EFFECTS OF SEDIMENTATION ON AQUATIC SALAMANDER LARVAE IN SMALL STREAMS IN THE SOUTHERN APPALACHIAN MOUNTAINS

A Thesis

by

DANIEL R. DIETRICH

Submitted to the Graduate School

Appalachian State University

in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

May 2005

Major Department: Biology

EFFECTS OF SEDIMENTATION ON AQUATIC SALAMANDER LARVAE IN SMALL STREAMS IN THE SOUTHERN APPALACHIAN MOUNTAINS

A Thesis

by

Daniel R. Dietrich

May 2005

APPROVED BY:

Robert Wayne Van Devender
R. Wayne Van Devender
Co-Chairperson, Thesis Committee

Michael Windelspecht
Co-Chairperson, Thesis Committee

Robert P. Creed
Robert P. Creed
Member, Thesis Committee

Steven W Seagle
Steven Seagle
Chairperson, Department of Biology

Edelma Huntley
Edelma D. Huntley
Dean of the Graduate School

ABSTRACT

EFFECTS OF SEDIMENTATION ON AQUATIC SALAMANDER LARVAE IN SMALL STREAMS IN THE SOUTHERN APPALACHIAN MOUNTAINS (May 2005)

Daniel R. Dietrich, B. S., University of North Carolina at Charlotte
M.S., Appalachian State University
Thesis Co-Chairs: R. Wayne Van Devender and Michael Windelspecht

Sediment is a common water pollutant in the United States and has negative effects on macroinvertebrates, fish, and stream-dwelling amphibians. However, the effect of sedimentation on lotic salamanders in the southern Appalachian Mountains has received little study, and few studies have evaluated how sedimentation produces declines in the abundance of salamanders. Between 2003 and 2004 I conducted a study in headwater streams of northwestern North Carolina to determine the effect of sedimentation on the abundance of salamander larvae. Stratified random sampling of salamander larvae showed that sediment-affected streams contained significantly fewer larvae than did unaffected streams. In sediment-affected streams, significantly fewer larvae were found in pools. Pebble count measurements showed significant effects of particle size distribution on salamander abundance. Abundance correlated negatively with fine particles (d= 2mm) and positively with large substrate particles (d=256->362mm). Relationships between the abundance of *Eurycea wilderae* and *Desmognathus quadramaculatus* and particle size distributions in streams differ, suggesting different niches for each species with respect to streambed characteristics.

The effect of sedimentation on the growth of *E. wilderae* was evaluated experimentally. After one month, growth of *E. wilderae* was not different in enclosures with no, low, and high sedimentation conditions. Since sedimentation did not reduce growth rates, changes in abundance in sediment-affected streams could not be attributed to growth effects. The role of predation in combination with sedimentation conditions was also evaluated in a six-day study of mottled sculpin (*Cottus bairdii*) predation on *E. wilderae.* Predation in enclosures was unaffected by sedimentation levels. The evaluation of how sedimentation causes declines in larvae salamander abundance deserves further attention. *E. wilderae* and *D. quadramaculatus* are considered common today, but larvae abundance declines as sedimentation increases. The prevalence of sediment pollution in headwater streams of northwestern North Carolina poses a threat to populations of salamander species with aquatic life stages.

ACKNOWLEDGMENTS

I am grateful to the many people that supported my endeavors in the present study. I am indebted to my wife, Susan Dietrich, for her love, her assistance in the field, her sharp eye for editing, and her unswerving support of me in this pursuit. I would like to thank the co-chairs of my thesis committee, Drs. Wayne Van Devender and Michael Windelspecht, and my committee member Dr. Robert Creed, for their guidance, assistance, and insight. My parents, Walt and Barbara Dietrich, provided emotional and financial support, as did my mother- and father-in-law, Sandra and Bob Balfour. I would like to thank Dr. Howie Neufeld for his patient assistance with statistics. I thank Dr. Gabrielle Katz of the Geography Department for her input on substrate sampling and for allowing me the use of field equipment. Chris Wood was instrumental in locating study sites. I thank Tracey Hutcherson and Andy Coleman for their assistance with field work. I thank Chris Ulrey and Bambi Teague of the National Park Service for their help in obtaining permits for access to sites along the Blue Ridge Parkway. Finally, I would like to thank the Cratis D. Williams Graduate School for funding.

DEDICATION

To Susan, whose inquisitiveness and joy of discovery have inspired me.

TABLE OF CONTENTS

	Page
List of Tables	ix
List of Figures	x
Introduction	1
Materials and Methods	11
Results	30
Discussion	54
Literature Cited	67
Vita	73

LIST OF TABLES

Page

Table 1. Study site characteristics. 15

Table 2. Sampling dates for salamander larvae. 23

Table 3. Reach-averaged pebble count results. 32

Table 4. Sedimentation rates observed in study streams. 38

Table 5. The number, density, and species of salamander larvae collected from each study site. 39

Table 6. Habitat use by salamanders in study reaches. 42

Table 7. Correlation analysis of salamander larvae abundance and stream characteristics. 45

Table 8. Number of salamander larvae in growth experiment enclosures. 50

Table 9. Randomized block ANOVA results for larvae growth (mg/mg/day) for *Eurycea wilderae* in response to three levels of sedimentation. 51

Table 10. ANOVA results for the instantaneous growth (mg/mg/day) of *Eurycea wilderae* larvae in enclosures with different final density. 51

Table 11. Effects of sediment on predation by sculpin (*Cottus bairdii*) on Blue Ridge Two-lined salamanders (*Eurycea wilderae*) larvae. 53

Table 12. Presence or absence of potential predators of salamander larvae 61

LIST OF FIGURES

Page

Figure 1. Location of study streams. 14

Figure 2. Zigzag pebble count along a preplanned path. 18

Figure 3. Experimental design for growth experiment. 26

Figure 4. Percentage of fine particles (≤ 2m) in study reaches. 33

Figure 5. Particle size distributions of stream reaches 34

Figure 6. Abundance of salamander larvae in sediment-affected (n=5) and unaffected streams (n=3). 40

Figure 7. Habitat selection by salamander larvae in streams. 43

Figure 8. Salamander abundance and coarse sand. 46

Figure 9. Salamander abundance and pebbles. 46

Figure 10. Salamander abundance and small boulders. 47

Figure 11. Salamander abundance and bedrock. 47

Figure 12. *Desmognathus quadramaculatus* abundance and coarse gravel. 49

Figure 13. Effects of sediment treatment on instantaneous growth rate of *Eurycea wilderae* after one month 52

INTRODUCTION

Water pollution is defined as any addition to water that is harmful to aquatic organisms and can degrade the quality of water for recreation, industrial uses, or drinking water purposes (Miller 1998). Common water pollutants include heavy metals and hydrocarbons, fertilizers, pesticides, chemicals that change the pH of water, microorganisms from various sources, and sediment from soil erosion (Cole 1983, Miller 1998, Beasley and Kneale 2002). Pollutants enter waterways from both point sources and non-point sources. Point sources of pollution are single, identifiable sources such as drainpipes, whereas non-point sources of pollution are the result of runoff from large land areas (Miller 1998).

The maintenance of water quality in headwater streams, the source of most watersheds, is important to the maintenance of water quality throughout the watershed (Waring and Schlesinger 1985). A common water pollutant in the small, headwater streams of the mountains of North Carolina is sediment. Sediment is also the most prevalent form of water pollution in the United States (USEPA 1990). Sediment in streams occurs naturally as a result of the erosional processes found in watersheds (Waters 1995), but sediment pollution is the result of greater-than-natural input caused by land disturbance (Sutherland et al. 2002). Sediment pollution results from point sources like road construction and runoff from existing roads (Reid and Dunn 1984). In addition, increases in sediment loads in streams come from non-point sources of sediment such as land clearing and agriculture (Waters 1995), forestry practices (Berschta 1978, Corn and Bury 1989), and the erosion of stream banks (Hooke 1979, Trimble 1994). Stream modifications, such as channelization and installation of culverts, causes changes in patterns of sediment accumulation within streams (Wellman et al. 2000).

Sediment pollution in streams may either be suspended in the water column or settle to the streambed. Turbidity results from sediment suspended in the water column, while sedimentation is the deposition of those suspended sediments on the substrate. The size of inorganic particles suspended in a water column depends on both sediment source and water velocity (Waters 1995). Particles 2-60μm in diameter (e.g., silts and clays) remain suspended in the water column for long periods but larger particles, such as sand and pebbles, settle more rapidly (Waters 1995). Sands and pebbles are deposited near their source on stream substrate, but the finest particles settle only in very slow water.

Sediment pollution, of all sorts, often has negative effects on stream organisms. Suspended sediments such as silts and clays can produce fish mortality (Lloyd 1987) or reduce the hunting efficiency of visually-hunting fish (Sweka and Hartman 2001). Suspended sediment in streams reduced growth rates in trout even though there was more invertebrate prey drifting in the water column (Shaw and Richardson 2001). Sediment-induced gill damage is also a common cause of reduced growth in fish (Shaw and Richardson 2001). Suspended sediment also affects macroinvertebrates. Silt impairs the efficiency of filter feeding macroinvertebrates by clogging their nets or mouthparts (Brunskill et al. 1973). Macroinvertebrate density in streams is reduced by increases in macroinvertebrate drift rates, which result from high turbidity (Shaw and Richardson 2001).

Sedimentation changes stream substrates and produces habitat change. Habitat alteration may result from a shift from coarse to finer particles. Local effects of sedimentation vary depending on the geological characteristics of the stream (Kedzierski and Smock 2001). Sedimentation probably causes greater change in gravel or cobble habitats because settling of fine sediment on sandy-bottomed streams would not produce much change (Kedzierski and Smock 2001). Mountain streams typically have substrates composed of large, rocky particles (Rosgen 1994). Fine particles from the water column alter rocky streambed habitats by filling in the interstitial spaces between larger particles (Waters 1995). Subsequent loss of habitat negatively affects many benthic organisms (Kedzierski and Smock 2001). Sedimentation effects include the loss of sunlight for photosynthetic periphyton, which can reduce or eliminate those periphyton from stream substrates (Power 1990). Sedimentation reduces water flow rates and dissolved oxygen availability in the substrate (Rinne 2001). Benthic organisms may also have difficulties finding food in sediment-affected habitats (Gillespie 2002).

Interstitial spaces are microhabitats used by a wide variety of macroorganisms, such as benthic macroinvertebrates. The loss of interstitial spaces and habitat alteration contribute to sedimentation-induced declines in macroinvertebrate abundance and diversity (Waters 1995). Sediment-induced loss of interstitial spaces in coarse-substrate streams results in decreased abundance of macroinvertebrates (Peckarsky 1985). Elimination of refugia from water current caused by fine sediment in coarse substrates increases the risk for benthos of being swept into the water column. Increased drift risk may be a mechanism of sediment-induced reductions in macroinvertebrate abundance (Bond and Downes 2003). Macroinvertebrate species diversity and biomass can also be decreased by sedimentation (Lenat et al. 1979). Organic material on stream substrates, a food source for many macroinvertebrates, is retained best in coarse substrate material (Rempel et al. 2000) and may be reduced when sedimentation increases embeddedness of coarse substrates. Sedimentation can reduce feeding opportunities for filter feeding insects (Lemly 1982). Sedimentation therefore appears to reduce the availability of food for macroinvertebrates as well as decreasing the availability of cover. In addition to abundance effects, sedimentation can alter the macroinvertebrate species composition of streambeds (Cooper 1987). There is often a correlation between the particle size distributions of the substrate and the abundance and diversity of benthic organisms; fewer species of macroinvertebrates are found in fine-particle substrates than in heterogeneous, coarse substrates (Minshall 1984).

Sedimentation impacts fish as well. The abundance of fish using coarse substrates for spawning is reduced in streams affected by sediment (Sutherland et al. 2002). In addition, land-clearing reduces cover for streams and results in increased sediment inputs and reduced fish abundance (Sutherland et al. 2002). Sediment may reduce fish abundance by increasing their egg mortality; Rinne (2001) demonstrated that experimental manipulations of fine sediments ($\leq$ 2mm) reduced the survival of trout eggs to emergence. Substrates composed with just 25% of fine sediments were affected enough to reduce egg survival by between 88% and 96% (Rinne 2001).

Sediment in streams can also indirectly affect fish populations. A loss of cover in streams, such as that caused by sedimentation, affects the growth of fish because they spend more time avoiding predation and not foraging (Allouche and Gaudin 2001). Allouche and Gaudin (2001) found that the amount of cover for fish in sediment-affected streams was reduced. Cover availability affected how fish responded to presence of terrestrial predators. With cover, fish forage more and grow faster (Allouche and Gaudin 2001).

Macroinvertebrates and fish have been the most studied stream organisms in terms of the effects of sedimentation on lotic organisms; the effect of sedimentation on stream amphibians has received less attention. Sedimentation effects on amphibians may be important because lotic amphibians often remain in streams for several years, and as such can be particularly sensitive to disturbances caused by sedimentation. It seems reasonable that amphibians share some of the same responses to sedimentation seen in macroinvertebrates and fish (Waters 1995). Amphibians are often the only vertebrate predators in very small streams (Davic 1983), so environmental disturbances that affect the abundance of important predators have the potential to alter stream community structure. For example, *Desmognathus quadramaculatus*, the black belly salamander, is a lotic salamander commonly found in northwestern North Carolina. *D. quadramaculatus* exerts a strong predation pressure on some highly competitive macroinvertebrates, and may act as a keystone species in streams without fish (Davic 1983). Because of their longevity and exposure to environmental conditions, the abundance of stream amphibians may be indicative of stream environmental conditions in headwater streams, more so than the abundance of macroinvertebrates (Welsh and Ollivier 1998).

Amphibian populations in general merit scrutiny. Populations world-wide are in decline, though there have not been not enough long term studies that document normal population fluctuations to ensure that observed regional declines in amphibian populations are not part of a normal population cycle (Blaustein 1994). Most agree, however, that anthropogenic disturbances of habitats are detrimental to amphibian populations. Aquatic amphibians are affected by water pollution, like pesticides found in the Piedmont of North Carolina (Hall and Prouty 1985).

Most research on the effects of sedimentation on lotic amphibians is from the Pacific Northwest. In the Pacific Northwest, amphibians are negatively affected by all sorts of land use changes that increase sediment inputs to streams. Corn and Bury (1989) evaluated the density of amphibians in 43 Oregon streams, 23 of which were in unlogged stands and 20 streams that passed through logged stands, where sedimentation in streams is characteristically higher. There were fewer lotic salamanders and frogs in the 20 streams that were in logged areas when compared to the 23 streams that were in unlogged areas (Corn and Bury 1989). Another study (Kelsey 1995) also indicates that amphibian abundance is reduced in streams subjected to sediment pollution.

Populations of *Ascaphus truei*, the tailed frog, and *Dicamptodon tenebrosus*, the Pacific Giant Salamander, were negatively affected by sedimentation in streams that flowed through logged areas. Kelsey (1995) found that amphibian population sizes were more variable and generally smaller in streams affected by sedimentation from logging sources when compared to streams in areas that were not logged. Streams in logged stands also had greater amounts of fine sediment and substrates composed of smaller particles than streams that were not logged. Kelsey (1995) also found that different stream habitat factors were correlated with differences in amphibian abundance. Frog populations were negatively affected by the amount of stream substrate embeddedness, the degree to which large substrate particles are buried by fine particles, whereas salamander abundance was associated only with the size of particles in the streambed. The diversity of particle sizes was negatively associated with the abundance of salamanders.

Kelsey's (1995) work indicates that the effects of sedimentation may also be more pronounced in some habitats within streams than others. Greater sediment depositions often occur in pools when compared to riffles (Lisle and Hilton 1992) because of slower water velocities in pools. Welsh and Ollivier (1998) found that the affect of sedimentation on amphibians in streams varied between species and in some cases, on the habitat where the species were found. The density of *Dicamptodon tenebrosus* and *Rhyacotriton variegatus*, the southern torrent salamander, was significantly lower in sediment-affected streams than in pristine streams (Welsh and Ollivier 1998). Habitat use also varied by species. The authors found that *Ascaphus truei* population density was not affected by sedimentation *per se* but was significantly reduced in pool habitats in sediment affected streams. *Dicamptodon tenebrosus* population density in contrast was significantly reduced by sedimentation but this effect was not dependent on habitat. These studies indicate that amphibian species responses to sediment vary in the streams of the Pacific Northwest .

Work in Australia demonstrates that sediment can reduce growth rates of amphibians (Gillespie 2002). Gillespie (2002) analyzed the rate of growth of lotic tadpoles of spotted tree frogs (*Litoria speceri*). The tadpoles live in streams and feed on periphyton, which can be smothered by sedimentation. Increased sedimentation produces slower growth rates and retards development of adult traits (Gillespie 2002). Apparently sediment results in an increased aquatic larval period, which increases exposure of amphibians to sub-optimal conditions in streams.

