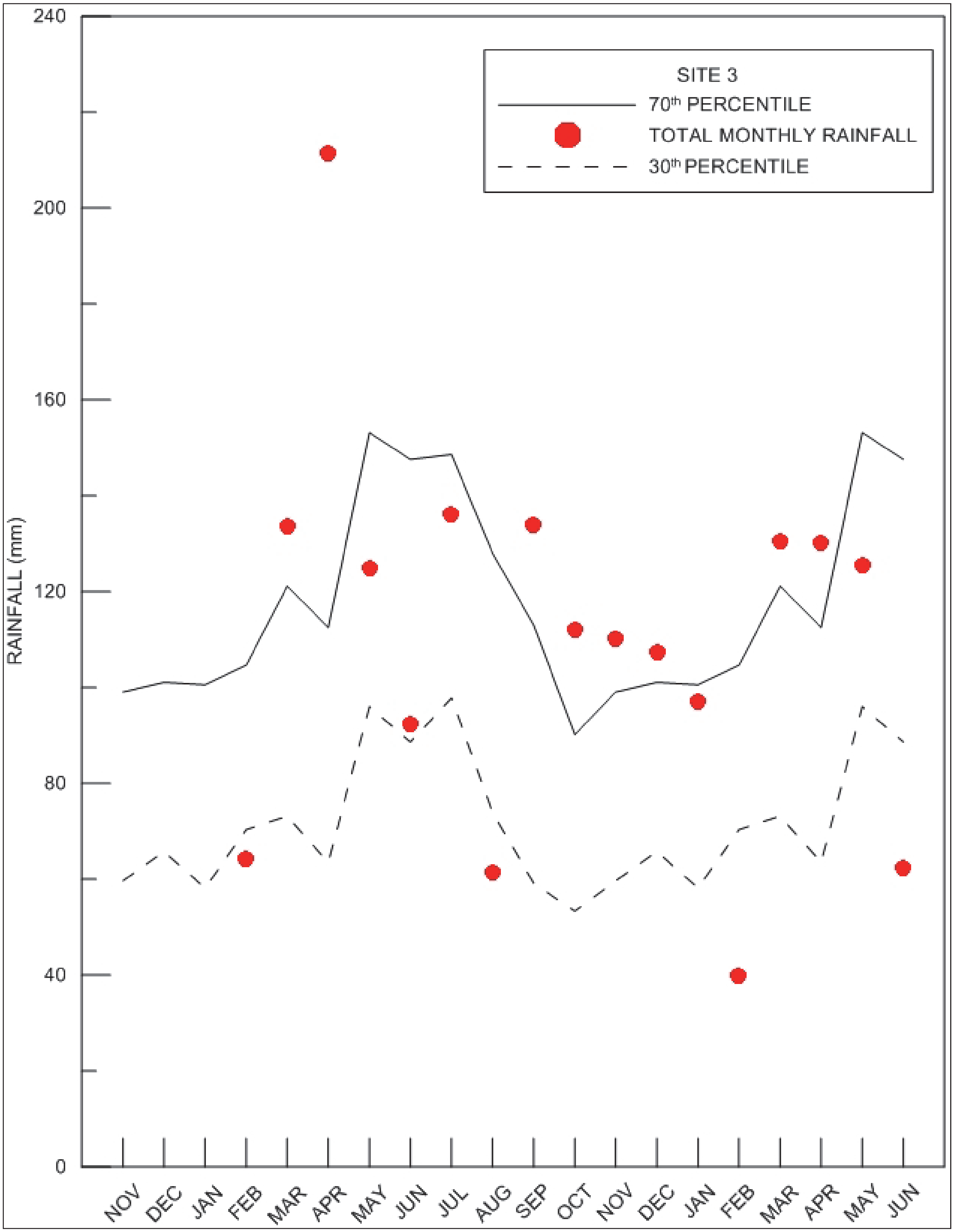


Figure B.4. Monthly rainfall site 4. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

