Recovery From and Effects of a Catastrophic Flood and Debris Flow on the Brook Trout (*Salvelinus fontinalis*) Population and Instream Habitat of the Staunton River, Shenandoah National Park, Virginia.

by

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(Abstract)

The Staunton River is a high gradient, second order stream approximately 6 km in length located on the eastern slope of the Blue Ridge Mountains in Shenandoah National Park, VA. In June 1995, a catastrophic flood and debris flow altered the instream habitat and *Salvelinus fontinalis* population of the Staunton River. The debris flow scoured the streambed, deposited new substrate materials, removed trees from the riparian zone, and eliminated fish from a 1.9km section of the stream. By June 1998, both young-of-year (YOY) and age 1+ *S. fontinalis* had recolonized the debris flow affected area. The event provided a rare opportunity to examine recovery of the *S. fontinalis* population and instream habitat in addition to addressing potential effects of the debris flow on movement, activity, and growth of fish in the debris flow affected and unaffected areas of the stream.

Post-recolonization movement and activity were monitored using two-way fish traps (weirs), mark-recapture techniques, and radio telemetry. The weirs failed to produce any movement data. Most fish (91%) in the mark-recapture study had range sizes less than 100m, however biases common to mark-recapture study designs (low recapture rate, flawed logic, etc.) hampered interpretation of results. For example, subsequent recapture of individually marked fish indicated that as many as 54% of marked fish confirmed to have been alive at the time of a recapture session were not recaptured.

Radio telemetry provided information on *S. fontinalis* movement and activity at seasonal and diel scales during summer and fall. Differences in movement and activity between the debris flow affected and unaffected areas were minimal when compared to seasonal variations. During summer, range sizes were near 0m and crepuscular activity patterns were observed. During the fall range size increased and diel activity was concentrated in the mid-afternoon with a much higher peak than during summer.

Basin-wide visual estimation technique (BVET) fish population surveys performed each spring and fall from 1993 – 1999 provided pre- and post-event fish population abundance and density estimates. Post-event fish growth in the debris flow affected and unaffected areas was monitored using mark-recapture techniques. Abundance and density of both YOY and age 1+ *S. fontinalis* exceeded pre-event levels within 2-3 years. Growth of YOY and age 1+ fish was significantly greater in the debris flow affected area until spring 1999. Population density appeared to have a strong negative influence on growth. The observed changes in fish growth and differences in fish size associated with population

density would be of minimal importance to the typical angler but may suggest a mechanism by which *S. fontinalis* populations can quickly recover from catastrophic events.

BVET habitat surveys provided information on total stream area, number of pools and riffles, pool and riffle surface area and depth, substrate composition, and large woody debris (LWD) before (1993), immediately following (1995), and four years post-event (1999). Immediately following the debris flow, the stream channel was highly disordered which resulted in an increase in the total number of habitat units and a decrease in average habitat unit surface area, total stream area, and average depth when compared with pre-event conditions. In addition, substrate composition had shifted from small to large diameter particles and LWD loading had increased in both debris flow affected and unaffected areas. Four years after the event, the total number of habitat units, average habitat unit surface area, total stream area, and average depth had all returned to near pre-debris flow levels and substrate composition had begun to shift towards smaller particle sizes. Changes in LWD loading from 1995-1999 reflected changes in the riparian zone following the debris flow. In the unaffected area, where riparian trees remained intact, LWD loading increased, whereas in the debris flow affected area, where riparian trees were eliminated, LWD loading decreased.

For the most part the effects of the debris flow, although immediately dramatic, were in the long term minimal. The debris flow affected area was recolonized rapidly and abundance and density quickly rebounded past pre-event levels. Differences in fish growth between the affected and unaffected area were short lived. Any effect the debris flow affected area may have had on movement or activity was minimal when compared with seasonal variations. Most habitat characteristics reverted to near pre-event levels just four years after the flood and debris flow. Although a number of factors will influence recovery time from such events, these results indicate that immediate management action, such as stocking or habitat modifications, are not necessary in all cases.