In West Coast studies, the abundance of amphibians in streams often declines in response to sediment pollution, as do fish and macroinvertebrates. In contrast to West Coast amphibians, the effect of sedimentation on amphibians that inhabit the East Coast of North America has received little attention. Answering questions about the effect of sedimentation on salamander larvae in streams on the East Coast will provide insight into the generality of sedimentation effects on amphibians. In Watauga County, and adjacent areas in northwestern North Carolina, amphibians with aquatic life stages in small streams are salamanders in Family Plethodontidae (Williams 1983, Conant and Collins 1998, Zug et al. 2001). *Desmognathus fuscus*, *D. monticola*, *D. marmoratus*, *D. quadramaculatus*, *Eurycea wilderae, E. guttolineata*, *E. longicauda*, *Gyrinophilus porphyriticus*, *Psuedotriton montanus*, and *P. ruber* are all species in Watauga County with lotic larvae (Williams 1983). Of these species, *E. guttolineata, E. longicauda,* and *P. montanus* were infrequently encountered in Watauga County by Williams (1983). The remaining salamander species were either commonly found or abundant in the survey (Williams 1983), making them candidates for an evaluation of the effects of sedimentation on the lotic amphibians of northwestern North Carolina.

Differences in larval periods between species mean that each species of aquatic salamander larvae receive different lengths of exposure to conditions in streams. Semi-terrestrial amphibians, such as the salamanders found in northwestern North Carolina, may be particularly sensitive to environmental degradation because they may be exposed to environmental degradation in one or both of their habitats—terrestrial or aquatic ecosystems (Pechmann and Wilbur 1994). *Desmognathus fuscus* has a larval period of only a few weeks (W. Van Devender, personal communication). *D. monticola* are thought to have larval periods of between 8 and 11 months (Petranka 1998). *Eurycea guttolineata* has a larval period of approximately one year, whereas *E. wilderae* larval periods can vary between one and two years (Bruce 1982, Petranka 1998). *Psuedotriton montanus* has a larval period of approximately one and a half years and *P. ruber* has a larval period of approximately two and a half years (Petranka 1998). *D. quadramaculatus* larvae metamorphose after three or four years as larvae (Petranka 1998). Finally, *Gyrinophilus porphyriticus* is thought to have a larval period of approximately four years (Petranka 1998).

Based on information from other studies of lotic amphibians, as well as fish, it is reasonable to suspect that sedimentation may affect growth and abundance of larval salamanders in streams in northwestern North Carolina, yet little is known about the effect of sedimentation on the salamanders in low-order streams in the Appalachian mountains. Headwater streams (first, second, and third order streams) represent the majority of stream length in mountainous areas (Eric Hiegl, personal communication). Headwater streams are abundant in mountainous areas of northwestern North Carolina, while larger streams are rare. These streams are often too small to support fish, but are habitat for salamanders, another vertebrate predator. Agriculture and rural development are prevalent forms of land use in Watauga County (Williams 1983) as they are in all of the northwestern mountains of North Carolina. Sediment pollution is the typical water quality threat in these streams, which are usually small, and remote enough not to receive point source pollution. The objective of this study was to determine if sedimentation affects lotic salamander larvae in small streams in northwestern North Carolina. Furthermore, several studies have shown that there is an effect of sedimentation on amphibians in streams but have not experimentally evaluated how sedimentation reduces abundance (e.g., Corn and Bury 1989, Welsh and Ollivier 1998). The objectives of this study were therefore twofold:

- To determine if sediment pollution changes the quality of streambed habitat for lotic salamander larvae.
- To evaluate the mechanism of the effect of sedimentation on salamander larvae if a negative sedimentation effect was observed.

Specifically, I set out to test the following hypotheses: 1) The total abundance of salamander larvae is lower in streams affected by sedimentation than unaffected streams. 2) Salamander larvae in moderate to high-sediment conditions grow more slowly than in substrates not affected by sedimentation. Finally, 3) Salamander larvae in sediment-affected habitats are more susceptible to predators than larvae in unaffected habitats.

MATERIALS AND METHODS

This study incorporated both field studies of salamanders in streams with different levels of sedimentation and field experiments to detect sediment effects on growth and survivorship. Sedimentation condition in several headwater streams in northwestern North Carolina was evaluated. The abundance of salamander larvae was compared between streams deemed to be affected by sedimentation and streams unaffected by sedimentation. Stream substrate measurements were compared to salamander larvae abundance to evaluate the effect of sedimentation on abundance. To evaluate mechanisms of sediment effects on larvae, salamander larvae growth under different sedimentation conditions was evaluated experimentally. Predation was then evaluated as a potential mechanism of sediment effects on abundance.

Selection of Field Sites

I searched for several first and second order streams in northwestern North Carolina for inclusion in the field study of sedimentation effects on salamanders. Between 2002 and 2004, as many streams as possible that were likely to either 1) be negatively affected by sedimentation or, 2) be unaffected by sedimentation were visited for evaluation as appropriate study streams. Streams that flowed through areas where land had been disturbed in the recent past, and streams that were crossed by either gravel or paved roads, were thought likely to be affected by sedimentation. Streams that flowed through undisturbed, forested tracts were thought likely to be unaffected by sedimentation.

A total of 23 headwater streams in Watauga, Caldwell, and Madison counties, North Carolina, were evaluated as potential study sites. Potential study streams were evaluated for inclusion in the study using several criteria. Stream accessibility, sedimentation condition, adequate study reach size, amount of riparian zone cover, and stream order were evaluated. The presence or absence of fine sediments and the presence or absence of identifiable sediment sources in potential streams were noted. USGS 7.5 minute maps were referenced for potential sources of sediment, such as road crossings. However, sedimentation is often caused by non-point sources (Waters 1995) and it is not surprising that I did not observe sediment point sources in all streams. An inability to identify a point source of sedimentation did not exclude a site from inclusion in the study.

Study streams were also selected based on their accessibility by automobile and the presence of continuous, wadable reaches of approximately 300 meters, a study reach (Kelsey 1995). I chose to sample a 300-meter study reach for two reasons. First, from observations of several stream reaches 300 meters in length, it appeared that 300m reaches were representative of sedimentation conditions in streams, where the degree of sedimentation often varied from point to point in the streambed. Secondly, many study streams were several kilometers long, too great a length to sample from mouth to headwater in a reasonable amount of time. Two streams had study reaches less than 300m because of the lack of continuous riparian zone cover for 300m along the study reach; this did not exclude these streams from inclusion in the study (see Table 1). The remaining study reaches were 300m long.

Only streams with forested riparian zones were selected so that potential habitat for adult salamander populations was present. Larvae in low order streams in northwestern North Carolina, with the exception of *Desmognathus marmoratus*, often have terrestrial adult stages. All of the streams used in the study had a vegetated riparian zone along the length of the study reach. Streams with patchy or missing vegetated riparian zones were excluded from study.

Only first or second order streams were considered for study because these streams provide potential habitat for salamanders and are also easily sampled for sedimentation and salamander larvae. The order of streams was determined from United States Geological Survey (USGS) 7.5 minute maps. Streams were considered first order when they had no tributaries and second order when all of the tributaries were first order streams (Cole 1983). Field observations of tributaries were used to determine stream order when streams were not shown on USGS maps.

Using Welsh and Ollivier (1998) as a model, I initially planned on 10 study sites. Five streams were to be sediment-affected streams and five streams were to be unaffected by sediment. However, after a survey of the accessible first and second order streams in northwestern North Carolina counties, I was only able to identify eight streams that met the criteria for study sites (Figure 1).

Physical characteristics were evaluated for each study site (Table 1). Latitude and longitude of the starting point of each study reach was determined using a Magellan 315 handheld GPS unit (Magellan Corporation, San Dimas, CA) and verified using USGS 7.5 minute maps. Drainage basin areas were determined by digitizing and analyzing USGS 7.5 minute maps with ARCView 3.3 (ESRI, Redlands, CA). Stream

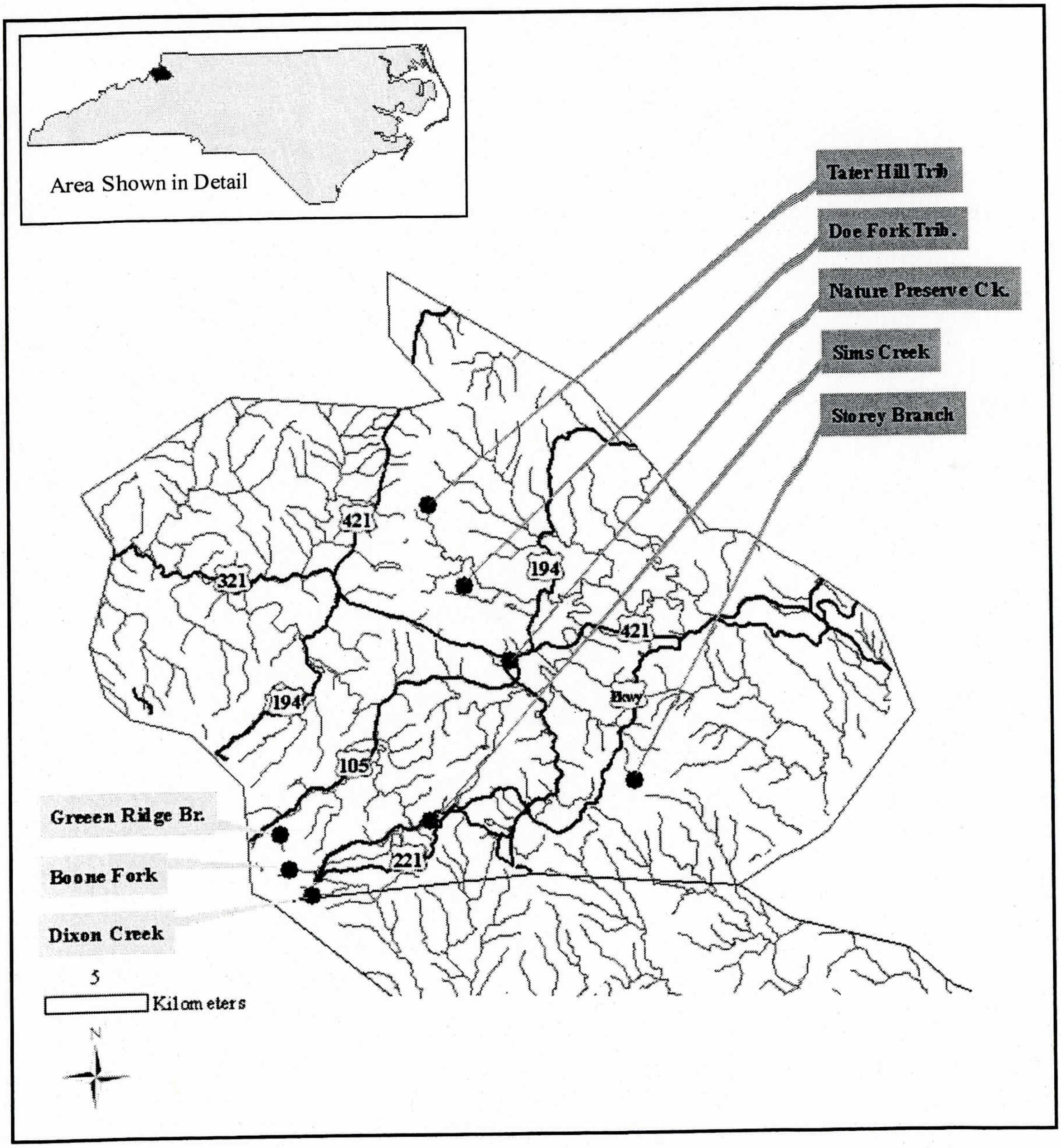

Figure 1. Location of study streams. Streams are in Watauga and Caldwell counties, in North Carolina. Green Ridge Branch, Boone Fork, and Dixon Creek were unaffected by sediment pollution. The Tater Hill Bog tributary, Doe Fork tributary, Nature Preserve stream, Sims Creek, and Storey Branch were affected by sedimentation.

Table 1. Study site characteristics. Latitude and longitude refer to the starting point of the study reach. N.A. denotes a drainage basin area that is not available.

Study Site	Latitude and Longitude	Sedimentation Condition	Stream Reach Length (m)	Stream Order	Stream Reach Drainage Basin Area (ha)	Study Reach Average Width (m)	Discharge (m^3/s)	Gradient (%)
Doe Fork tributary	36°14’20”N 81°41’08”W	High	227	First	112	2.4	0.015	18.6
Nature Preserve stream	36°12’48”N 81°41’57”W	High	300	First	n.a.	1.3	0.010	5.3
Sims Creek	36°08’46”N 81°42’42”W	High	300	Second	55.8	2.5	0.021	1.1
Storey Branch	36°10’30”N 81°36’38”W	High	300	First	203	3.2	0.037	4.7
Tater Hill Bog tributary	36°17’01”N 81°42’52”W	High	283	First	95.7	2.2	0.013	2.9
Boone Fork	36°07’15”N 81°46’38”W	Low	300	Second	287	5.1	0.123	8.5
Green Ridge Branch	36°07’56”N 81°48’57”W	Low	300	First	143	3.3	0.056	10.5
Dixon Creek	36°06’28”N 81°47’01”W	Low	300	First	n.a.	3.1	0.027	15.2

reach gradient was determined from USGS 7.5 minute maps as well. The average width of each study site was determined by measuring stream width at the beginning, mid-point, and end of each study reach. Discharge was determined at the down-stream end of the study reach using the velocity-area method (Dingman 1994).

Visual Evaluation of Sedimentation

The amount of sedimentation in stream substrates was initially assessed visually and later quantified using reach-averaged pebble counts (Bevenger and King 1995). Streams were initially categorized as sediment-affected or unaffected using visual evaluation of the entire study reach. The embeddedness of large substrate particles by fine particles (sand and silt) and the areal extent of fine sediments in the streambed was evaluated visually. For visual evaluation, fine sediments were assumed to be responsible for substrate embeddedness. Following the embeddedness classification scheme described by Kelsey (1995), streams where the majority of coarse particles in the streambed were embedded to 25% or less of their height by fine particles were considered low-sediment streams. Streams with the majority of coarse particles embedded between 25% and 100% by fine sediments were considered to be affected by sediment. In general, streams that appeared to be uninfluenced by sediment pollution had substrate habitats composed principally of large rocky particles with little fine sediment on the surface of the streambed. Sediment-affected streams were noted for their general lack of larger rocky particles; pebbles and gravels were often lacking in pool habitats and excessive fine sediments were often in riffle habitats of sediment-affected streams.

Substrate Measurements

Visual estimates of sedimentation condition were verified in each stream in two ways. First, reach-averaged zigzag pebble counts were conducted along the study reach of each stream. Secondly, sediment deposition rates were measured using tiles to collect sediment settling from the water column in three pools in each stream. Finally, sediment deposition rates were used to verify that differences in fine sediments in streambeds could be attributed to differences in sedimentation rates as opposed to pre-existing differences, such as differences in streambeds caused by variation in stream topography or local geology.

Distributions of particle sizes of stream substrates were determined using zigzag pebble counts. These counts produce quantifiable, spatially-integrated samples of stream substrates, which can be compared between stream reaches, or over time (Lisle et al. 1993). Zigzag pebble counts use a pre-planned path in each stream reach to eliminate or reduce errors associated with other pebble count methods (Bunte and Abt 2001b).

While conducting pebble counts, I stretched a meter tape in a pattern of diagonals across each stream reach so that the meter tape contacted the opposite streambank at approximately two times the stream width from the previous contact point (Figure 2). sampled 400 particles from each stream reach to obtain the minimum sample required to determine the median particle size with an error of ± 10% (Rice and Church 1996, Bunte and Abt 2001b). At each sample point, a particle was selected by removing the particle that was directly beneath the meter tape. Sampling interval distance at each site was determined using the equation of Bunte and Abt (2001a):

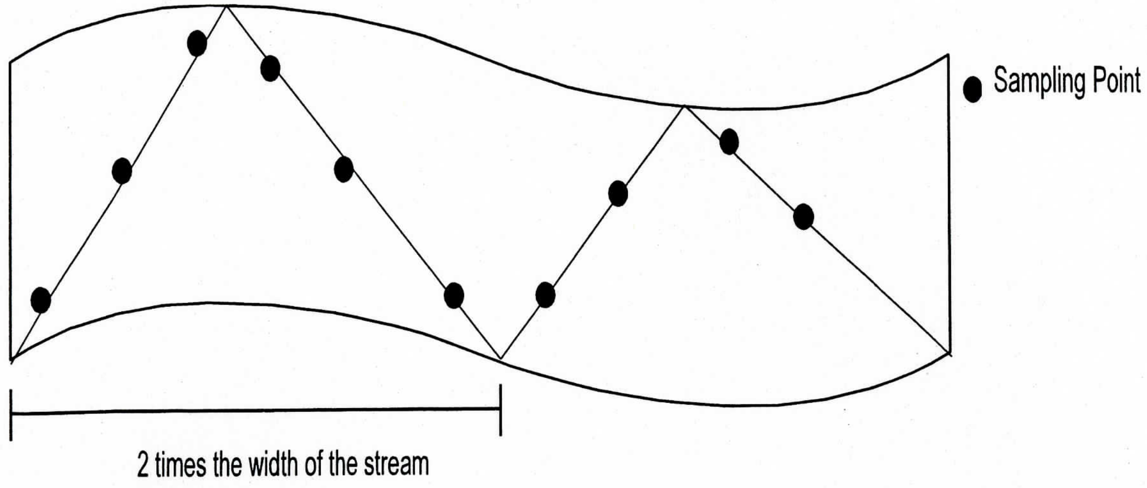

Figure 2. Zigzag pebble count along a preplanned path. Diagram after Bevenger and King (1995)

$$Lp=\frac{[\sqrt{(w^2+2w^2)} * Lt]}{(ns * 2w)}$$

where *Lp* is the spacing between particle sampling points, *w* is the average of stream width at the beginning, midpoint, and end of each study reach, *Lt* is the stream reach length, and *ns* is the number of particles to be sampled. Sampling intervals in study streams were typically between 0.40 and 0.60 meters.