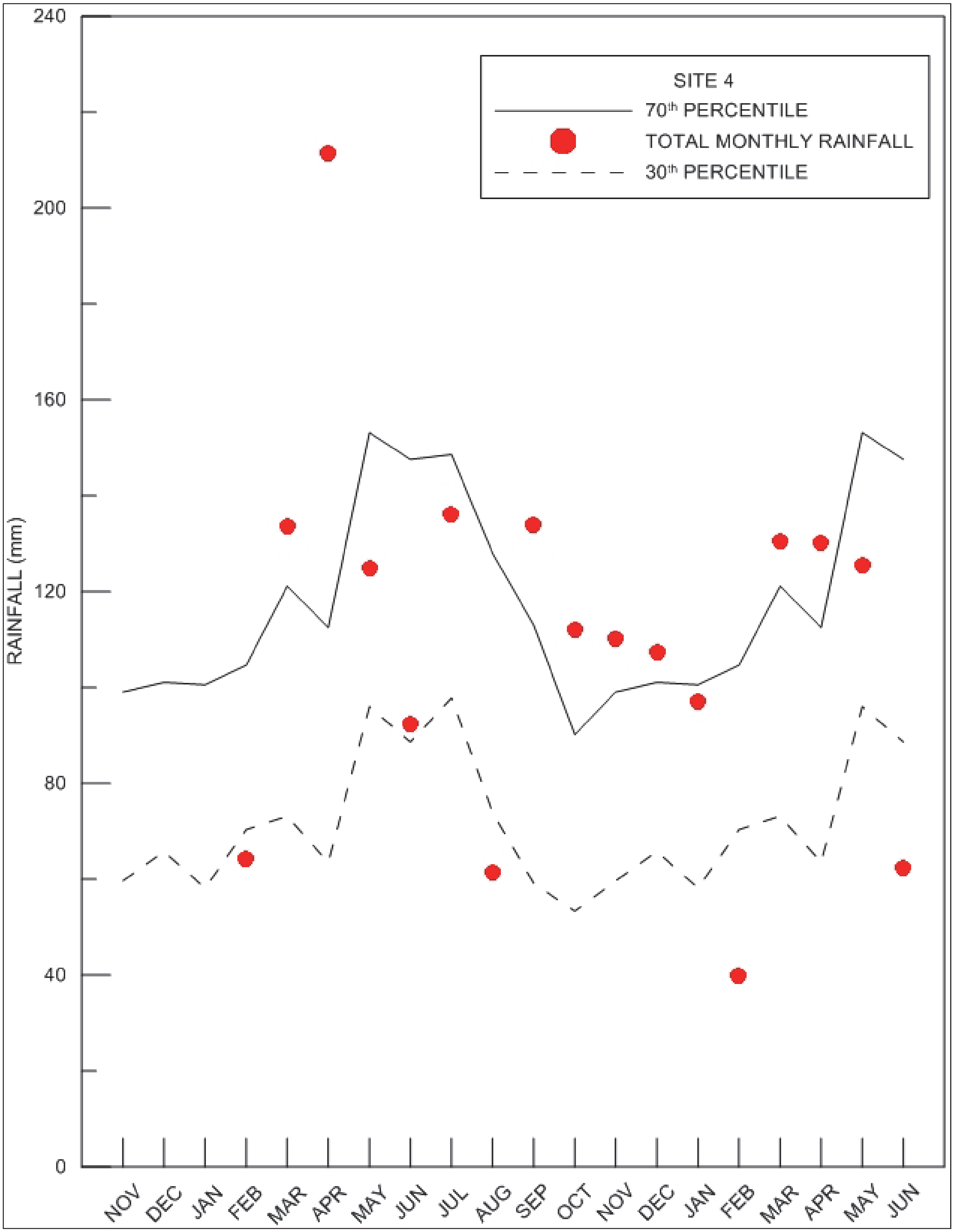


Figure B.5. Monthly rainfall site 5. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