Acknowledgments

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Introduction

Debris flows occur when rocks, soil, water, and trees from landslides combine with floodwaters and progress as a slurry down a stream channel at up to 10m/s (Hack and Goodlet 1960, Swanson et al. 1998). These events are rare on any given stream, with recurrence times >50 years in the Pacific Northwest (Swanson et al. 1987) and >100 years in the Central Appalachian region (Hack and Goodlet 1960). Debris flows have the potential to remove trees from the immediate riparian area, scour the stream channel to bedrock, and completely eliminate instream biota. Impacts of debris flows on stream biota, instream habitat, and riparian areas often appear devastating, however pre-event data and knowledge of the timing and severity of the event are rarely of sufficient detail to allow for a proper assessment of immediate effects on or recovery of fish populations or habitat (Lamberti et al. 1991, Sousa 1984).

In the past, the speed with which stream resident salmonids could recolonize defaunated high gradient streams had been regarded as inadequate (Moore et al. 1985) and furthermore, they were generally regarded as sedentary, occupying the same pool or small reach (<30m) of stream for most of their adult lives (Gowan et al. 1994). The majority of studies that indicated stream resident salmonids were sedentary were mark-recapture studies that suffered from two major flaws: 1) recapture effort was focussed in areas where fish were marked, and 2) notoriously low recapture rates. These studies recently came under scrutiny (Gowan et al. 1994, Gowan and Fausch 1996), and it was recommended that alternative techniques such as radio telemetry and two-way fish traps (weirs) be used to examine stream resident salmonid movement and its causes. The use of weirs (Gowan and Fausch 1996) and radio telemetry (Young 1994) in addition to recent studies that indicate that salmonids typically recolonize debris flow affected areas within 1-3 years (Lamberti et al. 1991, Propst and Stefferud 1997, Swanson et al. 1998), suggest that they may be more mobile than previously believed. Events such as debris flows provide rare opportunities to examine recolonization of affected areas and to examine post-recolonization movement and activity in affected and unaffected sections of a stream as part of a 'natural experiment' (Lamberti et al. 1991).

Recolonization can provide one measure of population recovery in debris flow affected streams, however a population would not likely be considered fully recovered until pre-event abundance and density levels were reached. Salmonids not only recolonize debris flow affected areas within 1-3 years, but also typically return to pre-event densities within that same time period (Lamberti et al. 1991, Propst and Stefferud 1997, Swanson et al. 1998). Debris flows can also increase primary production by opening the tree canopy and exposing the channel to direct sunlight, potentially increasing the food supply for salmonids (Keith et al. 1998, Murphy et al. 1986). Fish that immediately recolonize debris flow affected areas experience low population density and increased production and can displayed growth rates greater than those of fish in unaffected areas (Lennon 1961, Lamberti et al. 1991, Swanson et al. 1998). When post-event habitat conditions are suitable, fish abundance and density can actually rebound beyond pre-event levels (Lamberti et al. 1991, Thorpe 1994).

Extreme floods and debris flows can drastically alter the appearance of both the stream channel and riparian zone, however lack of pre-event data often precludes any meaningful assessment of impacts on instream habitat (Lamberti et al. 1991, Dolloff et al. 1994). In some cases changes in habitat can create unsuitable habitat conditions for fish species occupying impacted areas (Elwood and Waters 1969), although this appears to be rare in high gradient streams with rocky substrates (Hack and Goodlet 1960, Lamberti et al. 1991, Swanson et al. 1998).

The high gradient headwater streams located in the Blue Ridge Mountains of the Shenandoah National Park (SNP), VA have been subject to severe scouring floods in the past (Lennon 1961). In June 1995, rains in the Shenandoah National Park saturated soils over a five-day period, culminating with rainfalls of more than 20cm (8 inches) over the final 24-hours of the period (Karish et al. 1997). The result on the Staunton River, SNP, VA, was a massive stream-wide flood, coupled with a catastrophic debris flow along the lower 1.9km of the stream to its confluence with the Rapidan River (Figure 1).