Each particle was selected from the substrate by facing upstream, along the meter tape, and placing an index finger into the streambed directly beneath the calculated sampling point while averting my eyes from the streambed. The first particle that touched the inside right edge of the index finger (Kondolf 1997) was retrieved. The intermediate axis of particles was measured using an Al-Sci field sieve (Albert Scientific, West Trenton, NJ) as described by Bunte and Abt (2001a). The field sieve had square openings that correspond to ϕ particle size classes in 0.5 ϕ increments from –1 to –8.5 ϕ (2mm – 362mm). Template openings were: 2mm, 4 mm, 5.7mm, 8mm, 11.3mm, 16mm, 22.6mm, 32mm, 45mm, 64mm, 90mm, 128mm, 180mm, 256mm, and

362mm. Particles were assigned to a particle size class based on the smallest sieve hole that they would pass. When fine sediments were encountered, particle size was determined by taking a pinch of material and selecting a particle from that sample (Bunte and Abt 2001b); typically, all particles in these pinch samples were less than 2mm. Particles larger than the sieve length of 362mm were recorded as "greater than 362mm" and corresponded to boulders. Exposed bedrock was recorded as "bedrock."

Particle size data were grouped into categories adapted from the Wentworth scale for use in statistical analyses. The categories were: coarse sand ($\leq$ 2mm), fine gravel (4-8mm), coarse gravel (11.3-22.6mm), pebble (32-64mm), cobble (90-180mm), small boulder (256 - >362mm), and bedrock. Fine sediment ($\leq$ 2mm) was compared for each stream. Percentage of particles in the study reaches 2mm or less were arcsine transformed and subjected to one-way analysis of variance (ANOVA); (Zar 1999). Relationships between particles and the abundance of salamanders were evaluated using correlation analysis. Each count datum for salamanders and particles was transformed using the equation $x' = \sqrt{(x+0.5)}$ prior to analysis (Zar 1999). Significant correlations were further evaluated with regression analysis.

Sedimentation Rates

Sedimentation rates in each stream were measured to compare substrate compositions and sediment inputs. Five glazed ceramic tiles, measuring 11.43 cm by 11.43 cm by 0.7 cm were randomly placed in each of three randomly-selected pools. The location of each tile in the pool was marked with a surveyor's flag so that they could be relocated. After seven days, tiles were carefully removed from the streambed by hand to prevent the loss of accumulated sediment and immediately placed in sealed plastic bags. Bags were left open at room temperature in the lab for one week to allow the collected sediments to dry enough to be removed from the bag easily. Tiles and any residual sediment remaining in the bags were placed on sheets of aluminum foil in a drying oven at 60 °C for 48 hours. Dried sediment from each tile was weighed to the nearest 1.0×10^{-5} g using a Sartorius R200D scale (Sartorius Corporation, Bohemia, NY). Sedimentation rates were determined as the mass of sediment accumulated on the tiles per square centimeter per day. The total amount of accumulated sediment was determined by summing the weights of sediment collected on the 15 tiles from each stream. Sampling dates varied between 7/25/03 and 1/26/05. The sedimentation rates and total sediment accumulation were not normally distributed so were compared between sediment and unaffected streams using a Kruskal-Wallis test (Zar 1999).

Characterization of Habitats in Streams

Habitat types within streams were visually determined to facilitate amphibian sampling according to a habitat-based design (Welsh et al. 1997). Stream study reaches were measured with a 50-meter tape along the stream channel. Stream habitats were placed into either pool or riffle categories based on visual estimates of surface turbulence, velocity, and depth (Hawkins et al. 1993). If the water velocity was not visually apparent, it was roughly assessed by placing my hand in the water column. Slow, deep areas of a stream with visibly laminar flow at the surface were classified as pools. Fast, shallow areas of a stream with visibly turbulent flow at the surface were classified as riffles.

Salamander sampling

Abundance and density of salamander larvae were determined by randomly sampling five pools and five riffles in each stream using quadrats (Jaeger and Inger

1994). Quadrats were positioned randomly in each habitat by placing a meter tape along the bank of the stream and across the stream at each pool and riffle to be sampled. Quadrats were placed according to randomly-selected coordinates on the meter tapes. Each quadrat consisted of a square PVC frame constructed of 2.5cm diameter pipe measuring 50cm on each side. Three 0.25 m^2 quadrats were searched in each pool and riffle, for a total of 30 quadrats per stream and a total area of 7.5m^2 sampled in each stream. Habitats less than one meter long or less than 50cm wide were excluded from sampling as were pools deeper than one meter and stagnant side pools (Bar and Babbit 2001). However, deep pools and stagnant side pools were infrequently encountered in study reaches.

Different techniques were used to sample salamander larvae in pools and riffles. In pools, all available cover within a quadrat was removed and salamanders were located using a clear-bottomed, 10cm diameter length of PVC pipe. Salamanders were collected using turkey basters and aquarium nets (Barr and Babbitt 2001). Salamanders were collected from riffles by doing a visual search of the quadrat before placing a dip net downstream from the quadrat and vigorously disturbing the substrate with a hand-held rake; salamanders were then collected from the dip net. All salamanders were placed in plastic bags with stream water and taken to the lab for identification. Occasionally, salamanders were identified that eluded capture. The few salamanders that escaped were noted, but not included in analysis of abundance because identification was uncertain.

Salamander larvae were identified to species using keys by Petranka (1998). Salamanders were easily handled while in a plastic bag. A voucher specimen of each species encountered in a stream was anesthetized using MS-222, preserved in 3%

formaldehyde solution, and deposited in the Appalachian State University Vertebrate Collection. Salamanders not preserved were returned to their point of capture.

The total abundance of salamanders and the abundance of salamanders in pools and riffles in sediment-affected and unaffected streams were compared with a t-test. The occurrence of salamanders in pools and riffles was compared between streams using contingency table analysis with Yates Correction (Zar 1999). Each count datum for salamanders was transformed using the equation $x' = \sqrt{(x+0.5)}$ prior to correlation analyses (Zar 1999), as described above.

Streams were sampled for salamander larvae between 7/25/03 and 10/14/04 (Table 2). Streams were sampled as they were identified as suitable over a period of 15 months. Streams could not be simultaneously sampled. Sediment-affected streams and unaffected streams were sampled in both warm and cool months.

Growth Study

A growth experiment was conducted to determine the effect of sedimentation on salamander larvae. *Eurycea wilderae* larvae were chosen as the subject animals for three reasons. *E. wilderae* were common in seven of the eight streams sampled (see Results). Secondly, field observations suggested that *E. widerae* abundance was reduced in sediment-affected pools. Finally, *E. wilderae* would be expected to grow appreciably in a three month experiment whereas *Desmognathus quadramaculatus*, the other commonly-encountered species in study streams, might not grow appreciably during that time interval (Lugthart 1991).

The sedimentation experiment was conducted in Howards Creek, a third order stream in Watauga County, North Carolina. Howards Creek was chosen as the

Table 2. Sampling Dates for Salamander Larvae. Streams are listed in chronological order of sampling date. With the exception of Storey Branch, which was sampled on one day, streams were sampled over two days, with all pools sampled on one day and all riffles sampled on the subsequent sampling date. Salamanders were removed from the streams during sampling.

Study Site	Sedimentation Condition	Sampling Date(s)
Doe Fork tributary	High	7/25/2003, 7/28/2003
Green Ridge Branch	Low	7/30/2003, 8/4/2003
Tater Hill Bog tributary	High	8/18/2003, 8/20/2003
Storey Branch	High	11/10/2003
Boone Fork	Low	11/20/2003, 12/31/2003
Nature Preserve stream	High	2/19/2004, 2/24/2004
Sims Creek	High	3/5/2004, 3/8/2004
Dixon Creek	Low	10/14/2004, 10/17/2004

experimental site for the growth experiments for several reasons. First, Howards Creek is wide enough to accommodate 15 experimental enclosures. This is generally not possible with first or second order streams. Secondly, *Eurycea wilderae*, as well as the macroinvertebrates that are potential prey, can be found in abundance at Howards Creek. Third, Howards Creek is designated as a high quality water source by the State of North Carolina (Michael Windelspecht, personal communication). Finally, a small retention dam immediately upstream from the study area served to moderate water flow, water temperature, and suspended sediment regime, and to prevent enclosures from being washed downstream during storms.

The growth experiment was conducted in plastic enclosures placed directly into the stream. The enclosures were constructed from rectangular plastic boxes measuring 61cm x 40cm x 41.9cm. The bottom area of each enclosure was 0.244 m^2. An opening measuring 23cm x 26cm was cut out of each end of the enclosures and covered with a fiberglass screen with a mesh size of 1.2mm x 1.4mm. The mesh openings are small enough to prevent salamanders from escaping and large enough to allow macroinvertebrates access to the enclosures (Lugthart 1991). Steel hardware cloth with a mesh size of 6.3mm x 6.3mm was placed over the fiberglass screening to protect the screen from puncture. Screens were secured to the enclosures with steel nuts and bolts. The edges of all openings were sealed with waterproof silicon caulk to prevent escape of larvae. Fifteen enclosures allowed five replicates of three treatments in the growth experiment. Enclosures contained one of three treatments: no added sediment, low sediment, and high sediment. Each enclosure was filled with 10 kg of coarse rocky substrate with individual particles approximately 22mm in diameter. Ten kilograms of rocky substrate was adequate to cover the bottom of enclosures with approximately two layers of particles. Coarse substrate was visually selected from a road-accessible riffle in Gap Creek in Watauga County, North Carolina. Fine sediment for the enclosures was collected from a drainage ditch leading to Gap Creek and sifted using a screen with a 6.3 mm X 6.3mm opening. Sifting produces sediment with the same consistency, and secondly, a sediment that fills interstitial spaces (Bunte and Abt 2001a). The control treatment enclosures received no sediment. Each low sedimentation treatment enclosure received 1.4 kg of fine sediment to cause 25% embeddedness in the enclosure substrate. Each high sedimentation treatment enclosure received 2.3 kg of fine sediment, determined to cause 75% embeddedness in the enclosure substrate. Each treatment was replicated five times in a randomized block design (Figure 3).

Enclosures were placed in a pool in Howards Creek, below the retention dam. Enclosures were positioned outside of the stream thalweg, the area of greatest flow, in a pool with enclosure openings perpendicular to the direction of water flow (Lugthart 1991) to reduce sediment entering the enclosures from the water column. Enclosures were secured in the stream by tying them to three 1.27cm X 1.21 m steel rebar stakes driven into the stream substrate. Enclosures were covered with lids and allowed to acclimate in the stream for seven days prior to the addition of salamander larvae to permit colonization of the substrate by stream periphyton and macroinvertebrates.

Eurycea wilderae larvae for use in the growth experiment were collected from seeps and the main channel of Dixon Creek at US 221 in Caldwell County, North Carolina, on May 8 and May 17, 2004. A total of 109 *E. wilderae* were captured. The SVL and total length of the salamanders were measured to the nearest 0.1mm using calipers and wet weight was obtained by weighing salamanders in tared cups after blotting animals with a paper towel. Salamanders were maintained in water at 4 °C to minimize metabolism and growth prior to the start of the growth experiment.

Sixty salamanders were chosen from the pool of captured individuals for inclusion in the growth experiment. The remaining larvae were returned to Dixon Creek. Most larvae (48/60) used in the study were from the stream. It was not possible to acquire 60 salamander larvae of the same size. Animals of different sizes often experience different growth rates, so salamanders were split into size categories to mitigate differences in individual growth due to initial size. Salamanders were classified as one of four size classes on the basis of SVL. Classes were determined by arranging salamander larvae from shortest to longest SVL. Size classes were then separated on the basis of 15-salamander increments, that is, size classes consisted of 15 individuals so that one larva from each of four classes could be included in each of the 15 experimental enclosures. Size classes were (SVL): 13.8-16.3mm, 16.4-17.1 mm, 17.4-18.6 mm, and 18.7-19.6 mm. There were no individuals with a SVL between 17.1 and 17.4 mm in this

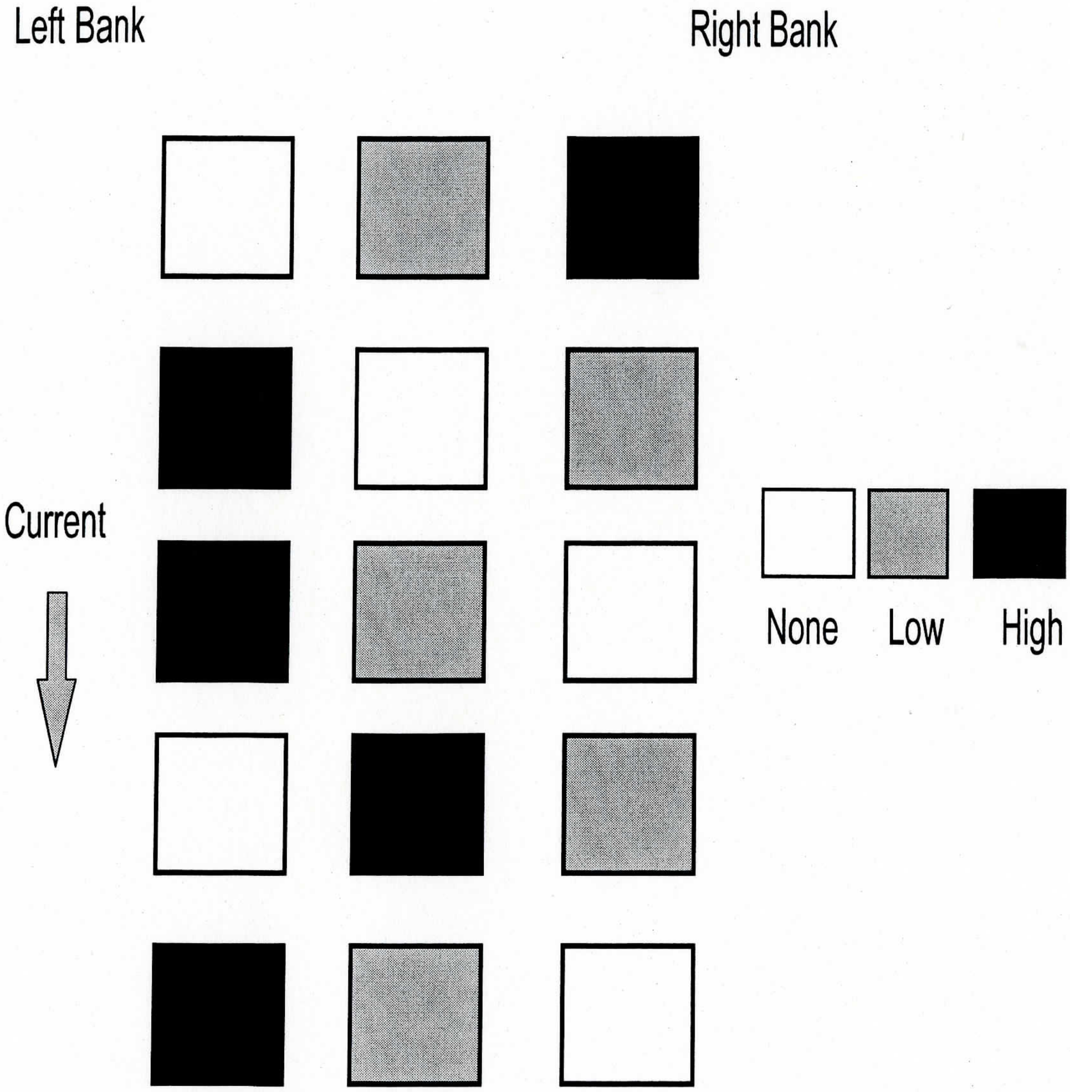

Figure 3. Experimental design for growth experiment. Treatments were none, low, and high added fine sediment. Each enclosure contained four *Eurycea wilderae* larvae.

study group. *E. wilderae* larvae typically metamorphose when they reach an SVL of between 21-23 mm (Petranka 1998). Eight *E. wilderae* were captured during May 8th and 17th that appeared to have recently metamorphosed from larvae. These individuals had an average SVL of 21.6 mm. Therefore, salamander larvae greater than 19.6mm SVL were excluded from the study to avoid possible metamorphosis from larvae to juveniles during the experiment.

Four salamanders were added to each enclosure on May 22, 2004. Enclosure densities of four *E. wilderae* larvae/0.244 m^2 were similar to the highest salamander densities observed in pools in Green Ridge Branch and Dixon Creek. Enclosures were monitored weekly. Screens were cleaned of any accumulated periphyton and debris using a toothbrush. Water in the enclosures was agitated by hand to reduce the accumulation of stream sediment, which was a different color than experimentally-added sediment. Macroinvertebrates (mainly Diptera and Ephemeroptera) were noted in all of the experimental enclosures. Water temperatures and velocity between enclosures was recorded.