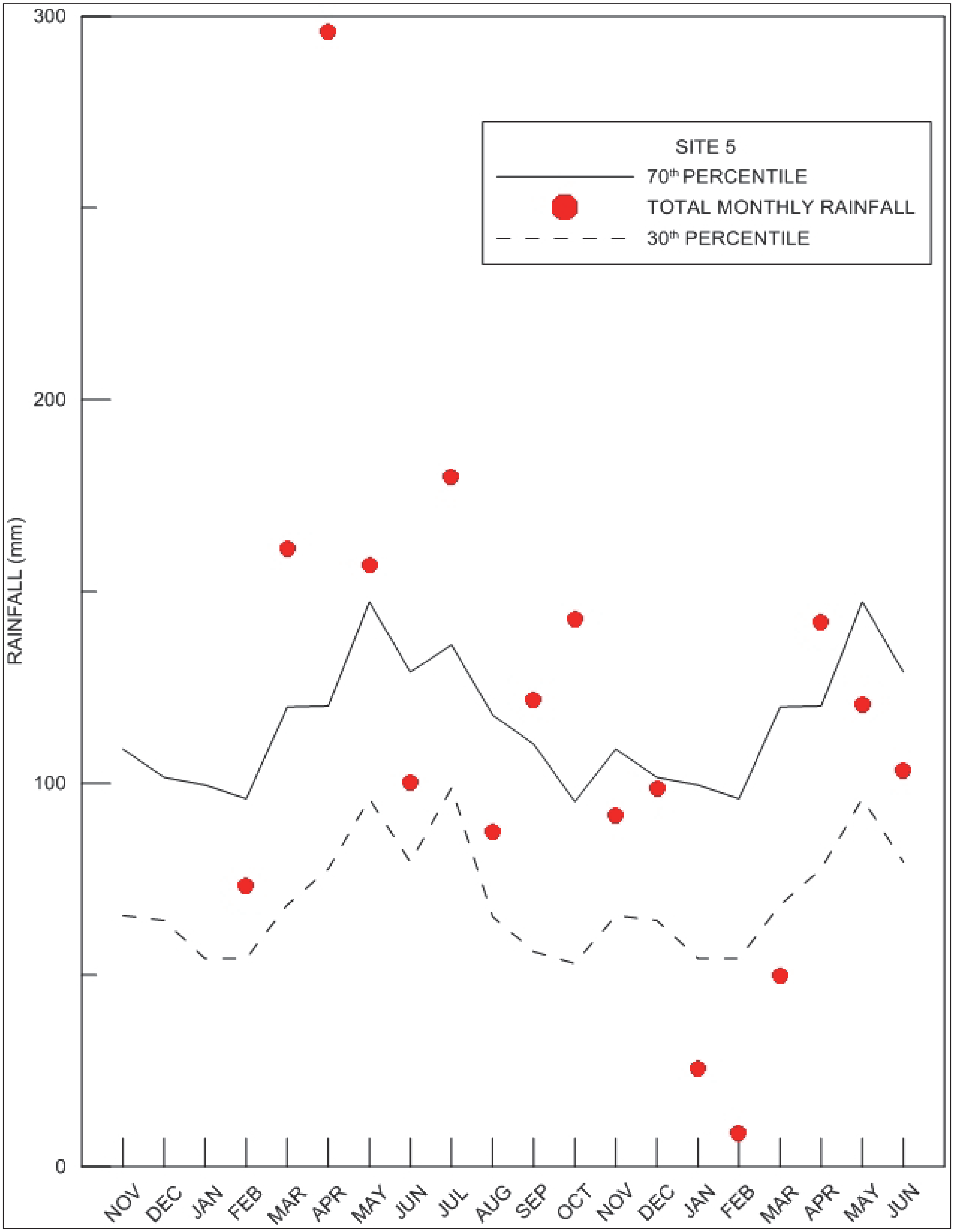


Figure B.6. Monthly rainfall site 6. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

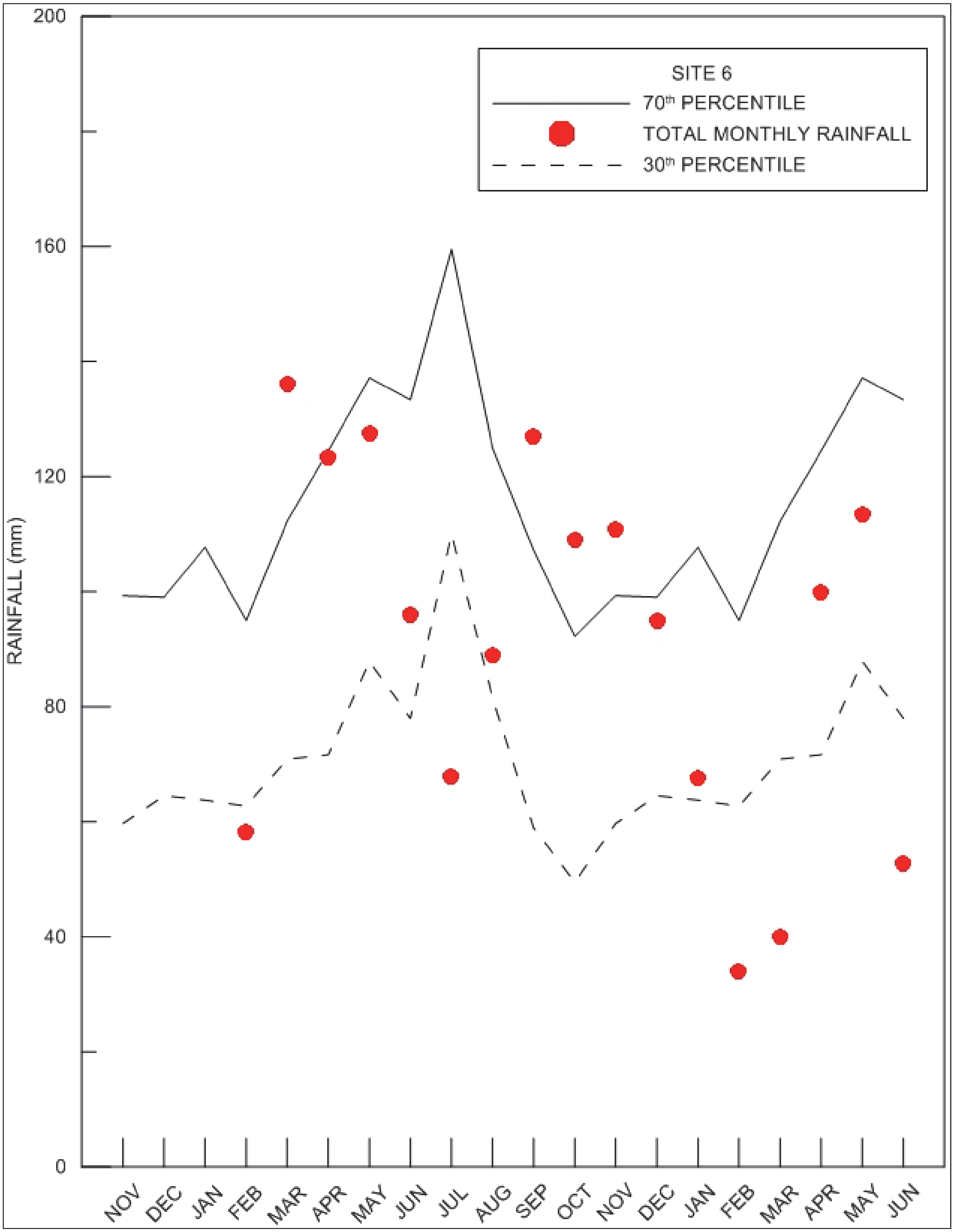


Figure B.7. Monthly rainfall site 7. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