The immediate effects of the debris flow were dramatic and included scouring of the stream channel and deposition of new substrates, elimination of trees from the immediate riparian area, piling of large woody debris (LWD) in massive stacks on the stream banks, and complete extirpation of fish species from the debris flow affected area (US Forest Service CATT, unpublished data). The event provided the opportunity to examine recolonization of the debris flow affected area by *Salvelinus fontinalis* and to address potential effects of conditions created by the flood and debris flow on movement, activity, and growth of *S. fontinalis* in the affected and unaffected areas of the stream. The existence of pre-event data (Newman 1996) provided a rare opportunity to examine recovery of the *S. fontinalis* population and instream habitat from the flood and debris flow.

The following three chapters are split between three topics, all of which revolve around the central themes of recovery from and effects of the June 1995 debris flow on the *S. fontinalis* population and instream habitat of the Staunton River. The first chapter focuses on fish movement with regards to both recolonization of the debris flow affected area and post-recolonization movements and activity in debris flow affected and unaffected areas. The focus of the second chapter is recovery of the *S. fontinalis* population from the debris flow with respect to abundance and density, and post-event fish growth in debris flow affected and unaffected areas of the stream. The third chapter documents changes in instream habitat characteristics both immediately and four years after the debris flow. Together these three chapters provide an overview of how *S. fontinalis* and instream habitat respond to processes that shape high gradient streams.