Salamander growth was to be evaluated monthly for three months. After one month, larvae were removed from seven randomly-selected enclosures on June 21st and from the remaining eight enclosures on June 22nd. Larvae were again removed from study enclosures after two months on July 22nd. No larvae were recovered after three months. Larvae were recovered from enclosures by first visually searching enclosures and capturing visible salamanders with a turkey baster or aquarium net. Then, all coarse substrate was carefully removed from the enclosure and placed in a bucket. The bottom of the no sediment added treatment enclosures were searched for salamanders not encountered while removing the coarse substrate. Sediment in low and high-sediment treatment enclosures was removed with an aquarium net and placed in a pan, were the sediment was searched by hand to verify no salamanders remained buried in sediment. After ensuring that all salamanders in the enclosure were recovered, coarse substrate and then fine sediment were returned to the enclosures. Salamanders were transported to the laboratory in water in small containers placed on ice. Total length, SVL, and wet mass were measured for each salamander. Salamanders were kept at 4°C in water until they were returned to their respective enclosures after 24 hours.

Snout-to-vent length measurements were converted to ash free dry mass (AFDM) using Lugthart's (1991) equation for *E. wilderae*:

$$M=0.0023L^{3.09}$$

where M is the larval ash free dry mass in mg and L is the SVL in mm. Differences in growth between sediment treatments were tested using calculated AFDM values. The initial average AFDM of the four salamander larvae in an enclosure was subtracted from the average AFDM for that enclosure after one month. The difference was divided by the number of days the salamander larvae were in the enclosures (30 days for salamanders recovered on June 21st and 31 days for salamanders recovered on June 22nd). Instantaneous growth rate, the mass increase per unit mass of salamander per day, was compared between treatments using randomized complete block ANOVA.

Predation Experiment

The effect of sedimentation condition on the ability of salamander larvae to avoid predation was observed in two enclosures similar to those used in the growth experiment. Each enclosures contained 10kg of coarse sediment. One enclosure also contained 2.4 kg of added sediment as in the high sediment treatment of the growth experiments. Enclosures were also placed in Howards Creek. Two *Eurycea wilderae* larvae with SVLs of approximately 18-19mm and one sculpin (*Cottus bairdii*) approximately 7-8cm long was added to each enclosure. Sculpin were collected from the New River in Watauga County on September 1, 2004 and maintained in an aquarium until the start of the observation on September 11, 2004. Sculpin are ambush predators known to feed upon salamander larvae (R.Creed, personal communication). Sculpin were deprived of food for 48 hours before being placed in enclosures. After 24 hours, the sculpin were removed from the enclosures and the larvae remaining in the enclosures were counted. Salamanders were returned to enclosures approximately five minutes before sculpin. Salamanders missing from enclosures were replaced to maintain prey density in enclosures. This procedure was repeated for six days, between September 11 and September 17, 2004.

RESULTS

Selection of Field Sites

Five streams with significant sedimentation were easily located, but only three streams that were unaffected by sediment could be identified. Sediment sources in sediment-affected streams were not always evident. Sediment sources I observed included roads and streambank erosion. The Doe Fork tributary, Nature Preserve stream, and Tater Hill Bog tributary showed sings of streambank erosion. Additionally, the Nature Preserve stream appeared to receive large amounts of sediment in runoff from a gravel parking lot. A dirt road bisecting the Tater Hill Bog tributary apparently is a sediment source for that stream. Obvious sediment sources were not found in Storey Branch or Sims Creek. Sediment-affected streams tended to have smaller average drainages than unaffected streams (116 ha vs. 215 ha), narrower channels (2.35m vs. 3.83m), lower discharge rates (0.019m^3/s vs. 0.069m^3/s) and have lower gradients (6.5% vs. 11.9%); (Table 1).

Sedimentation Assessment

A comparison of the amount of fine particles ($\leq$ 2mm) in stream reaches, obtained from pebble counts, supported the initial visual classification of stream reaches as sediment affected or unaffected. Streams visually classified as sediment-affected because of the areal extent of fine sediments and the degree of substrate embeddedness had a greater amount of 2mm particles in stream reaches than streams categorized as unaffected by sedimentation (Figure 4). The percentage of fine particles in sediment-affected reaches was significantly higher than the percentage of fine particles in sediment-unaffected reaches ($F_{1,7}$=65.92, p=0.0002). Pebble count results for study streams are provided in Table 3. The particle size distributions of each stream substrate are shown in Figure 5. In addition, total sediment and sedimentation rates were greater in sediment-affected streams than in unaffected streams (H=27.9, d.f.=1, p<0.0001); (Table 4).

Salamander Sampling

The number of salamander larvae captured in quadrats varied between sediment-affected and unaffected streams. A total of 83 salamander larvae were captured from the eight study sites (Table 5). *Eurycea wilderae* (n=55) and *Desmognathus quadramaculatus* (n=25) were the most commonly encountered species. *D. marmoratus* were rarely found (n=3). Adults of all species were encountered in stream study reaches but were not included in analysis. One *Gyrinophilus porphyriticus* larva was seen in a riffle in Dixon Creek but escaped.

Salamander Abundance and Sediment

Several patterns were evident in where salamanders were found in streams that differed in sediment load. Overall, streams that were unaffected by sedimentation produced significantly more salamanders than did more pristine streams. The average of 15.6 ± 4.7; n=3 (mean ± SE) salamanders in 30 quadrats (7.5m^2 total sampling area) was significantly greater than the average 7.2 ± 1.2; n=5 (mean ± SE) salamanders in sediment-affected streams (t=2.46, d.f.=6, p=0.04); (Figure 6). In fact, more salamanders were found in three pristine streams than in five sediment-affected streams. However, Dixon Creek had a large number of *E. wilderae* captures (Table 5).

Table 3. Reach-averaged pebble count results. The table includes number of particles in each size class from the study reach of 300m. Points in the study reach selected for sampling that were bedrock were included in the pebble count results.

Study Site	Sediment Condition	Particle Size (mm)																	
		Coarse Sand	Fine Gravel			Coarse Gravel			Pebble			Cobble			Small Boulder				
		2	4	5.7	8	11.3	16	22.6	32	45	64	90	128	180	256	362	>362	Bedrock	Total
Doe Fork tributary	High	69	19	17	22	20	12	18	26	30	35	32	18	15	10	14	16	29	402
Nature Preserve stream	High	86	21	24	25	16	18	25	27	32	22	20	33	9	16	6	14	10	404
Sims Creek	High	85	16	13	22	24	23	27	35	33	40	30	22	13	8	1	4	5	401
Story Branch	High	60	23	30	21	21	18	29	18	27	30	22	22	10	18	19	20	16	404
Tater Hill Bog tributary	High	105	11	17	28	31	25	29	37	33	29	24	18	8	8	0	2	0	405
Boone Fork	Low	9	2	11	17	17	14	20	25	24	27	28	26	24	48	33	72	8	405
Dixon Creek	Low	15	5	12	16	19	17	18	20	23	22	17	26	8	14	19	55	102	408
Green Ridge Branch	Low	17	10	22	17	30	22	17	29	22	22	27	33	31	28	26	47	0	402

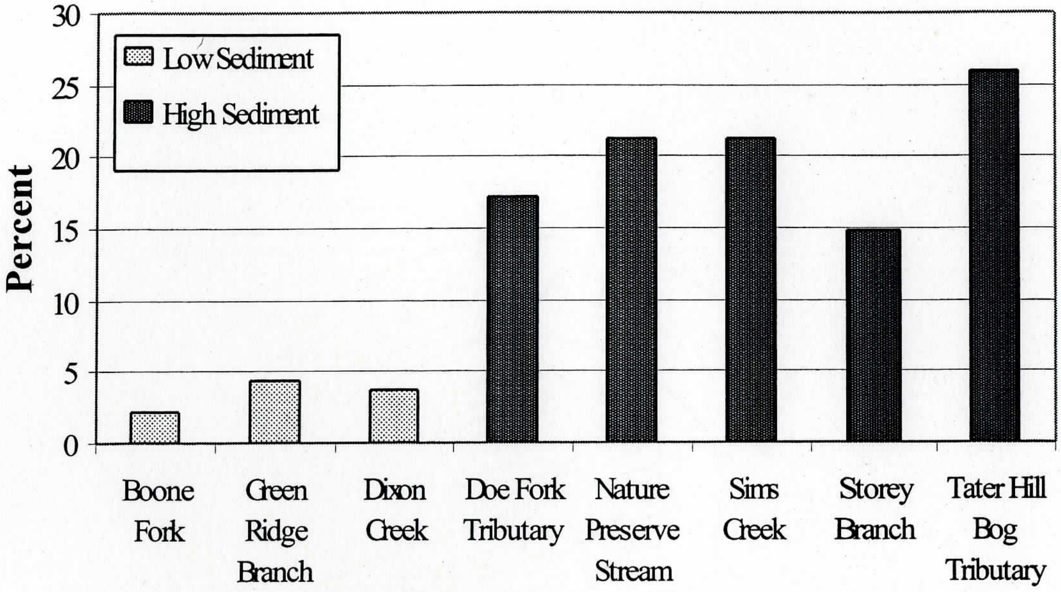

Figure 4. Percentage of fine particles (≤ 2 mm) in study reaches. The amount of fine sediments low-sediment streams is significantly lower in low sediment streams ($F_{1,7}$=65.9, p=0.002).

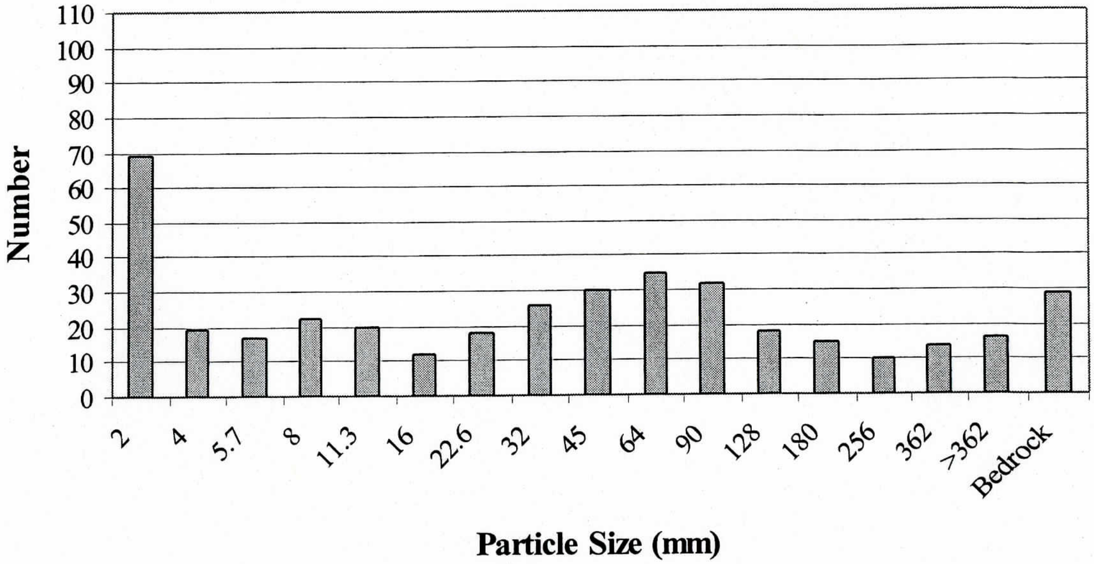

Figure 5a. Doe Fork tributary size distribution of approximately 400 particles.

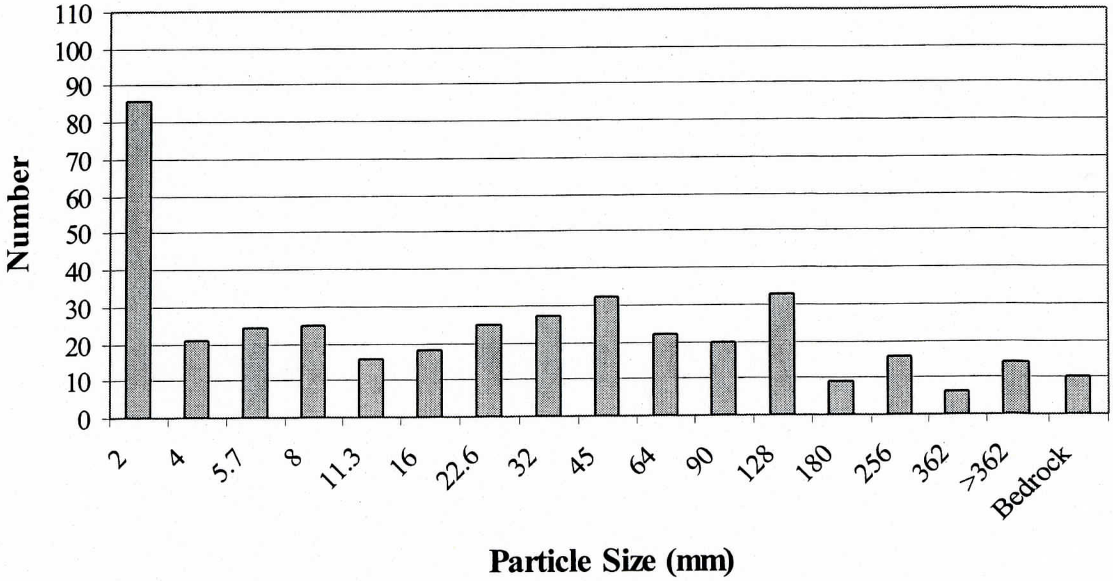

Figure 5b. Nature Preserve stream size distribution of approximately 400 particles.

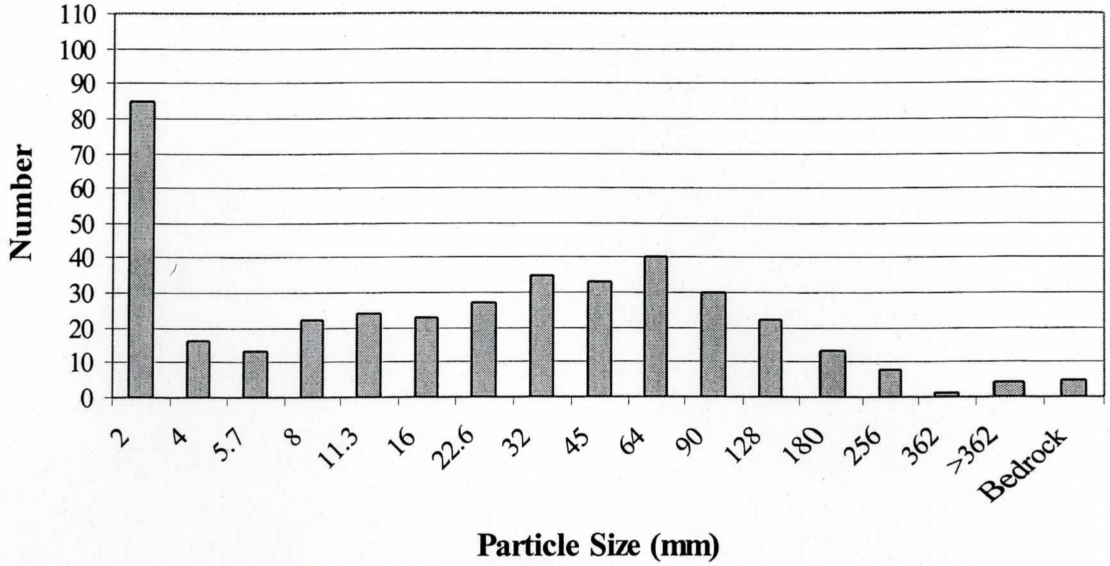

Figure 5c. Sims Creek size distribution of approximately 400 particles.

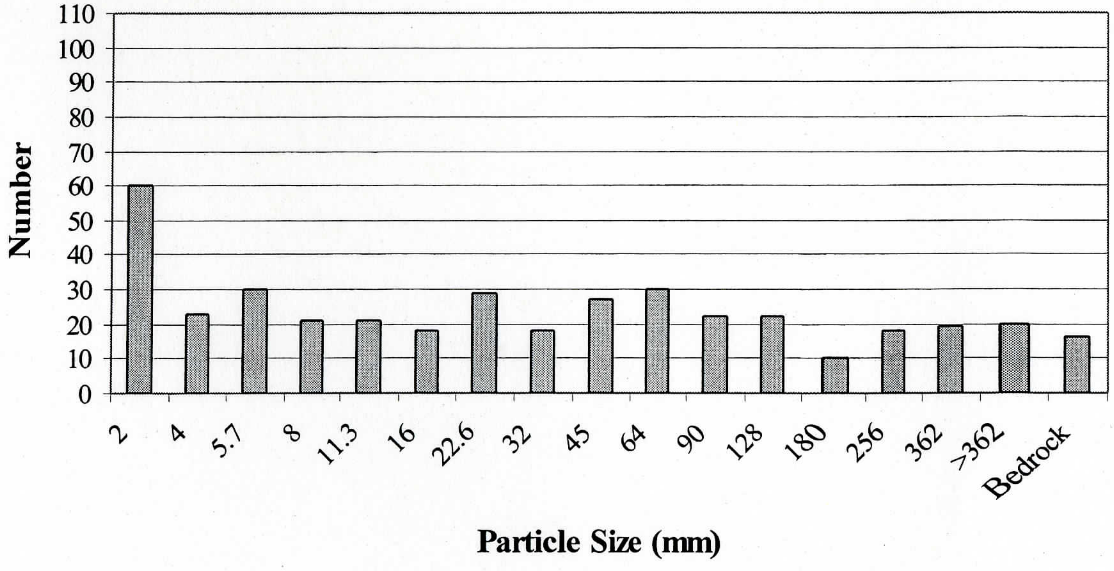

Figure 5d. Storey Branch size distribution of approximately 400 particles.