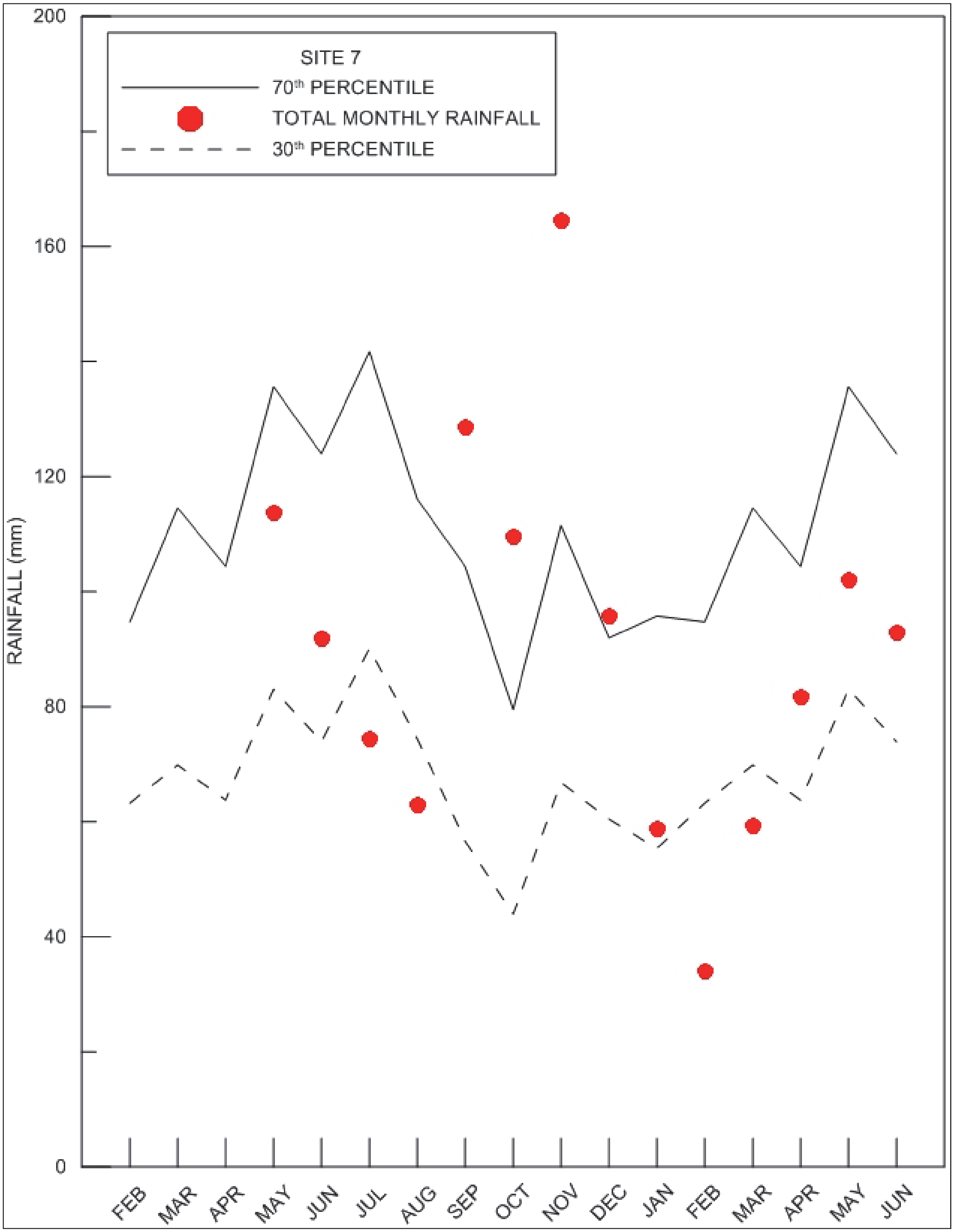


Figure B.8. Monthly rainfall site 8. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