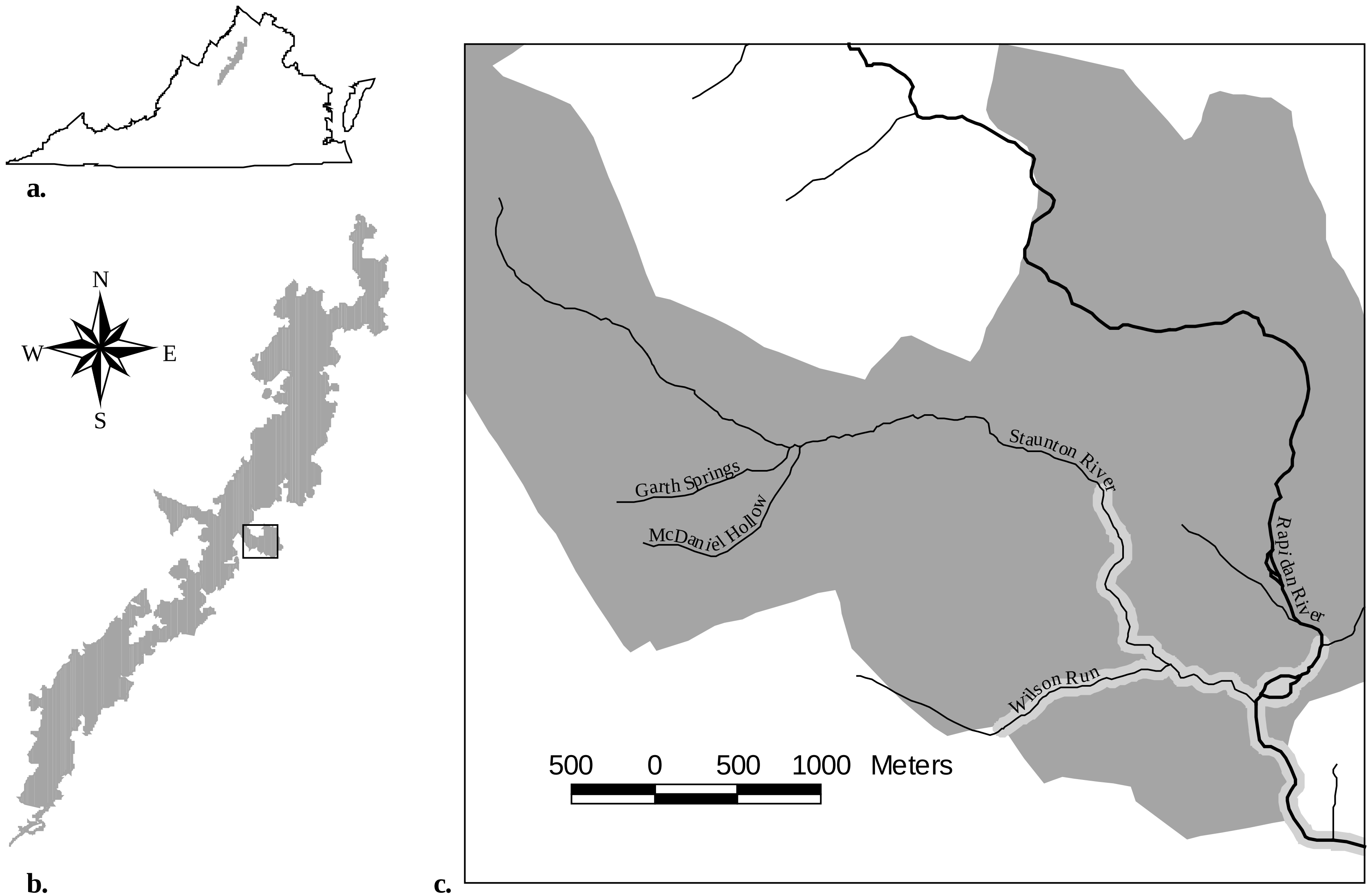

Figure 1. Location of a. Shenandoah National Park (SNP) in Virginia, b. SNP with study site inset in box, c. Rapidan River, Staunton River and tributaries. Darkly shaded areas indicate SNP. Lightly shaded areas indicated debris flow affected areas.

Study Site

The Staunton River is a second-order stream than flows east from an elevation of 975 meters through the central district of SNP to its confluence with the Rapidan River (Figure 1). The stream is approximately 6.3km long with a 10% gradient and an average width of 3.5m. The channel consists of pools separated by step-pool cascades, small (<2 m) waterfalls, and bedrock slides. For most of its length the Staunton River contains mainly two species of fish, *S. fontinalis* and *Rhinichthys atratulus*. *Anguilla rostrata* are found throughout the stream at very low densities and a warmwater fish assemblage occupies the Staunton from its confluence with the Rapidan River to the base of a steep bedrock cascade approximately 200m upstream of the confluence. Most of the Staunton River watershed was cleared before it became part of SNP in the 1930's. The watershed is now completely forested, with chestnut oak (*Quercus prinus*) forests on the higher slopes and tulip poplar (*Liriodendron tulipifera*) as the dominant species nearer to the stream (Karish et al. 1997). The stream is considered to be moderately acid sensitive (Newman 1995) with ANC ranging from 50-100 μeq/L and pH typically ranging from 6.3-7.0.

On June 22, 1995 moderate rain began falling in SNP and culminated with heavy rains on June 27 and June 28. Measured rainfall amounts in some areas of SNP exceeded 20cm (8 inches) on June 27 alone. Graves Mill, a small community just southeast of the Staunton River, may have received as much as 60cm (24 inches) of rain during the five day event. More than 800 people in the vicinity were displaced from their homes, 8 deaths were attributed to flooding, and 2000 homes were damaged. An excellent account of the storm that produced the heavy rains and its immediate effects on SNP can be found in Karish et al. (1997).

The results of the extraordinarily heavy rain in the Staunton River watershed included extensive flooding, landslides, and a massive debris flow along the lower third of the stream and in Wilson Run (Figure 1). Debris flows also occurred on the Rapidan River near its confluence with the Staunton River. The debris flows on the Staunton River and Wilson Run removed most trees from the riparian area and left them in massive log piles on the stream banks, completely scoured the streambed and deposited new substrate materials, and eliminated all fish from the debris flow affected area. *S. fontinalis* and *R. atratulus* populations remained intact, although at decreased densities, in the debris flow unaffected area of the stream.

Chapter One:

Seasonal and Diel Movements and Activity of *S. fontinalis* in Debris Flow Affected and Unaffected Areas of the Staunton River, Shenandoah National Park, VA.