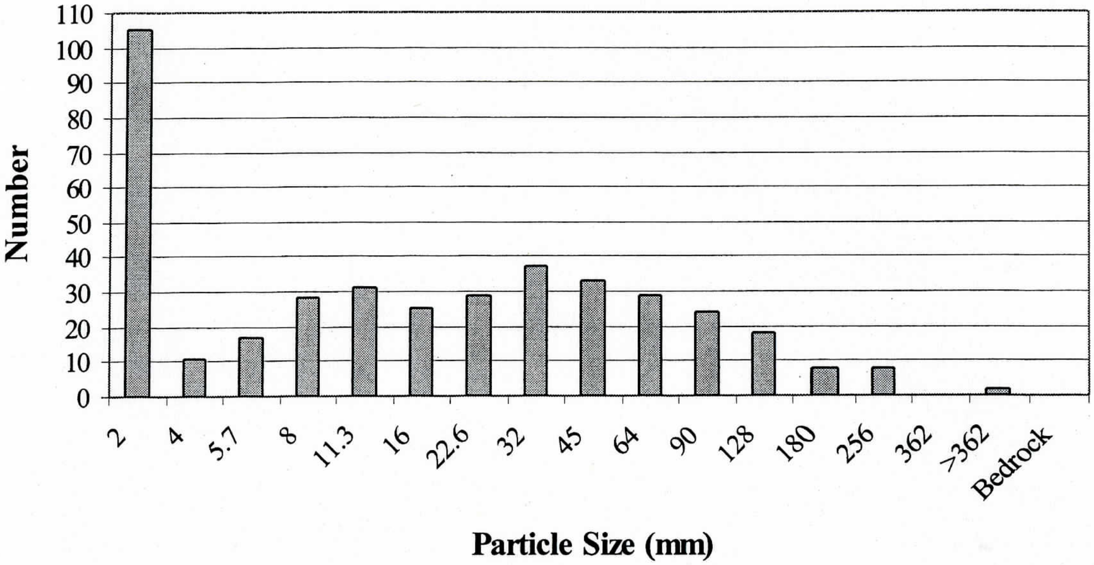

Figure 5e. Tater Hill Bog tributary size distribution of approximately 400 particles.

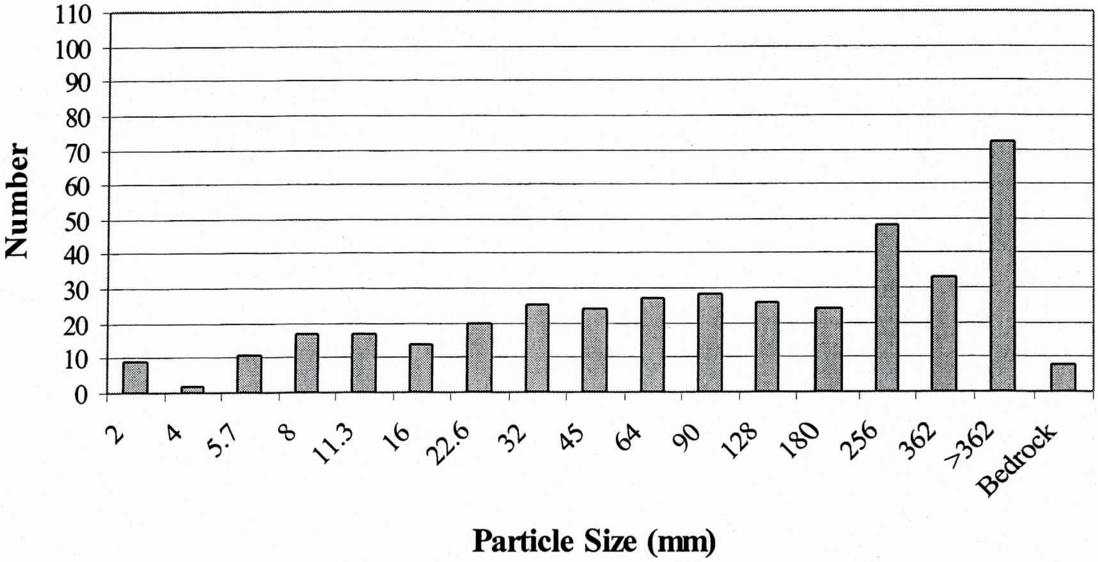

Figure 5f. Boone Fork tributary size distribution of approximately 400 particles.

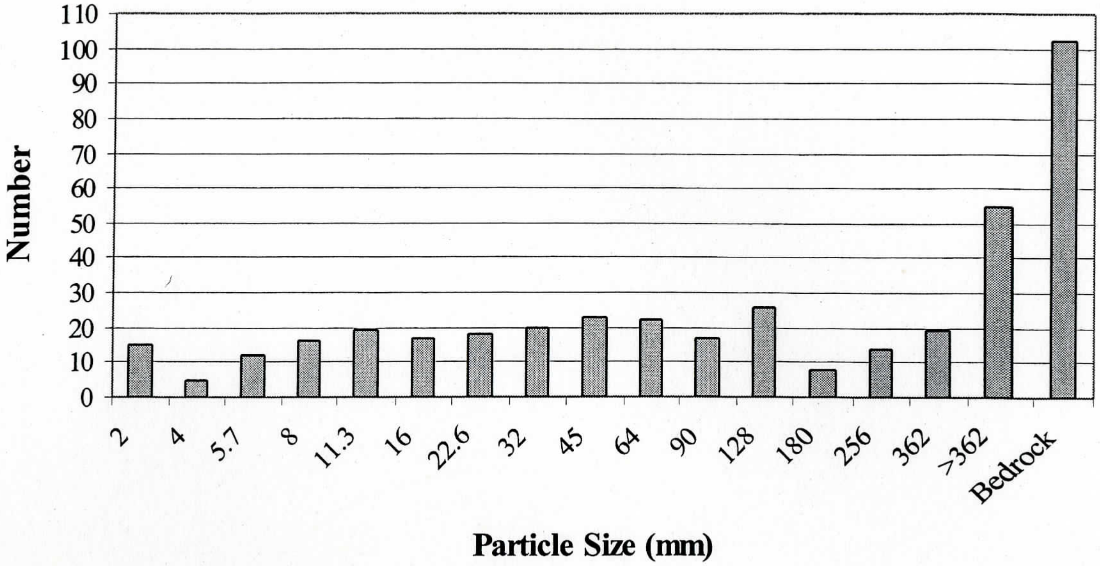

Figure 5g. Dixon Creek size distribution of approximately 400 particles.

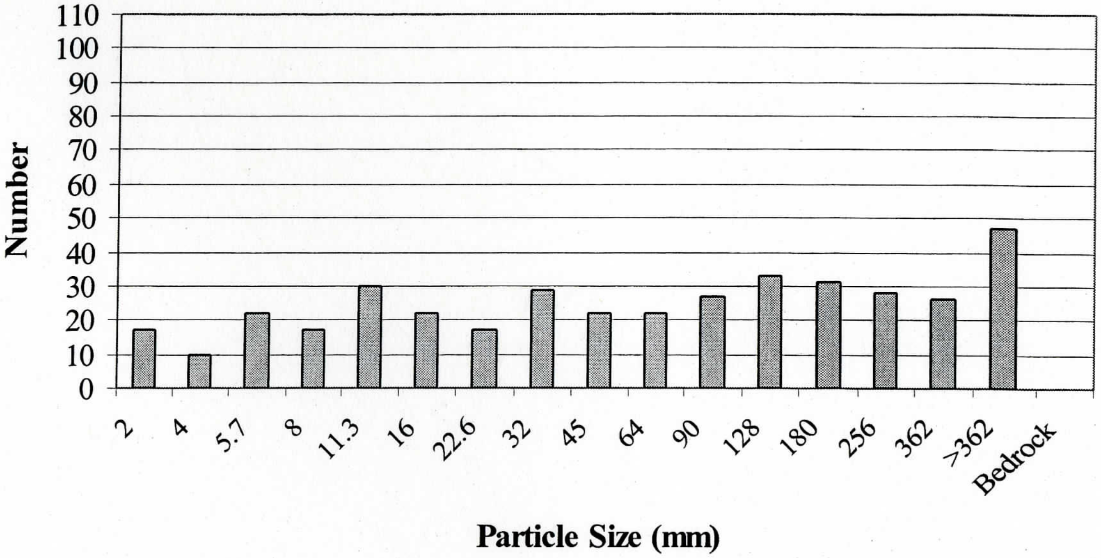

Figure 5h. Green Ridge Branch size distribution of approximately 400 particles.

Table 4. Sedimentation rates observed in study streams. Data are from 15 tiles, five tiles in each of three pools per stream.

Stream	Sampling Dates (Tiles in, Tiles out)	Sedimentation Rate (g/cm^2/day) (Mean ± SE)	Average Sediment per tile in 7 Days (g/cm2) (Mean ± SE)
Doe Fork tributary	7/25/03, 8/1/03	0.02683 ± 0.00976	24.543 ± 8.9229
Nature Preserve stream	1/21/04,1/28/04	0.00032 ± 0.28783	0.28783 ± 0.10383
Sims Creek	10/8/03, 10/15/03	0.00028 ± 0.00012	0.25548 ± 0.11072
Storey Branch	10/08/03, 10/15/03	0.00251 ± 0.00202	2.2943 ± 1.8490
Tater Hill Bog tributary	8/12/03, 8/19/03	0.00333 ± 0.00198	3.0483 ± 1.8118
Boone Fork*	10/15/03, 10/22/03	0.00013 ± 0.00012	0.12188 ± 0.10487
Green Ridge Branch	7/30/03, 8/6/03	0.00012 ± 0.00005	0.10910 ± 0.04258
Dixon Creek	1/26/05, 2/2/05	0.00001 ± 0.00001	0.01328 ± 0.00466

* Five tiles out of 15 were recovered after seven days

Table 5. The number, density, and species of salamander larvae collected from each study site.

Study Site	Sediment Condition	*Eurycea wilderae*	*Desmognathus quadramaculatus*	*Desmognathus marmoratus*	Total	Total Density (larvae/m^2)
Doe Fork tributary	High	0	10	1	11	1.5
Nature Preserve stream	High	5	4	0	9	1.2
Sims Creek	High	6	0	0	6	0.8
Storey Branch	High	3	2	0	5	0.7
Tater Hill Bog tributary	High	5	0	0	5	0.7
Boone Fork	Low	8	1	2	11	1.5
Green Ridge Branch	Low	9	3	0	12	1.6
Dixon Creek	Low	19	5	0	24	3.2

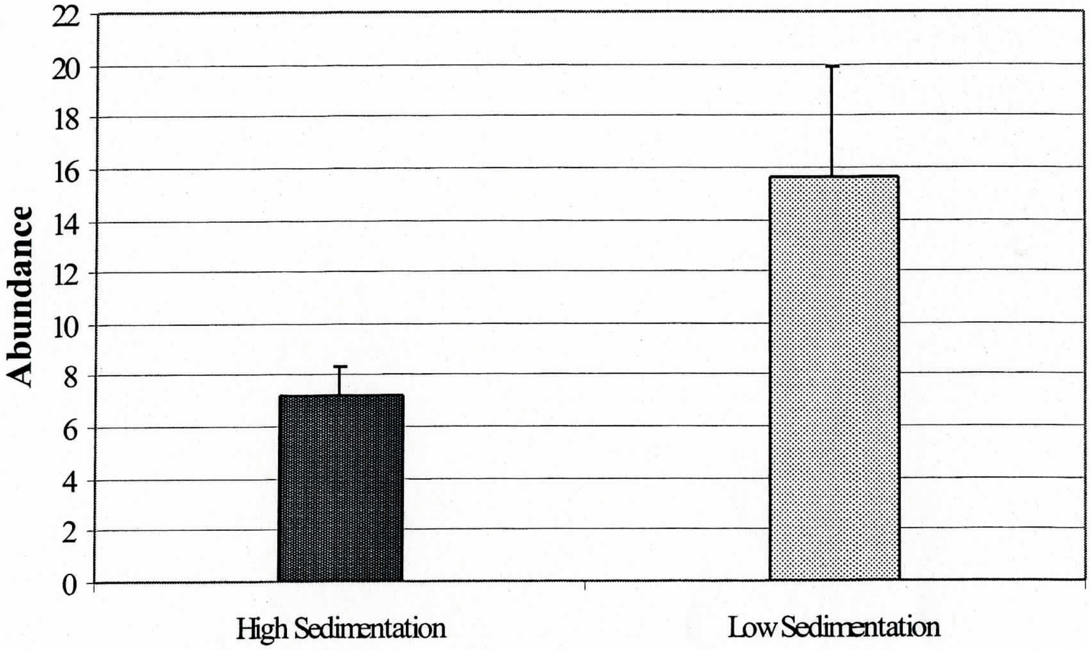

Figure 6. Abundance of salamander larvae in sediment-affected (n=5) and unaffected streams (n=3). Error bars represent the standard error of the mean. Thirty quadrats totaling an area of 7.5m^2 were searched in each stream for salamander larvae. Larvae are significantly more common in unaffected streams (t=2.46, d.f.=6, p=0.04).

Abundance trends in the sediment-affected and unaffected streams varied by species. Average abundance of *Eurycea wilderae* in unaffected streams (12 ± 3.5) (mean ± SE) was significantly greater than the average of 3.8 ± 1.1 (mean ± SE) *E. wilderae* in sediment-affected streams (t=2.8, d.f.=6, p=0.03). The abundance of *Desmognathus quadramaculatus* on the other hand was not significantly different between sediment-affected and unaffected streams. The abundance of *D. quadramaculatus* averaged 3.2 ± 1.9 (mean ± SE) larvae in sediment-affected streams and 3 ± 1.2 (mean ± SE) in unaffected streams.

Salamander Habitat Use

Salamanders utilized habitats differently in streams affected by sediment. In sediment-unaffected streams, more larvae (n=36) were in pools than in riffles (n=11). The reverse pattern was seen in sediment-affected streams (Table 6). In sediment-affected streams, more larvae (n=27) were found in riffles than in pools (n=9). This pattern of habitat use (Figure 7) was found to be significantly different between sediment-affected and unaffected streams using contingency table analysis (χ^2=19.8, d.f.=1, p<0.001). *Eurycea wilderae* habitat use patterns were similar. *E. wilderae* larvae were also more common in pools in unaffected streams (n=34) than in pools of affected streams (n=5). *E. wilderae* larvae use of riffles was greater in silted streams (n=14) than in unaffected streams (n=2). This pattern of habitat use by *E. wilderae* was significantly different (χ^2=24.7, d.f.=1, p<0.001). *Desmognathus quadramaculatus* was absent from several habitats in streams (Table 6) which precluded a contingency table analysis of

Table 6. Habitat use by salamanders in study reaches. E.w. denotes *Eurycea wilderae* and D.q. denotes *Desmognathus quadramaculatus*. D.m.denotes *D. marmoratus*. Numbers in the habitat column refer to all larvae captures in all pools and all riffles for a stream.

Study Site	Sedim ent Condition	Spec ies	Habitat	
Doe Fork	High	E.w.	0	0
		D.q.	3	7
		D.m	0	1
Nature Preserve	High	E.w	2	3
		D.q.	1	3
		D.m	0	0
Sims Creek	High	E.w.	2	4
		D.q.	0	0
		D.m	0	0
Storey Branch	High	E.w	1	2
		D.q.	0	2
		D.m	0	0
Tater Hill Bog	High	E.w.	0	5
		D.q.	0	0
		D.m	0	0
Boone Fork	Low	E.w	6	2
		D.q.	0	1
		D.m	0	2
Green Ridge	Low	E.w.	9	0
		D.q.	1	2
		D.m	0	0
Dixon Creek	Low	E.w	19	0
		D.q.	1	4
		D.m	0	0

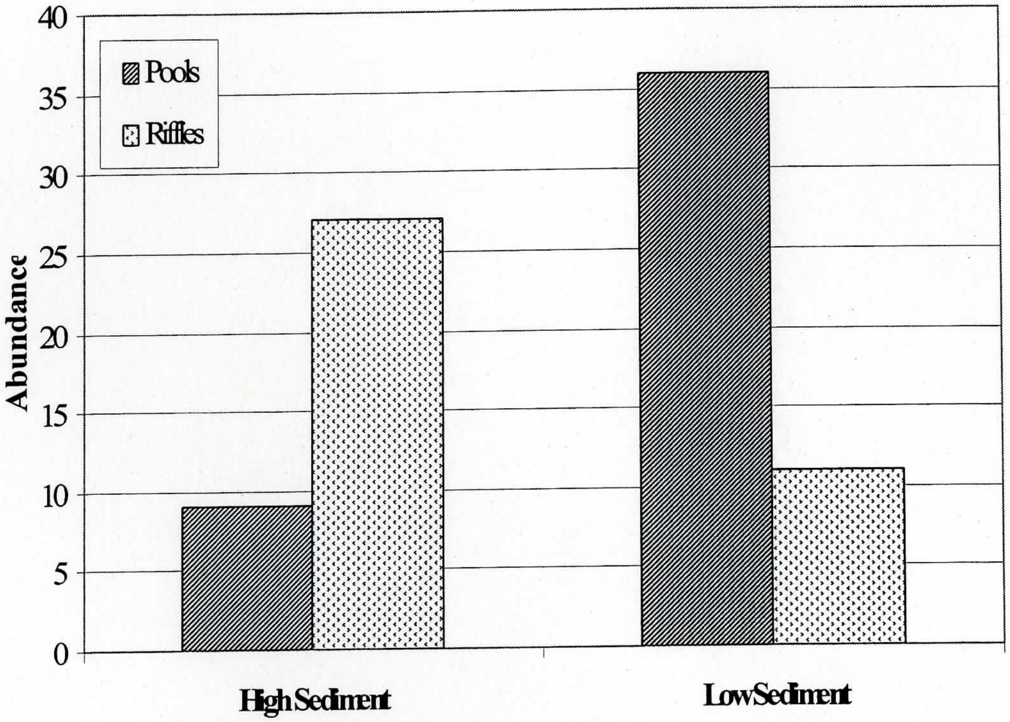

Figure 7. Habitat selection by salamander larvae in streams. The bars represent the combined sum of the numbers of salamander larvae captured in all pools and riffles for both the sediment- affected and unaffected streams. The numbers of salamanders occurring in pools and riffles is significantly different between sediment-affected and unaffected streams (χ^2=19.8, d.f.=1, p<0.001).

habitat use. Total abundance data in pools and riffles could also be used to evaluate whether differences in habitat use was produced by differential mortality or a movement from habitats. Examination of abundance data for individual plots in pools and riffles showed significant departures from normality in pools, while riffle data were normally distributed. Non-parametric analysis showed that the average of 1.8 ± 0.58 (mean ± SE) salamander larvae in sediment-affected pools was significantly less than the average of 12 ± 4.16 (mean ± SE) larvae in unaffected streams (H=5.06, d.f.=1, P<0.05). The number of salamander larvae occurring in riffles in sediment-affected streams (5.4 ± 0.74) was greater than the number in unaffected riffles (3.6 ± 0.88) but this difference was not significant (t=1.46, d.f.=6, p=0.19). Decreased abundance of salamanders in sediment-affected pools did not coincide with increased abundance in riffles in sediment-affected streams.