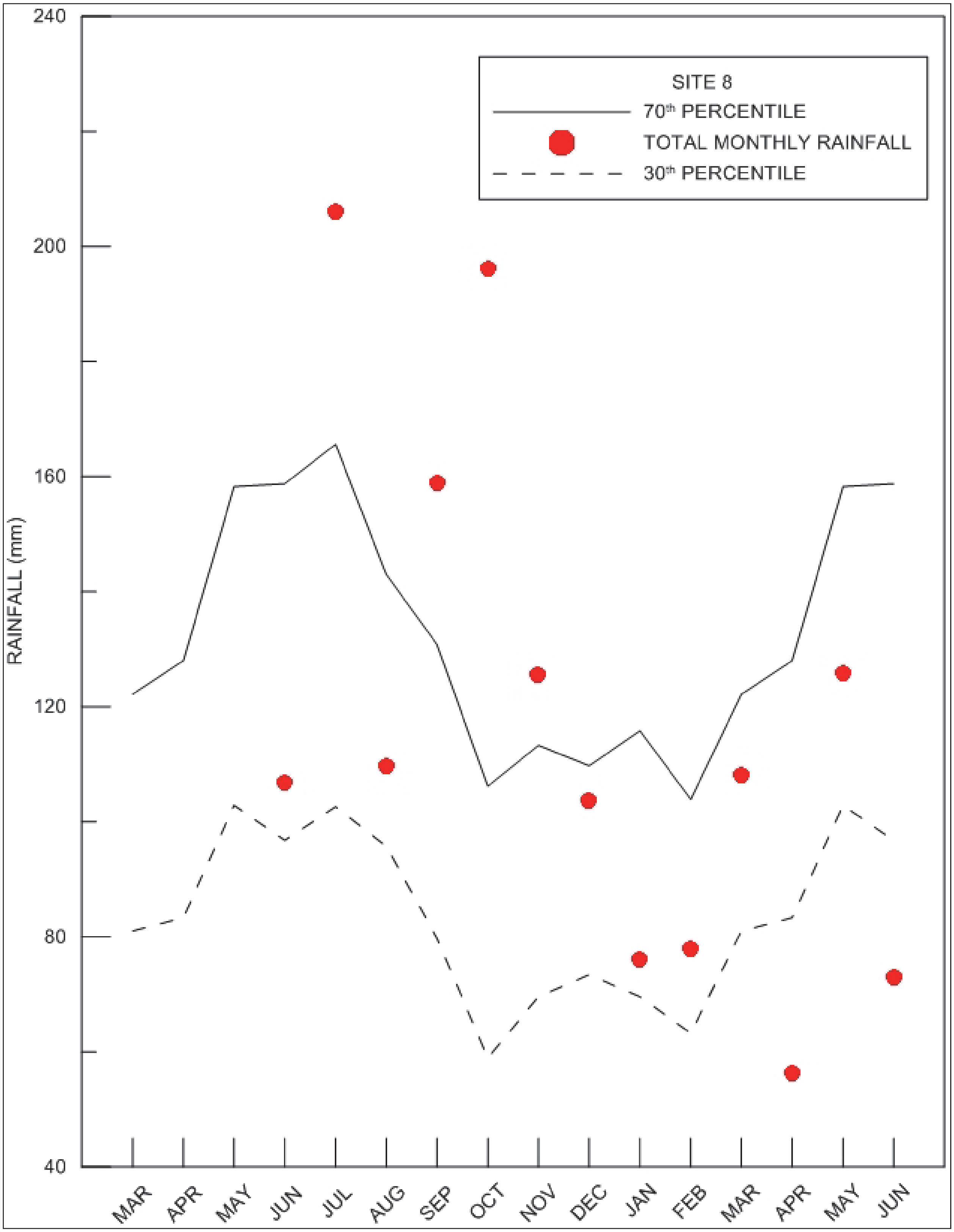


Figure B.9. Monthly rainfall site 9. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.