Introduction

The *S. fontinalis* population of the Staunton River has been the subject of study since 1993. A flood and debris flow completely eliminated fish from the lower 1.9 km of the stream in June 1995 (Figure 1). The existence of pre-event population data provided a rare opportunity to examine recolonization of the debris flow affected area. Location and size of the stream and recent advances in technology allowed for use of weirs and radio telemetry in addition to mark-recapture techniques to examine post-recolonization movement in the debris flow affected and unaffected areas.

Recolonization studies following natural catastrophic events are rare for *S. fontinalis*. Most of the existing salmonid recolonization literature indicates that depending on the timing and severity of the event, location of a source population, and pre-flood condition of the population, it takes one to four years to recolonize impacted areas (Detenbeck et al.1992, Lamberti et al. 1991, Propst and Stefferud 1997, Swanson et al. 1998). *S. fontinalis* have been shown to recolonize experimentally defaunated areas within a year (Phinney 1975), however Moore et al. (1985) indicated that recolonization rate following defaunation was not satisfactory in a high gradient situation. These man-made defaunations do not mimic the kind of instream and riparian habitat modifications produced by natural disturbances, and typically do not encompass long stream reaches (>1km) or long periods of time (>2yrs).

Diel movement and activity studies also have received little attention in the *S. fontinalis* literature. Although salmonids are generally considered to be diurnal with additional peaks in activity near dawn and dusk, nocturnal and crepuscular activity patterns have also been observed, with patterns differing between species or within a single species between seasons (Alanara and Brannas 1997, Young et al. 1997, Bunnell et al. 1998). These studies indicated that seasonal changes in water temperature and prey density resulted in changes in feeding strategies, movement, and activity levels within and between species. Bridges (1972) found that during summer *S. fontinalis* exhibited a diurnal activity pattern with most movement consisting of short foraging trips.

The majority of past *S. fontinalis* movement studies have focussed on seasonal movements. Both mark-recapture and weir studies have found little *S. fontinalis* movement during summer months (Shetter 1937, Bjornn 1971, Bridges 1972, Manion 1977, LeClerc and Power 1980, Bryan 1993). However other studies have indicated that during summer *S. fontinalis* are capable of long distance movements (Stefanich 1952, Gowan and Fausch 1996), and that populations can be “relatively unstable” (Shetter and Hazzard 1937) and display “substantial” movement (Gowan and Fausch 1996). During the fall (spawning season) populations tend to become more mobile (Shetter 1937, Bjornn 1971, Bridges 1972, Manion 1977, Bryan 1993) and may move downstream to overwintering habitat following spawning season (Chisholm et al. 1987).

There is little consensus among studies as to factors other than spawning that cause *S. fontinalis* to become mobile. Other possible explanations include fluctuations in water temperature and flow (Manion 1977, Moore et al. 1981, Chisholm et al. 1987), genetic predisposition (Van Offelen et al. 1993), seeking refugia from episodic acidification (Baker et al. 1996), physical condition (Bryan 1993, Gowan and Fausch 1996), and patchy habitat distribution (Riley et al. 1992). Highly variable environments are the norm in streams occupied by *S. fontinalis* and movement allows fish to seek refugia and recolonize defaunated areas (Lamberti et al. 1991) and promotes the long-term persistence of populations (Fausch and Young 1995, Reeves et al. 1995).

Movement of *S. fontinalis* has been the subject of several studies since the 1930's. The majority of studies before the early 1990's concluded that adult stream resident *S. fontinalis* are generally sedentary, occupying the same pool or small reach of stream for most of their adult lives (Stefanich 1952, Gerking 1959, Shetter 1968, LeClerc and Power 1980, Moore et al. 1981, Whitworth and Strange 1983, Schrader 1989, Bryan 1993). Long distance movements that were detected during these studies were considered atypical and spawning movements were not considered part of the normal movements of fish (Gerking 1959). The notion of restricted movement became commonplace and influenced everything from fish production estimates, population models, habitat structure installation, and stocking decisions (Gowan et al. 1994), to identifying evolutionarily significant units (Fausch and Young 1995).