Salamander Abundance and Particle Size Distribution Relationships

In addition to significant relationships between sedimentation condition and the abundance of salamander larvae, there were significant relationships between some particle size classes and the abundance of larvae across all streams. Significant relationships between the abundance of salamanders and stream characteristics were identified using correlation analysis (Table 7). Regression analyses produced models predicting relationships between particles and salamander larvae abundance in all types of streams. Total abundance of salamander larvae decreased with the amount of course sand in the stream (r^2=0.519, p=0.04); (Figure 8) and amount of pebbles in the substrate (r^2=0.544, p=0.03); (Figure 9). Conversely, salamander abundance increased with the amount of small boulders (Figure 10) and bedrock (Figure 11) in the substrate (r^2=0.617, p=0.02 and r^2=0.536,p=0.03, respectively).

Table 7. Correlation analysis of salamander larvae abundance and stream characteristics. Significant correlation coefficients and their sign are given. Associations were considered significant if $p<0.05$. Correlations that were not significant are denoted with n.s. Marginal correlations ($p<0.10$) for *Eurycea wilderae* are also given.

Variable	Total Abundance	*Eurycea wilderae* Abundance	*Desmognathus quadramaculatus* Abundance
Coarse Sand	-0.720	-0.629[a]	n.s.
Fine Gravel	n.s.	-0.681[b]	n.s.
Coarse Gravel	n.s.	n.s	-0.784
Pebble	-0.738	n.s.	n.s.
Small Boulder	0.785	n.s.	n.s.
Bedrock	0.732	n.s.	n.s.
Stream Gradient	n.s.	n.s.	0.936
Stream Discharge	n.s.	n.s.	n.s.
Stream Width	n.s.	n.s.	n.s.

[a] $p=0.095$
[b] $p=0.063$

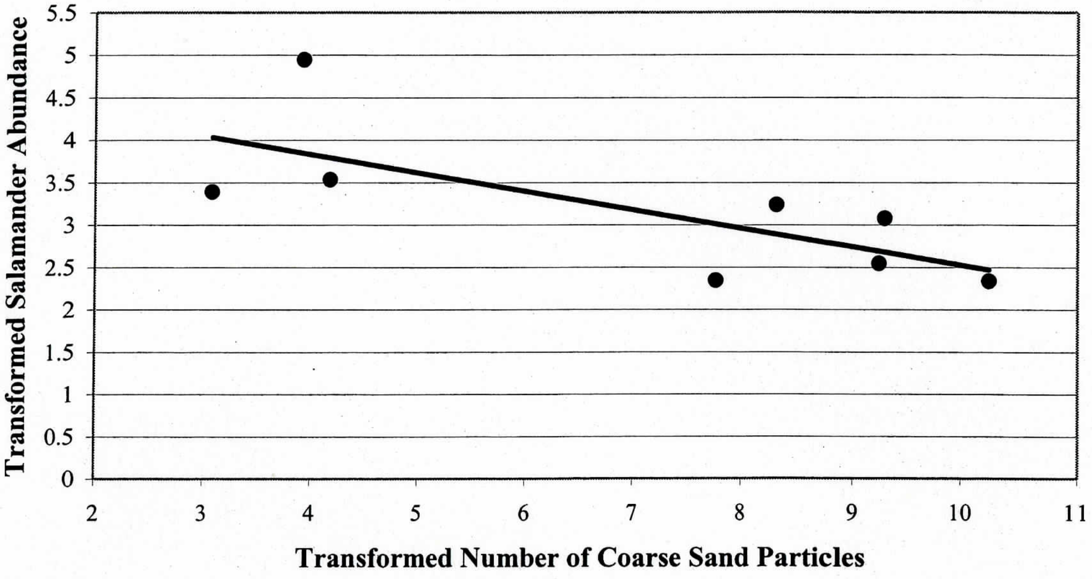

Figure 8. Salamander abundance and coarse sand. All data were transformed for analysis using √ (x +0.5). Coarse sand consisted of particles 2mm in diameter. The regression equation for the line is $y= - 0.2173x + 4.705$ ($r^2 =0.519$, $p=0.0439$).

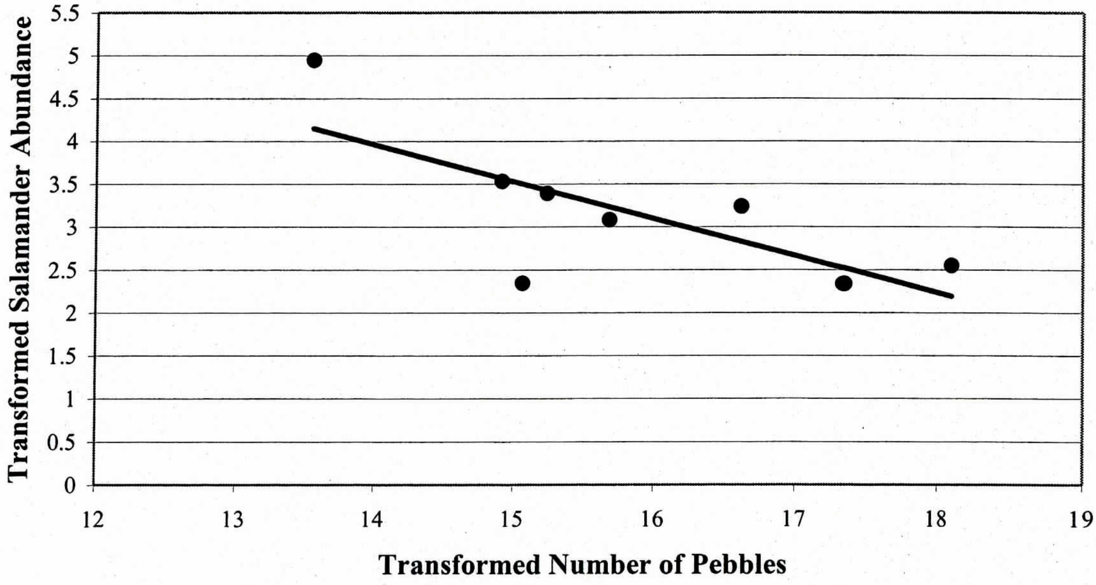

Figure 9. Salamander abundance and pebbles. All data were transformed for analysis using √ (x +0.5). Pebbles included 32mm, 45mm, and 64mm particles. The regression equation for the line is $y= -0.4296x + 9.9763$ ($r^2 =0.544$, $p=0.0367$).

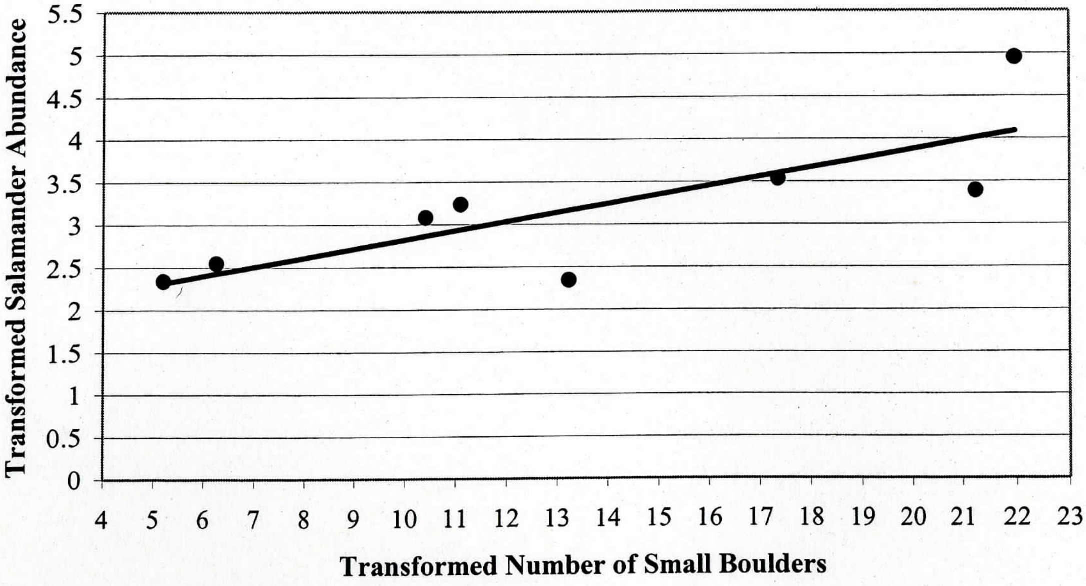

Figure 10. Salamander abundance and small boulders. All data were transformed for analysis using √ (x +0.5). Boulders included 256mm, 362mm, and >362mm particles. The regression equation for the line is y= 0.1054x + 1.7721 (r^2 = 0.617, p=0.021).

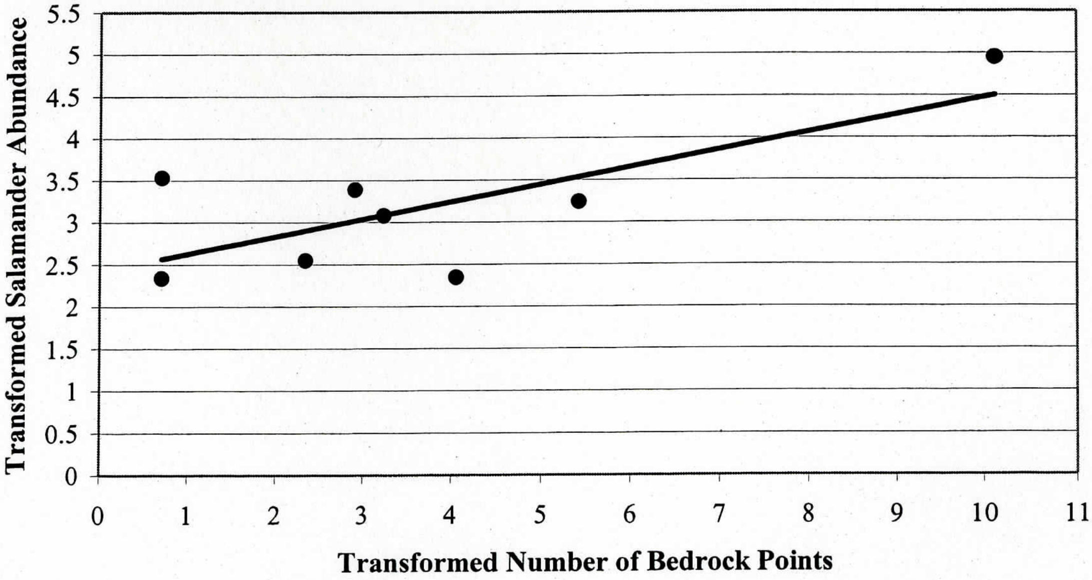

Figure 11. Salamander abundance and bedrock. All data were transformed for analysis using √ (x +0.5) Bedrock in the streambed counted during pebble counts were used analysis. The regression equation for the line is y= 0.2053x + 2.4220 (r^2=0.536, p=0.039).

The relationships between stream characteristics and the abundance varied between species. Correlation analysis revealed that total salamander abundance was not affected by physical differences between streams other than substrate conditions (Table 7). *Eurycea wilderae* and *Desmognathus quadramaculatus* were associated with the frequency of different particle size distributions in the study streams, for example. *E. wilderae* was not shown to be significantly correlated to any specific size particle class, where significance relationships were considered to be present when $\alpha<0.05$ (see Table 7). A negative relationship, though not statistically significant at $\alpha=0.05$, between coarse sand and fine gravel and the abundance of *E. wilderae* was evident from correlation analysis ($p=0.09$ and $p=0.06$, respectively). The abundance of *Desmognathus quadramaculatus* was negatively related to the amount of coarse gravel in the substrate ($r^2=0.538$, $p=0.03$); (Figure 12). No other particle size parameters were related to the abundance of *D. quadramaculatus* abundance (Table 7).

Growth Study

The growth of salamander larvae subjected to three different sedimentation treatments was to be evaluated over a three-month period. Water temperatures in enclosures were between 14 °C and 17 °C and water velocity between enclosures averaged 0.1m/s. After one month, 45 of the 60 salamander larvae initially placed in enclosures were recovered from the enclosures (Table 8). At least two larvae were found in each enclosure after one month. Larvae in the no sediment added treatment had an average instantaneous growth rate of 0.01717 ± 0.00056 mg/mg/day (mean ± SE) for the first month of the experiment. Larvae in the low sediment treatment had an average instantaneous growth rate of 0.01719 ± 0.00271 mg/mg/day (mean ± SE) for the first month. Larvae in the high sediment treatment had an average instantaneous growth rate

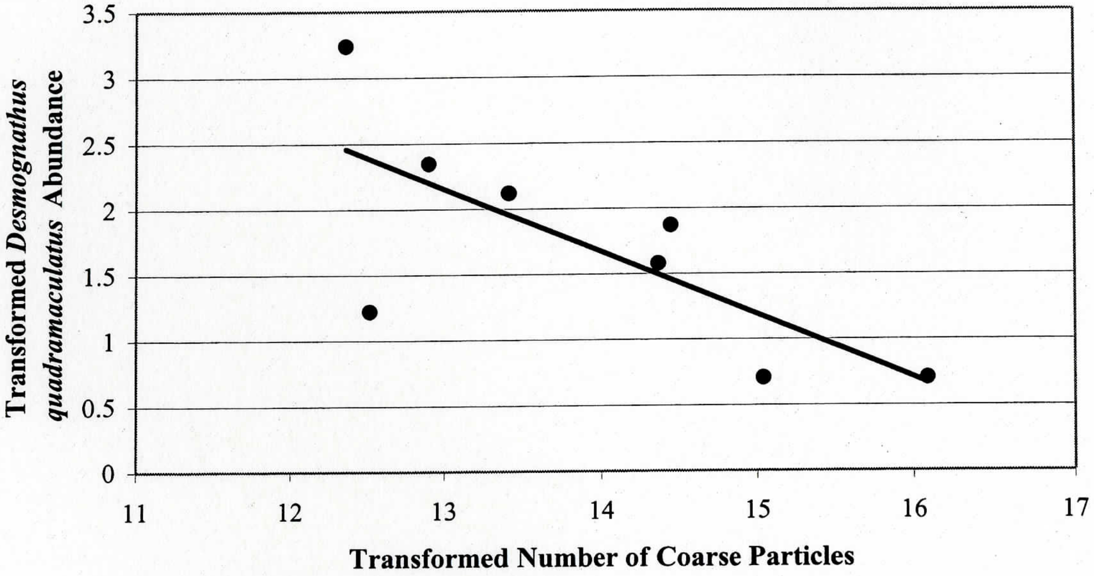

Figure 12. ***Desmognathus quadramaculatus* abundance and coarse gravel.** Particles that were in the 11.3mm, 16mm, and 22.6mm size classes were categorized as coarse gravel for analysis. The regression equation for the line is $y= -0.480x + 8.399$ ($r^2 =0.538$, $p=0.038$).

of 0.01837 ± 0.00056 mg/mg/day (mean ± SE) for the first month. After two months, 18 salamanders were recovered from enclosures (Table 8); of those, four had undergone metamorphosis. Two enclosures contained no salamander larvae and one enclosure contained only one juvenile. The growth study was terminated after the second month.

Table 8. Number of salamander larvae in growth experiment enclosures. Initial number and number of salamander larvae recovered after one and two months from experimental enclosures in Howards Creek.