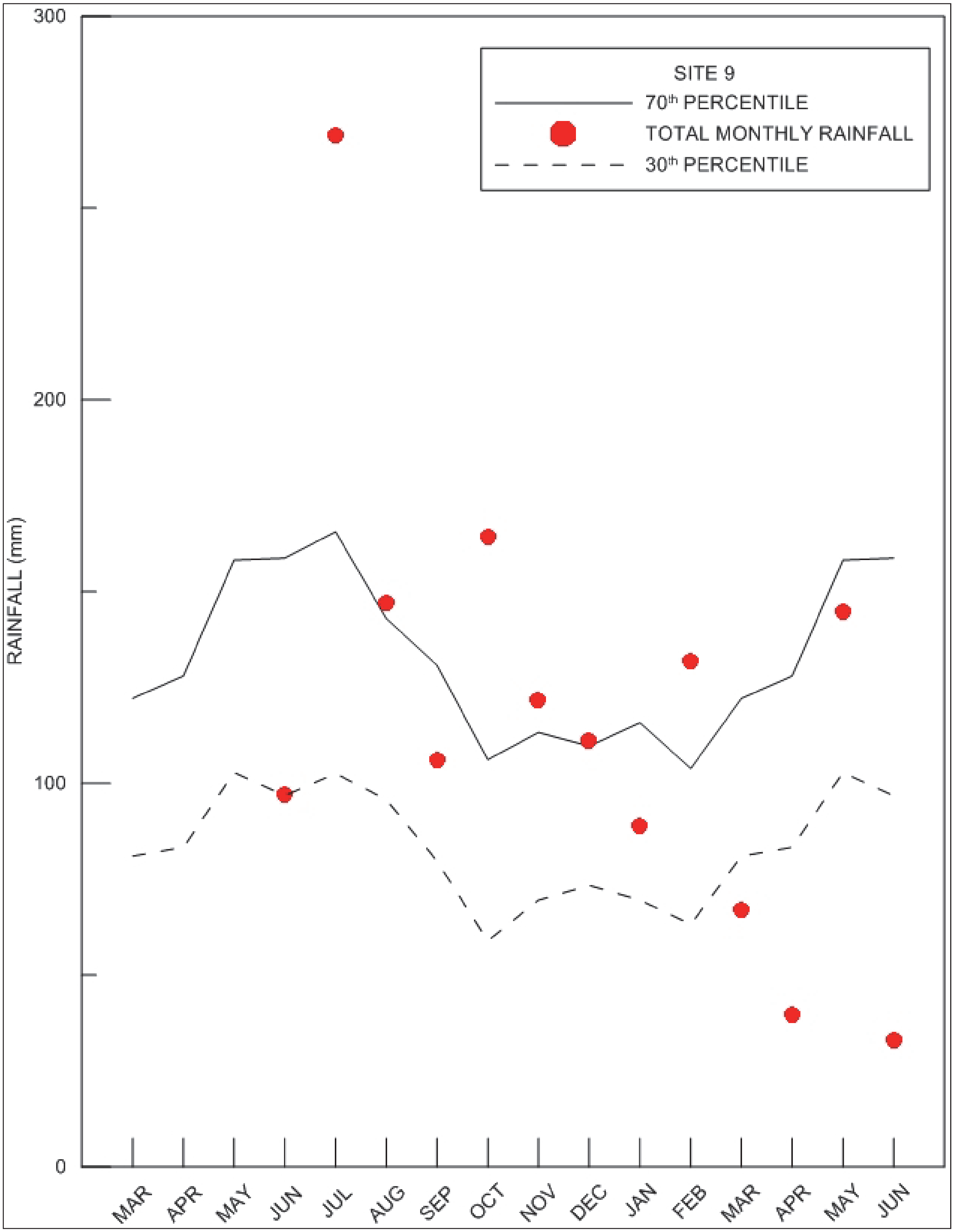
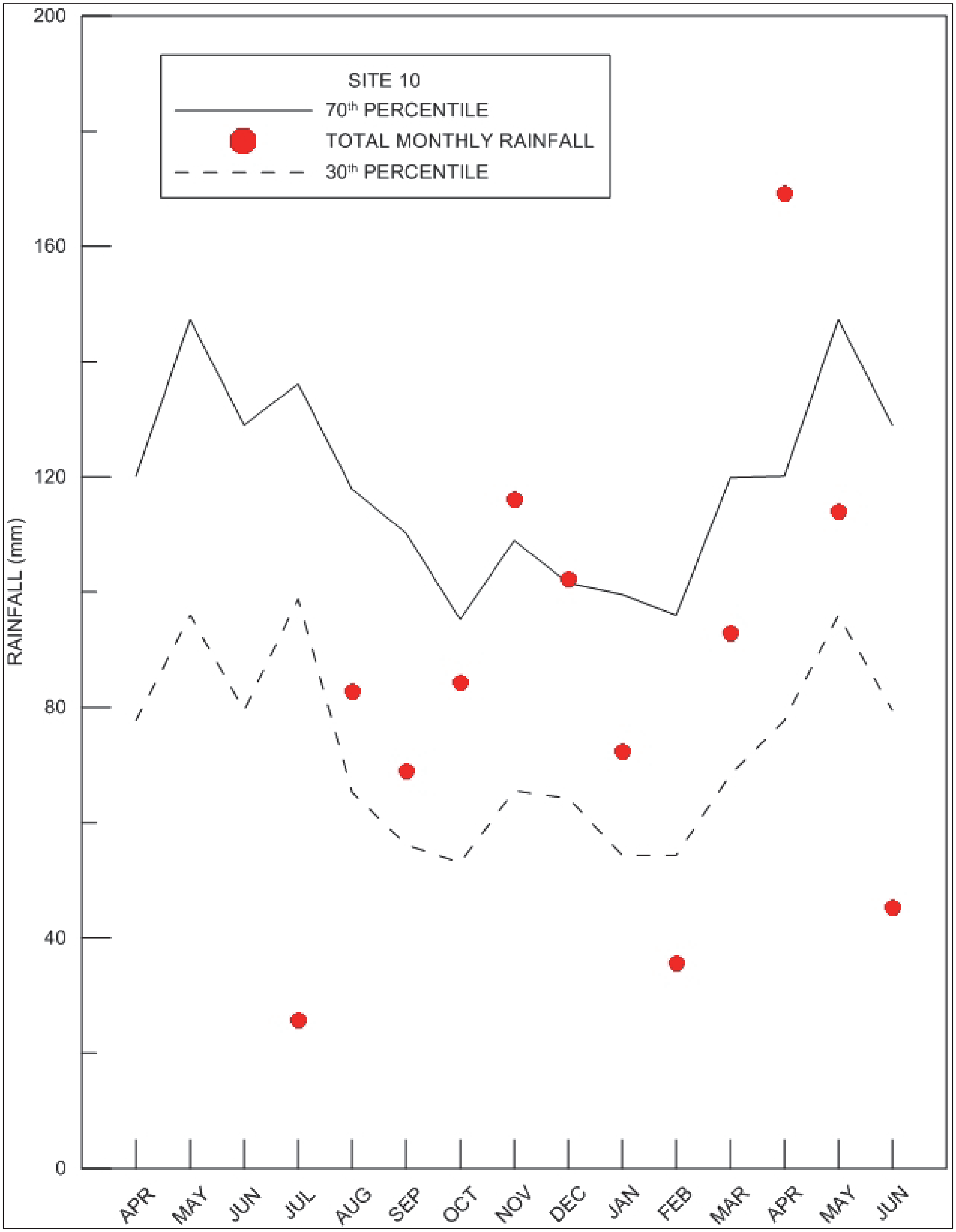


Figure B.10. Monthly rainfall site 10. Drier-than-normal conditions occurred in months falling below the 30th percentile line, wetter-than-normal months fall above the 70th percentile line, and normal rainfall months fall between the two lines.



Appendix C: Project Delivery Team Meeting: Summary of Recommendations

On January 11-12, 2011, a Project Delivery Team (PDT) consisting of 31 individuals representing federal agencies [USACE, US Environmental Protection Agency (EPA), US Fish and Wildlife Service (FWS)], state organizations (Kentucky Division of Water, West Virginia Department of Transportation, West Virginia Department of Natural Resources, others), and academia convened for a 2-day workshop in Charleston, West Virginia (see table in Preface). Prior to the meeting, PDT members received a draft list of potential independent measures, drawn from guidance provided by the HGM guidebook and available literature sources, for each of the ecological functions assessed by the HGM guidebook. PDT members discussed each proposed independent measure of ecological function, evaluated the appropriateness of each measure, considered potential alternatives, and provided in-depth recommendations for sampling procedures. Additionally, PDT members visited a proposed validation study site in person to discuss potential sampling strategies. Discussion of sampling methodologies for each independent measure continued until a consensus was reached among all participants. USACE staff took detailed notes containing the comments and recommendations of each contributor. A summary of the meeting proceedings, along with a finalized list of sampling variables and methodologies based on PDT recommendations, was provided to all PDT members. The following methods were agreed upon by the PDT for collecting independent, quantitative measures of function and selected ecosystem characteristics and processes. All variables collected and presented in the validation report were approved by the PDT.

Site selection criteria:

1. All study sites will be located within the reference domain and the HGM subclass addressed in the Guidebook.
2. Sites will capture the variety of common impacts observed within the reference domain and will include: forested areas with intact channels containing limited disturbance, constructed channels associated with valley fill activities, and impaired channels displaying silvicultural, agricultural, and urban impacts.

3. Sites will be located in areas with unlimited access to investigators. Public and private land managers must agree to allow access to study sites throughout the study period.
4. Study sites must be located close enough together to ensure timely access throughout the year. Investigators will be visiting site on an approximate weekly basis, and they must be able to access all sites within a two day period during rain events and intensive sampling initiatives.
5. Study sites will be evaluated on the basis of aspect, catchment size, estimated hydroperiod, underlying geology, and other factors. However, because the assessment model is designed to be robust across a large physiogeographic area site selection will not be limited and site will include some of the variability seen throughout the reference domain.

Function 1: Hydrology

Stream Discharge

Purpose: Stream discharge characteristics define HGM subclasses and drive the transport of abiotic and biotic factors. Stream water transport function measured. Increased stream discharge can lead to channel scouring. Measurements will be used to calculate stream discharge (cubic feet per second (cfs)) on an event (hydrograph) and temporal (monthly/annual) basis.

Method: A 60 degree trapezoidal flume will be installed at each study site. Each flume will be located at level positions within the study area; minor engineering of the site may be required to properly place weirs. Each flume will be outfitted with an automated pressure transducer (vented, AQUA TROLL 200, 15 PSIG) set to record stream level at 15 minute intervals.

Rainfall

Purpose: Precipitation and associated runoff are key contributors to stream flow in headwater systems. Rainfall data will be evaluated on event (hydrograph) and temporal (monthly/annual) basis in inches.

Method: One tipping bucket rain gauge will be installed at each study site. The gauge will be located in the nearest suitable open area to ensure accurate measurements of rainfall (inches). Rain gauges will be outfitted with event dataloggers.

Topographic Stream Survey

Purpose: These surveys will provide data on channel morphology.

Method: Stream surveys will be conducted at each study site in order to document channel width:depth ratio.

Sediment transport

Purpose: Transport of sediments by stream water provides abiotic nutrient sources and stream building materials to downstream reaches. Increased sediment transport can lead to stream sedimentation and stream instability.

Method: A sediment trap will be installed within the settling pool upstream of the trapezoidal weir at each sample site. Traps will be examined during each site visit and excavated when sediment has collected. Accumulated materials will be weighed and sieved to determine particle size classes (fines vs. coarse materials) and measured in lbs/cfs/yr.