Enclosure	Sediment Treatment	Larvae Placed in Enclosures	Larvae Recovered After One Month	Larvae Recovered After Two Months
1	None	4	2	1
2	Low	4	2	1[b]
3	High	4	2	1
4	High	4	3	1
5	None	4	3	2
6	Low	4	3	0[c]
7	High	4	4	0
8	Low	4	3	1[b]
9	None	4	4[a]	1
10	None	4	4	1
11	High	4	2	0
12	Low	4	2	1[b]
13	High	4	3	1
14	Low	4	4	2
15	None	4	4	1[b]
	Total	60	45	14[d]

[a] One larvae recovered was dead.
[b] One recently metamorphosed individual was also recovered from the enclosure
[c] One recently metamorphosed individual was recovered from the enclosure.
[d] A total of 18 salamanders were recovered, 14 of which were larvae.

Eurycea wilderae growth in the Howards Creek environment did not vary significantly among sediment treatments (F= 0.15, p=0.87); (Table 9, Figure 13). Growth rates in enclosures showed no significant effect from the number of larvae recovered (F=1.23, p=0.33); (Table 10).

Table 9. Randomized block ANOVA results for larvae growth (mg/mg/day) for *Eurycea wilderae* in response to three levels of sedimentation.

Source	d.f.	MSE	F	P
Model	6	0.00000878	0.54	0.76
Error	8	0.00001632		
Total	14			
Sediment treatment	2	0.00000241	0.15	0.87
Block	4	0.00001196	0.73	0.59

Table 10. ANOVA results for the instantaneous growth (mg/mg/day) of *Eurycea wilderae* larvae in enclosures with different final density.

Source	d.f.	MSE	F	P
Model	2	0.000031	1.23	0.328
Error	12	0.000152		
Total	14	0.000183		

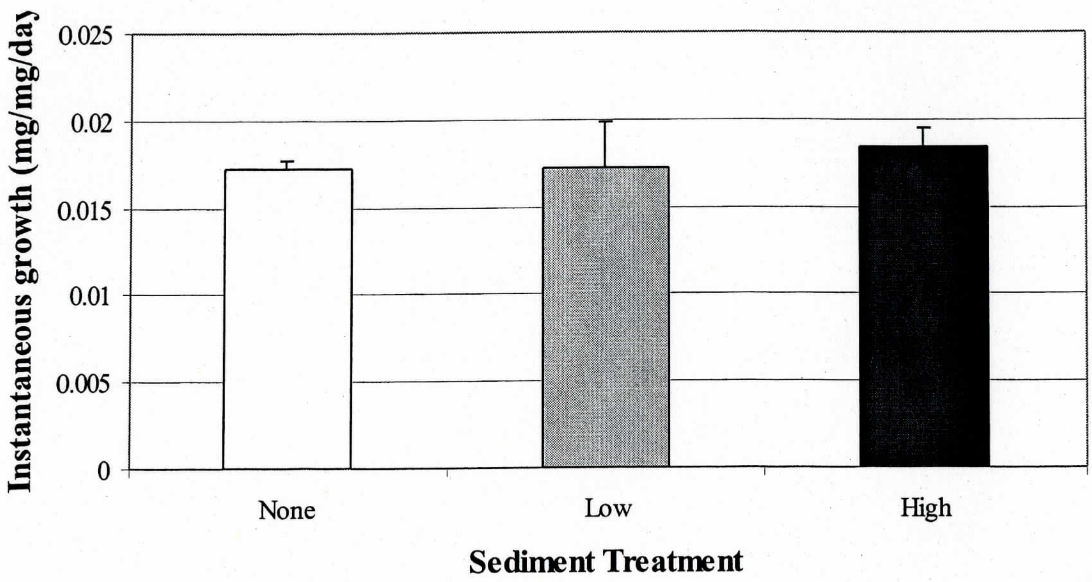

Figure 13. Effects of sediment treatment on instantaneous growth rate of *Eurycea wilderae* after one month. Bars indicate the standard error of the mean.

Predation Experiment

Predation rates of larvae were observed in two trial enclosures, one of which had no added sediment and the other had a high sedimentation level. Observations of the predation of *Eurycea wilderae* larvae by sculpin (*Cottus bairdii*) provided no information on the effect of sediment on salamander survivorship. There only larva that disappeared was in the low silt treatment (Table 11).

Table 11. Effects of sediment on predation by sculpin (*Cottus bairdii*) on Blue Ridge Two lined salamanders (*Eurycea wilderae*) larvae The experiment was conducted between September 11-17, 2004.

Date	Sediment Treatment	Larvae Added	Larvae Recovered
9/11/04	High	2	--
	None	2	--
9/12/04	High	0	2
	None	0	2
9/13/04	High	0	2
	None	0	2
9/14/04	High	0	2
	None	0	2
9/15/04	High	0	2
	None	0	2
9/16/04	High	0	2
	None	1	1
9/17/04	High	0	2
	None	0	2

DISCUSSION

Sediment is a ubiquitous pollutant in streams in the United States (Waters 1995) and was prevalent in many streams that I observed in the northwestern mountains of North Carolina. It was difficult to locate streams without some degree of sediment pollution. Headwater streams currently are protected by stream buffer requirements in the mountains of northwestern North Carolina (R. Woodrow, personal communication; NCFS 1991). However, from my observations of sedimentation conditions in headwater streams, stream buffer requirements and other sediment-control methods are not adequate to prevent sedimentation in these streams. Reductions in sediment inputs in headwaters will help to reduce sediment inputs to larger streams and improve water quality throughout watersheds.

Sedimentation reduced the abundance of salamander larvae in first and second order streams observed in northwestern North Carolina. There were 50% fewer salamander larvae captured in streams affected by sedimentation than streams that had predominantly coarse substrates. The observed abundance trends of plethodontid salamander larvae and sedimentation in the headwater streams of northwestern North Carolina is similar to the negative effects of sedimentation observed in amphibians studied in the Pacific Northwest (Corn and Bury 1989, Kelsey 1995, Welsh and Ollivier 1998). Sedimentation may universally negatively affect amphibian abundance in streams.

Habitat quality, how conducive the environment is to the abundance, survival or reproduction of a species in that habitat (Van Horne 1983), was reduced for salamanders by sedimentation. The abundance pattern of larvae in pools and riffles in sediment-affected and unaffected streams indicated that sedimentation results in a decline in salamander larvae survival in sediment-affected streams. Fewer salamanders were found in pools in sediment-affected streams than the pools of streams not affected by sedimentation. There was not a corresponding increase in abundance in riffles in sediment-affected streams, suggesting that declines are due to mortality, not that larvae are moving out of pool habitats.

Salamander species varied in their response to sedimentation. *Eurycea wilderae* abundance declined in sediment-affected streams but the abundance of *Desmognathus quadramaculatus* was not shown to be affected by sedimentation. Larvae abundance in habitats within streams was altered by sediment-induced changes in substrates, depending on species. The most abundant salamander larvae found in the study streams was *Eurycea wilderae*, which comprised approximately two-thirds of larvae captured from all streams. *E. wilderae* normally prefers pool habitats (Petranka 1998). The reduction in *E. wilderae* abundance in sediment-affected pools indicates a loss of habitat for these salamanders caused by sedimentation. Drift was not measured in this study, yet may be responsible for the observed abundance of *E. wilderae* in sediment-affected streams. Small *E. wilderae* larvae are susceptible to being swept into the water column by the stream current and drifting downstream (Bruce 1986).

The observed declines in salamander abundance also have the potential to indirectly effect macroinvertebrates. Reductions in salamander larvae abundances, which are often the only vertebrate predators in streams too small to support fish populations (Davic 1983), could result also result in changes in the abundance of organisms in lower trophic levels. For example, *Desmograthus quadramaculatus* may be a keystone species in streams without fish (Davic 1983); the reduction or elimination of predation pressure on the competitively dominant macroinvertebrates that *D. quadramaculatus* prey on may result in reductions in the abundance of inferior macroinvertebrate competitor populations, if sedimentation does not directly impact macroinvertebrate abundance.

In my research it was possible to determine relationships between particle size distributions and salamander abundance using the zigzag pebble count. The total abundance of salamander larvae in streams was negatively related to fine particles and pebbles in the streams. However, neither the abundance of *Eurycea wilderae* or *Desmognathus quadramaculatus*, taken separately, was significantly correlated to the amount of fine particles. This result may have been due to a reduction in sample size for comparison when looking at the abundance of each species separately. Fine sediments are associated with sediment pollution (Reid and Dunn 1984, Waters 1995). The relationship between fewer total salamander larvae and the amount of fine sediments in the substrate is probably a result of the disturbance sediment produces by filling interstitial spaces in the substrate. Sediment pollution with particles less than 6.3 mm in diameter is known to result in the loss of interstitial space because they readily embed larger substrate particles (Bunte and Abt 2001a). In addition to fine particles, the prevalence of pebbles also negatively affected salamander larvae abundance. Williams (1978) found that particles that averaged 24.2mm in diameter trapped the most sediment in substrates. Williams' classification of 24.2mm particles as medium gravels is similar to the pebble category of particles between 32 and 64mm used in my research. A possible explanation for the negative relationship between the amount of pebbles in the stream substrate and the abundance of salamander larvae is that pebbles trap fine sediments more than other particle sizes (Williams 1978) and contribute to high levels of fine sediments retained in the streambed. Large particles in streams on the other hand positively affect abundance. Abundance was positively correlated with the amount of small boulders and the amount of bedrock in streams. It appears that streams with coarse substrates are high-quality habitats capable of supporting higher abundances of salamander larvae than streams beds with high amounts of fine sediments in the substrate.

Correlation analyses also indicated species interactions with substrate habitats varies. Different substrate particle sizes often have different influences on organism abundance, depending on the organism (Young et al. 1991). The abundance of *Eurycea wilderae* was not significantly associated with the abundance of any particle size class, though there was a negative relationship between abundance and the amount of fine gravel in the stream. The negative association between the abundance of *E. wilderae* and fine gravel may be due to a loss of interstitial spaces caused by fine gravels in the substrate. Interstitial spaces are often filled by particles $\leq$ 6.3mm in diameter (Bunte and Abt 2001a). Increases in the amount of fine gravels in the substrate appear to reduce the amount of suitable habitat for *E. wilderae*. The abundance of *D. quadramaculatus* larvae in streams, on the other hand, was significantly related to stream gradient and negatively associated only with the amount of coarse gravel in the stream substrates.

High gradient streams with few coarse gravels, though not caused by sediment pollution *per se*, appears to be high-quality habitat for *D. quadramaculatus*.

Differences in relationships of different species with different particle size distributions is probably a reflection of the different sizes of *Eurycea wilderae* and *Desmognathus quadramaculatus* larvae. *D. quadramaculatus* larvae are almost twice as large as *E. wilderae* larvae (Petranka 1998). It may be expected that larger animals would prefer larger cover objects than smaller animals, and that sedimentation affects the available cover for the two amphibians differently. There is evidence that *D. quadramaculatus* prefers substrates with a heterogeneous mixture of particles to homogenous particles (Davic and Orr 1987). Too much coarse gravel in a substrate may reduce habitat heterogeneity. Also, *D. quadramaculatus* and *E. wilderae* have different diets, possibly due to the difference in gape size between the two salamander larvae (Davic 1983). *E. wilderae* were found to eat predominantly chironimid larvae, whereas the majority of prey for *D. quadramaculatus* belonged to large invertebrates in the shredder and collector functional feeding groups (Davic 1983). Prey choice differences between the two species may indicate use of separate niches in streambed habitats that could have produced the observed particle size class and salamander abundance relationships.

Studies that evaluate sedimentation effects on amphibians often rely upon field observations to determine if there is a sedimentation effect on amphibian abundance, but fail to evaluate the mechanism by which sediment pollution affects individuals. Welsh and Ollivier (1998) advocate the use of stream amphibians as bioindicators of stream habitats without evaluating how sediment reduces amphibian abundance. I evaluated two potential mechanisms through which sedimentation may reduce the abundance of salamander larvae in streams but did not identify a basis for the observed effects of sediment on abundance. The absence of an identified mechanism of sedimentation on declines in lotic plethodontid larvae abundance makes the use of larval salamanders as indicator species for sedimentation undesirable.

Declines in larvae abundance suggested that habitat quality was reduced in sediment-affected streams. Resource availability affects the abundance of organisms in a habitat (Brown 1984), and subsequently determines the habitat quality for that organism. I tested how sedimentation affected food availability for *E. wilderae* larvae by measuring the growth of larvae in enclosures. Silt-induced reductions in the availability of larvae to find adequate macroinvertebrate prey would result in lowered growth rates and could have been responsible for reduced numbers of salamander larvae observed in sediment-affected pools. However, growth experiment results did not support this hypothesis. Growth study results indicated that *E. wilderae* salamander larvae were able to find adequate prey for growth regardless of the sedimentation condition of the substrate. Chironimid larvae, which are associated with fine sediments, were observed in salamander enclosures during the growth study; Davic (1983) found that 71% of the diet of *E. wilderae* is composed of chironimid species. The growth study results indicate that sedimentation does not reduce food availability for *E. wilderae* to the point that it would limit larval growth. Sedimentation therefore does not appear to produce a limitation of a key food resource for *E. wilderae*.

Desmognathus quadramaculatus growth in high sedimentation conditions on the other hand remains unknown. *D. quadramaculatus* growth in high sediment substrates was not evaluated due to the long larvae period of the species (Lugthart 1991). *D. quadramaculatus* feeds predominately on macroinvertebrates negatively associated with fine sediments (Davic 1983), unlike chironomids, and may experience declines in growth rates if sediment makes finding prey difficult.

Predation was also evaluated as a potential mechanism for a sediment-induced abundance declines, but experimental results did not suggest that sculpin (*Cottus bairdii*) are heavy predators of *Eurycea wilderae*. Higher-than-usual predation rates hypothetically could have been caused by a lack of sufficient cover from predators in sediment-affected substrates. Yet predation remains a possible mechanism for the reduced abundance of salamander larvae in sediment-affected streams and should be evaluated further. Fish were not observed in all study reaches (Table 12) and no effort was made to identify fish species in study streams. As mentioned previously, small *E. wilderae* larvae are susceptible to being dislodged by water currents and drifting in the water (Bruce 1986). Drifting larvae are especially vulnerable to fish predation (Sih et al. 1992). Sih et al. (1992) found that only 4-8% of lotic *Ambystoma barbouri* larvae that drifted into pools occupied by green sunfish were not eaten. Predation by fish could have been a factor in the observed abundances of salamander larvae in some sediment-affected streams.

Other potential predators of salamander larvae were also not observed in every study stream (Table 12) but may include crayfish and salamanders such as adult *Desmognathus quadramaculatus*. Crayfish are omnivorous but might not be likely candidates as potential predators of salamander larvae because they were not observed in all study streams. Additionally, crayfish diet may vary by the size of the individual--

younger, smaller individuals are typically carnivores of macroinvertebrates whereas larger individuals are typically herbivores (Lorman and Magnuson 1978, Creed 1994). Predators are typically larger than their prey, so it seems unlikely that predation pressure by small, carnivorous crayfish contributed to the observed differences in abundance between sediment-affected and unaffected stream reaches.

Table 12. Presence or absence of potential predators of salamander larvae. If fish were observed in a study stream, the stream was considered to be capable of supporting predaceous fish. Streams considered to lack fish may support fish, but no fish were observed during sampling. The presence or absence of crayfish and *Desmograthus quadramaculatus* adults in streams is based on observations during salamander sampling.

Study Site	Sedimentation Condition	Fish observed?	Crayfish (*Cambarus* spp.) Observed?	Adult *D. quadramaculatus* observed?
Doe Fork tributary	High	No	No	No
Nature Preserve stream	High	No	No	Yes
Sims Creek	High	Yes	Yes	No
Story Branch	High	Yes	No	Yes
Tater Hill Bog tributary	High	Yes	No	No
Boone Fork	Low	Yes	Yes	No
Dixon Creek	Low	No	No	Yes
Green Ridge Branch	Low	No	Yes	No

Adult salamanders are another potential predator of larvae. *D. quadramaculatus* adults are known to prey on both *D. quadramaculatus* and *E. wilderae* larvae (Beachy 1993, Petranka1998). Gut content analysis by Davic (1983) revealed *E. wilderae* larvae in the gut of 17% of *D. quadramaculatus* adults. I observed predation of an *E. wilderae* adult by a *D. quadramaculatus* adult during sampling in Dixon Creek. However, like fish, *D. quadramaculatus* adults were not observed in all streams. Predation effects on salamander larvae abundance in sediment-affected habitats versus unaffected habitats warrant further study.

Other potential causes of the differences in abundance observed between sediment-affected and unaffected study reaches were not evaluated experimentally. Future studies may evaluate the suggestions that sediment results in the loss of oviposition sites or increased mortality of salamander eggs in sediment-affected streams (Kats and Sih 1992, Kelsey 1995). Sedimentation can reduce the dissolved oxygen concentration in the substrate and has been shown to negatively affect the survival of fish eggs (Rinne 2001). *Desmognathus quadramaculatus* and *Eurycea wilderae* females both lay eggs on the undersides of rocks (W. Van Devender, personal communication). *D. quadramaculatus* typically lay eggs under rocks in fast flowing water (Organ 1961, cited in Petranka 1998). *D. quadramaculatus* eggs may be affected by oxygen deprivation from sedimentation in a manner similar to fish eggs. On the other hand, *E. wilderae* females appear to prefer to lay eggs under rocks that are embedded in the substrate (Petranka 1998). In some populations, *E. wilderae* females will even lay eggs in sediments with little water flow or dissolved oxygen availability (Marshal 1996). Sedimentation may therefore not result in the same physiological effect on *E. wilderae* eggs as *D. quadramaculatus* eggs.

Methodology for sedimentation measurement in streams for the purpose of evaluating relationships between habitat conditions and stream organisms varies greatly between studies. Visual estimates of sedimentation in streams have been used to compare amphibian abundance and sedimentation condition of stream substrates (e.g., Corn and Bury 1989). Welsh and Ollivier (1998) used visual estimates of substrate embeddedness in the pools of streams to characterize their sedimentation condition. My research classifies the study streams as sediment-affected or unaffected based on an initial visual evaluation of stream substrates. Subsequent pebble counts indicated that affected streams had much greater amounts of fine sediments ($\leq$ 2mm) than streams visually categorized as unaffected by sedimentation. Visual estimation of sedimentation for gross classification of stream substrates as sediment-affected versus sediment-unaffected therefore seems warranted for quick sediment assessments. However, visual evaluations of stream substrates should only be relied upon for quick evaluations of stream sedimentation conditions.

Visual estimates of sedimentation are not reproducible measurements (Kondolf and Li 1992). Visual evaluations of substrates are by nature subjective and probably should not be used in statistical analysis in future studies, given the availability of simple, reproducible methods of measuring sediment conditions in substrates, such as pebble count techniques. Pebble count methodology for measuring particle size distributions in streambeds is relatively quick and inexpensive, and produces repeatable results (Potayondy and Hardy 1994). Pebble counts can be used to track changes in the amount of fine sediments in a stream substrate or can be used to compare the amounts of fine particles between different reaches (Bevenger and King 1995, Schnackenburg and MacDonald 1998), as was done in this study. The zigzag pebble count is well suited to studies that evaluate sediment and benthic organisms because it incorporates different habitats along an entire study reach as one integrated unit rather than individual habitats or sections of streams (Bevenger and King 1995). This trait is important as sedimentation conditions in streams can vary greatly between localities in streams

(Bunte and Abt 2001a). Large study reaches of 300m are desirable to incorporate the variability in substrate particle sizes and salamander larvae abundance found within streams (Carpenter 1996).

Pebble counts produce particle size distribution results with known confidence intervals, making comparisons between particle sizes and larvae abundance derived from random samples within reaches statistically reliable. The pebble count method used in my research evaluated approximately 400 particles. A sample size of 400 particles results in an accurate description of the particle size distribution; small increases in confidence limits of particle size distributions result only when much greater numbers of particles are collected (Rice and Church 1996). One of the goals of my research was to be able to rapidly assess stream sedimentation, thus the use of 400 particles was warranted. My research indicates that significant relationships between streambed particles and salamander larvae abundance are evident with a particle sample sizes of 400.

Land managers can use regression models such as the ones produced from this study to evaluate the impact of sediment pollution on the abundance of salamanders in streams. For example, the effect of fine sediments on salamander abundance can be modeled with pebble counts. Pebble counts can be used to evaluate the extent of sediment pollution in streams from both unidentified sources and from identifiable sources such as roads, which in turn can be used to model the areal extent of sediment effects on salamander abundance. Furthermore, pebble counts can be used to monitor improvements or declines in habitat quality over time, making them useful tools for evaluating sediment pollution remediation and control efforts.

Williams (1983) found both *Eurycea wilderae* and *Desmognathus quadramaculatus* were abundant in Watauga County. Petranka (1998) considers both *E. wilderae* and *D. quadramaculatus* to be in minimal need of conservation efforts. However, sedimentation in streams in northwestern North Carolina has resulted in 50% reductions in the larvae of these species. Given the prevalence of sediment pollution in headwater streams, sedimentation is likely to eventually have detrimental effects on the population sizes of *E. wilderae* and *D. quadramaculatus*. Future studies should evaluate how a reduction in the abundance of aquatic salamander larvae affects the abundance of salamander adults over time. Sedimentation in streams has the potential to reduce the abundance of terrestrial adults, a potential example of how environmental degradation in the aquatic environment could affect salamander populations in terrestrial ecosystems.

Sedimentation in stream substrates is a persistent pollutant (Waters 1995). Yet land managers may consider sedimentation in the high gradient streams found in mountainous areas to be a temporary problem (L. Stroup, personal communication) because of the potential for high velocity flows to flush fine sediments from the substrate. Yet flushing of fine sediments from substrates often requires flows with great enough velocities to move gravels (Adams and Berschta 1980). Adams and Berschta (1980) also found that high flows did not remove sediments from streambeds but rather tended to flush fines only from particular areas in streams. Furthermore, they found that high flows that flush fines often carry more suspended sediment that settles onto the substrate after a stream returns to baseflow. Flushing of sedimentation from mountain stream substrates alone should probably not be relied upon to maintain habitat quality in streams. Even a small increase in fines can affect organisms in streams (Sutherland et al.

2002). Land managers should concentrate on efforts to minimize sedimentation pollution from its sources because of the persistence of sedimentation and the difficulty and expense in removing sediment from streambeds. Plethodontid salamanders in northwestern North Carolina are likely to benefit from better protection of streams from sedimentation caused by land disturbances.

LITERATURE CITED

Adams, J. N. and R. L. Beschta. 1980. Gravel bed composition in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Science 37: 1514-1521.

Allouche, S. and P. Gaudin. 2001. Effects of avian predation threat, water flow and cover on growth and habitat use by chub, Leuciscus cephalus, in an experimental stream. Oikos 94:481-492.

Barr, G. E. and K. J. Babbitt. 2001. A comparison of two techniques to sample larval stream salamanders. Wildlife Society Bulletin 29: 1238-1242.

Beachy, C. K. 1993. Guild structure in streamside salamander communities: a test for interactions among larval plethodontid salamanders. Journal of Herpetology 27: 465-468

Beasley, G. and P. Kneale. 2002. Reviewing the impact of metals and PAHs on macroinvertebrates in urban watercourses. Progress in Physical Geography 26: 236-270.

Berschta, R. L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon coast range. Water Resources Research 14: 1011-1016.

Bevenger, G. S. and R. M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. U.S. Forest Service Research Paper RM-RP-319.

Blaustein, A. R. 1994. Chicken Little or Nero's Fiddle? A perspective on declining amphibian populations. Herpetologica 50: 85-97.

Bond, N. R. and B. J. Downes. 2003. The independent and interactive effects of fine sediment and flow on benthic invertebrate communities of small upland streams. Freshwater Biology 48: 455-465.

Brown, J. H. 1984. On the relationship between abundance and distribution of species. The American Naturalist 124: 255-279.

Bruce, R. C. 1982. Larval periods and metamorphosis in two species of salamanders of the genus Eurycea. Copeia 1982: 117-127.

Bruce, R. C. 1986. Upstream and downstream movements of Eurycea bislineata and other salamanders in a southern Appalachian stream. Herpetologica 42: 149-155.

Brunskill, G. J., D. M. Rosenberg, N. B. Snow, G. L. Vascotto, and R. Wagemann. 1973. Ecological studies of aquatic systems in Mackenzie-Porcupine drainages in relation to proposed pipeline and highway developments, volume 1. Northern Pipelines Task Force, Environmental-Social Committee. Northern Oil Development Report 73-40, Winnipeg, Manitoba.

Bunte, K. and S. R. Abt. 2001a. Sampling surface and subsurface particle-size distributions in wadable gravel- and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. U.S. Forest Service General Technical Report RMRS-GTR-74.

Bunte, K. and S. R. Abt. 2001b. Sampling frame for improving pebble count accuracy in coarse gravel bed streams. Journal of the American Water Resources Association 37: 1001-1014.

Carpenter, S. R. 1996. Microcosm experiments have limited relevance for community and ecosystem ecology. Ecology 77:677-680.

Cole, G. A. 1983. Textbook of limnology, 3rd edition. Waveland Press, Prospect Heights, Illinois.

Conant, R. and J. T. Collins. 1998. A field guide to the reptiles and amphibians of eastern and central North America, 3rd ed. Houghton Mifflin, Boston, Massachusetts.

Cooper, C. M. 1987. Benthos in Bear Creek, Mississippi: effects of habitat variation and agricultural sediments. Journal of Freshwater Ecology 4: 101-1 13.

Corn, P. S. and R. B. Bury. 1989. Logging in Western Oregon: response of headwater habitats and stream amphibians. Forest Ecology and Management 29: 39-57.

Creed, R. P. 1994. Direct and indirect effects of crayfish grazing in a stream community. Ecology 75: 2091-2103.

Davic, R. D. 1983. An investigation of salamander guild predation in a North Carolina stream: an experimental approach. Doctoral dissertation. Kent State University, Kent, Ohio.

Davic, R. D. and L. P. Orr. 1987. The relationship between rock density and salamander density in a mountain stream. Herpetologica 43: 357-361.

Dingman, S. L. 1994. Physical Hydrology. Prentice Hall, Upper Saddle River, New Jersey.

Gillespie, G. R. 2002. Impacts of sediment loads, tadpole density, and food type on the growth and development of tadpoles of the spotted tree frog Litoria spenceri: an in-stream experiment. Biological Conservation 106: 141-150.

Hall, R. J. and R. M. Prouty. 1985. Pesticide and PCB residues in the Neuse River Waterdog, Necturus lewisi. Brimleyana 10: 107-109.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, L. M. Decker, S. V. Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18: 3-12.

Hooke, J. M. 1979. An analysis of the processes of riverbank erosion. Journal of Hydrology 42: 39-62.

Jaeger, R. G. and R. F. Inger. 1994. Quadrat Sampling. In: W. R. Heyer, M. A. Donnelly, R. W. McDiarmid, L. C. Hayeck, and M. S. Foster (eds). Measuring and monitoring biological diversity: sampling methods for amphibians. Smithsonian Institution Press, Washington, DC.

Kats, L. B. and A. Sih. 1992. Oviposition site selection and avoidance of fish by streamside salamanders (*Ambystoma barbouri*). Copeia 2: 468-473.

Kedzierski, W. M. and L. A. Smock. 2001. Effects of logging on macroinvertebrate production in a sand-bottomed, low-gradient stream. Freshwater Biology 46: 821-833.

Kelsey, K. A. 1995. Responses of headwater stream amphibians to forest practices in Western Washington. Doctoral dissertation. University of Washington, Seattle, Washington.

Kondolf, G. M. 1997. Application of the pebble count: notes on purpose, method, and variants. Journal of the American Water Resources Association 33: 79-87.

Kondolf, G. M. and S. Li. 1992. The pebble count technique for quantifying surface bed material size in instream flow studies. Rivers 3: 80-87.

Lemly, A. D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. Hydrobiologia 87: 229-245.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1979. Biological evaluation of non-point source pollutants in North Carolina streams and rivers. North Carolina Department of Natural Resources and Development, Biological Series 104, Raleigh.

Lisle, T. and S. Hilton. 1992. The volume of fine sediment in pools: an index of sediment supply in gravel-bed streams. Water Resources Bulletin 28: 371-383.

Lisle, T. E., F. Iseya, and H. Ikeda. 1993. Response of a channel with alternate bars to a decrease in supply of mixed-size bed load: a flume experiment. Water Resources Research 29: 3623-3629.

Lloyd, D. S. 1987. Turbidity as a water quality standard for salmonids habitats in Alaska. North American Journal of Fisheries Management 7: 34-45.

Lorman, J. G. and J. J. Magnuson. 1978. The role of crayfishes in aquatic ecosystems. Fisheries 3: 8-10.

Lugthart, G. J. 1991. Macrofaunal community structure and production in disturbed and undisturbed headwater streams. Doctoral dissertation. University of Georgia, Athens, Georgia.

Marshal, J. L. 1996. Eurycea cirrigera (southern two-lined salamander). Nest site. Herpetelogical Review 27: 75-76.

Miller G.T. 1998. Living in the environment, 10th edition. Wadsworth Publishing Company, Belmont, California.

Minshall, G. W. 1984. Aquatic insect-substratum relationships. In: V. H. Resh and D. M. Rosenberg (eds). Ecology of aquatic insects. Praeger Publishers, New York, New York.

NCFS. 1991. Streamside Management Zones. NorthCarolina Division of Forest Resources BMP-2.

Pechmann, J. H. K. and H. M. Wilbur. 1994. Putting declining amphibian populations in perspective: natural fluctuations and human impacts. Herpetologica 50: 65-84.

Peckarsky, B. L. 1985. Do predaceous stoneflies and siltation affect the structure of stream insect communities colonizing enclosures? Canadian Journal of Zoology 63: 1519-1530.

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press. Washington, DC.

Potyondy, J. P. and T. Hardy. 1994. Use of pebble counts to evaluate fine sediment increase in stream channels. Water Resources Bulletin 30: 509-520.

Power, M. E. 1990. Resource enhancement by indirect effects of grazers: armored catfish, algae, and sediment. Ecology 71: 897-904.

Reid, L. M. and T. Dunn. 1984. Sediment production from forest road surfaces. Water Resources Research 20: 1753: 1761.

Rempel, L. L., J. S. Richardson, and M. C. Healy. 2000. Macroinvertebrate community structure along gradients of hydraulic and sedimentary conditions in a large gravel-bed river. Freshwater Biology 45: 57-73.

Rice, S. and M. Church. 1996. Sampling surficial fluvial gravels: the precision of size distribution percentile estimates. Journal of Sedimentary Research 66: 654-665.

Rinne, J. N. 2001. Effects of substrate composition on Apache Trout fry emergence. Journal of Freshwater Ecology 16: 355-365.

Rosgen, D. L. 1994. A classification of natural rivers. Cataena 21:161-199.

Schnackenberg, E. S. and L. H. MacDonald. 1998. Detecting cumulative effects on headwater streams in the Routt National Forest, Colorado. Journal of the American Water Resources Association. 34: 1163-1177.

Shaw, E. A. and J. S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (Oncorhynchus mykiss) growth and survival. Canadian Journal of Fisheries and Aquatic Science 58: 2213-2221.

Sih, A., L. B. Kats, and R. D. Moore. 1992. Effects of predatory sunfish on the density, drift, and refuge use of stream salamander larvae. Ecology 73: 1418-1430.

Sutherland, A. B., J.L. Meyer, and E. P. Gardiner. 2002. Effects of land cover on sediment regime and fish assemblage structure in four southern Appalachian streams. Freshwater Biology 47: 1791-1805.

Sweka, J. A. and K. J. Hartman. 2001. Influence of turbidity on Brook Trout reactive distance and foraging success. Transactions of the American Fisheries Society 130: 138-146.

Trimble , S. W. 1994. Erosional effects of cattle on streambanks in Tennessee, U.S.A. Earth Surface Processes and Landforms 19: 451-464.

USEPA. 1990. The quality of our nation's water: a summary of the 1988 National Water Quality Inventory. U.S. Environmental Protection Agency, EPA Report 40/4-90-005, Washington, DC.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47: 893-901.

Waring, R. H. and W.H. Schlesinger. 1985. Forest ecosystems: concepts and management. Academic Press, Orlando, Florida.

Waters, T. F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society Monograph 7. American Fisheries Society, Bethesda, Maryland.

Wellman, J. C., D. L. Combs, and S. B. Cook. 2000. Long-term impacts of bridge and culvert construction or replacement on fish communities and sediment characteristics of streams. Journal of Freshwater Ecology. 15: 317-328.

Welsh, H. H. and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. Ecological Applications 8: 1118-1132.

Welsh, H. H., L. M. Ollivier, and D. R. Hankin. 1997. A habitat-based design for sampling and monitoring stream amphibians with an illustration from Redwood National Park. Northwestern Naturalist 78: 1-16.

Williams, D. D. 1978. Substrate size selection by stream invertebrates and the influence of sand. Limnology and Oceanography 23: 1030-1033.

Williams, J. K. 1983. A survey of the amphibians and reptiles of Watauga County, North Carolina. Master's thesis. Appalachian State University, Boone, North Carolina.

Young, M. K., W. A. Jubert, and T. A. Wesche. 1991. Selection of measures of substrate composition to estimate survival to emergence of salmonids and to detect changes in stream substrates. American Journal of Fisheries Management 11: 339-346.

Zar, J. H. 1999. Biostatistical Analysis, 4th ed. Pentice Hall, Upper Saddle River, New Jersey.

Zug, G. R., L. J. Vitt, and J. P. Caldwell. 2001. Herpetology: an introductory biology of amphibians and reptiles, 2nd ed. Academic Press, San Diego, California.

VITA

Daniel R. Dietrich was born in Tiffin, Ohio, on April 1, 1977. His parents are Walter and Barbara Dietrich. Daniel attended Pine Forest High School, in Fayetteville, North Carolina, and graduated in 1995. Daniel then attended the University of North Carolina at Charlotte, in Charlotte, North Carolina. He earned his Bachelor of Science in Biology with a concentration in Ecology/Environmental Science in 1999. Daniel married Susan Balfour in 2002. Also in 2002, Daniel began working towards a Master's Degree in Biology at Appalachian State University; he was awarded his Master of Science in May 2005. Daniel plans to pursue a career in environmental science and education in Richmond, Virginia.