Function 2: Biogeochemical cycling

Riparian/Buffer Zone and Stream Nutrient Inputs

Purpose: The influx of debris from the surrounding riparian/buffer zone provides a source of nutrients to the stream channel and downstream locations.

Method: Four debris traps will be installed at each site. Debris traps will be located within the riparian/buffer zone. Traps will be designed to evaluate quantitative and qualitative measurements of organic materials entering the system and will include measurements of overall debris inflow (lbs dry weight/yr) and debris nutrient content (C, N, P).

Nutrient Decomposition and Cycling

Purpose: The breakdown of debris in the stream channel and from the surrounding riparian/buffer zone provides a source of nutrients to the stream channel and downstream locations.

Methods: Eight mesh leaf decomposition bags will be placed at each site in the riparian/buffer zone. All materials used to fill the leaf bags will be collected at one sample location and thoroughly mixed to ensure an equal distribution of species composition and nutrient content within bags across the all sample sites. Leaf bags will be collected after 6 months. Material from the collected leaf bags will be dried at 175 °F (80 °C) in a drying oven for 24 hours and weighed to the nearest mg. Decomposition will be evaluated based on loss of material weight (dry weight basis) and nutrient composition (C, N, P).

Dissolved Nutrient Flux

Purpose: Dissolved nutrients within the water column represent nutrient flux out of the stream channel to downstream locations.

Methods: Water samples will be collected at each site on an event basis. Water samples will be analyzed for pH, conductivity, total nitrogen, nitrate, total phosphorous, soluble reactive phosphorous, total organic carbon, and dissolved organic carbon. Additionally at the time of sampling onsite measurements of pH, conductivity, and stream temperature will be collected. The export of nutrients from each sample location will be evaluated based on stream discharge (nutrient flux).

Riparian/Buffer Zone and Stream Temperature

Purpose: Temperature within the stream and riparian/buffer zone affect many biotic and abiotic cycling processes including stream bank particle shrink-swell, microbial growth and reproduction, and decomposition of organic materials.

Method: Five temperature loggers will be installed at each site. They will record temperature (degrees F) twice daily, and will be located in the ambient air, above and below riparian/buffer zone detritus, in the stream channel, and 4 inches (10 cm) below the soil surface. Temperatures will be evaluated on event and temporal basis.

Function 3: Habitat

Aquatic Benthic Macroinvertebrate Habitat

Purpose: Species abundance and diversity represent direct measures of habitat functionality.

Aquatic benthic macroinvertebrates are a metric used to assess water quality and habitat functions across a variety of ecosystem impacts and environmental stressors.

Method: Four basket samplers will be installed at each site. Baskets will be placed at approximately equal distances throughout the study reach. Baskets will be placed flush with stream substrate and staked down in order to maintain moisture and hold trap in place. Baskets will be sampled seasonally for macroinvertebrates. On each sampling occasion, the contents of each trap will be thoroughly washed and sieved. Benthic macroinvertebrates will be preserved in 95% ethyl alcohol and transported to the lab for identified to genus. Community metrics including taxa diversity and abundance will be calculated across all sampling periods.

Amphibian population

Purpose: Headwater streams are unique habitat types in which close interactions occur between aquatic and riparian ecosystems. These systems do not contain fish; making these areas prime rearing zones where amphibians are often considered the top predator species. Amphibians are a metric used to assess water quality and habitat functions across a variety of ecosystem impacts and environmental stressors.

Method: Eight plywood cover boards (4 on each side of the channel) will be installed at each site. Each board will measure 24 in. x 48 in. x 0.5 in. (61 cm x 122 cm x 1.3 cm). Cover boards will be checked once per month for the presence of amphibians. Amphibian species present in the channel will be evaluated using the basket samplers described above. All individuals observed will be identified and released. Species richness and abundance will be calculated across all sampling periods.

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14. ABSTRACT

Headwater streams within the Appalachian region perform a number of ecosystem functions potentially impacted by human alterations. The current study sought to validate a Hydrogeomorphic (HGM) assessment, examining 1) Habitat, 2) Biogeochemical Cycling, and 3) Hydrology functions of highgradient headwater streams in western West Virginia and eastern Kentucky. Validation of the Habitat function focused on: 1) abundance and richness of amphibians, 2) richness and composition of benthic macroinvertebrates, and 3) floristic quality. Validation of the Biogeochemical Cycling function examined 1) nutrient inputs, 2) processing, and 3) the stream loading of nutrients and materials. The Hydrology validation study examined: 1) surface hydrology, 2) sediment transport, and 3) stream channel geomorphology. Within all three functions, sites with high HGM scores performed Habitat, Biogeochemical Cycling, and Hydrology functions at increased levels compared to sites with lower scores. The HGM assessment effectively differentiated between high and low functioning headwater stream ecosystems, and validation results support the use of these functional assessments in headwater stream ecosystems.

15. SUBJECT TERMS

Appalachian region	Functional assessment	Intermittent streams
Biogeochemical cycling	Headwater streams	Salamander habitat
Ephemeral streams	HGM assessment	Stream hydrology
Floristic quality	High gradient streams	Stream macroinvertebrate
	Hydrogeomorphic assessment	Stream sediment transport

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