The notion that stream resident fish are sedentary has recently come under scrutiny (Gowan et al. 1994, Gowan and Fausch 1996). The phrase 'restricted movement paradigm' (RMP) was coined to describe the predominant view of adult stream resident fish movement. Review of the literature revealed that most information of movements of *S. fontinalis* was derived from mark-recapture studies (Gowan et al. 1994). These studies had two major flaws: 1) recapture efforts were focused mainly in the area where fish were originally marked, 2) movement information was based on a small number of recaptures. Gowan et al. (1994) and Young et al. (1997) recommended that methods such as radio telemetry and two-way fish traps (weirs) be employed to re-examine movement at differing spatial and temporal scales.

The objectives of this study were to 1) examine *S. fontinalis* recolonization of the debris flow affected area of the Staunton River, 2) examine post-recolonization movement and activity using weirs, mark-recapture, and radio telemetry, and 3) to investigate potential causes of movement. Recolonization was examined with diver counts. Post-recolonization movements were investigated using weirs, mark-recapture, and radio telemetry. Activity was examined using radio telemetry. Both activity and movement were monitored seasonally and daily. Where possible, the effect of temperature, flow, and physical condition on movement were investigated. Use of several methods allowed for a qualitative comparison of results and a discussion of the effectiveness of each method for studying post-recolonization movement. These results also provide the basis for a discussion of the RMP.

Methods

Recolonization

The distribution of *S. fontinalis* prior to the June 1995 debris flow was known from a previous study by Newman (1996), who used the basinwide visual estimation technique (BVET) to sample about 6km of the Staunton River each spring and fall from June 1993 through June 1995. During the BVET surveys every 5th pool and every 10th riffle was snorkeled, fish were counted, and the unit number recorded (Dolloff et al. 1993). An approximation of the distribution of *S. fontinalis* could be obtained from these data. Post-debris flow BVET surveys were performed in October 1995, May 1996, October 1996, June 1997, October 1997, May 1998, October 1998, June 1999, and October 1999. Recolonization patterns by both young of the year (YOY) and age 1+ *S. fontinalis* were described from the snorkeling data. Throughout this thesis, age 1+ refers to all fish older than YOY. Data from the surveys were used to construct maps showing patterns in the recolonization of the debris flow affected area.

Weirs

Weirs consisted of an upstream trap, a downstream trap, and wings (Figure I.1a). Wings served to block the stream and guide fish towards the traps, which captured and held fish until they could be processed and released. Wings consisted of 0.64 cm plastic mesh, 1.22 m in height, mounted on triangular aluminum braces, 0.6 m in height. The lower third of the mesh was laid flat against the stream bed and rocks were placed on top of it to prevent fish from moving under the wing (Figure I.1b). The upper two-thirds of the mesh were attached to the aluminum braces using a combination of screws and thin metal strips. Wire ties and screws were used to hold the wings tightly against the traps.

Each trap was a 60 cm x 60 cm x 60 cm box constructed of 0.64 cm plastic mesh mounted on aluminum frame with a removable lid (Figure I.1c). The entrance of the trap consisted of a 0.64 cm mesh cone attached with a hose clamp to a 15 cm clear plastic tube. The mesh cone tapered from 60 cm to 6.4 cm in diameter over the 38 cm from the front of the trap to the beginning of the 6.4 cm diameter plastic tube. The total distance from the front of the trap to the end of the clear plastic tube was 53 cm. The taper of the cone was offset from center so that the middle of the plastic tube was 15 cm from the bottom of the trap. The entrance funnel was kept from collapsing by attaching it to the back corners of the traps with small bungee cords or strings. The strings were unhooked from the back of the trap and the cone was collapsed against the front of the trap to prevent fish from escaping when traps were checked. Rocks were placed inside the traps to provide fish with cover.

One weir was installed in Barbour's Creek, Craig County, VA from February to December 1999 to test trap durability and fish escapement. The test site was chosen based on similarities in stream discharge, gradient, and width to the Staunton River. Escapement trials were conducted on 13 days

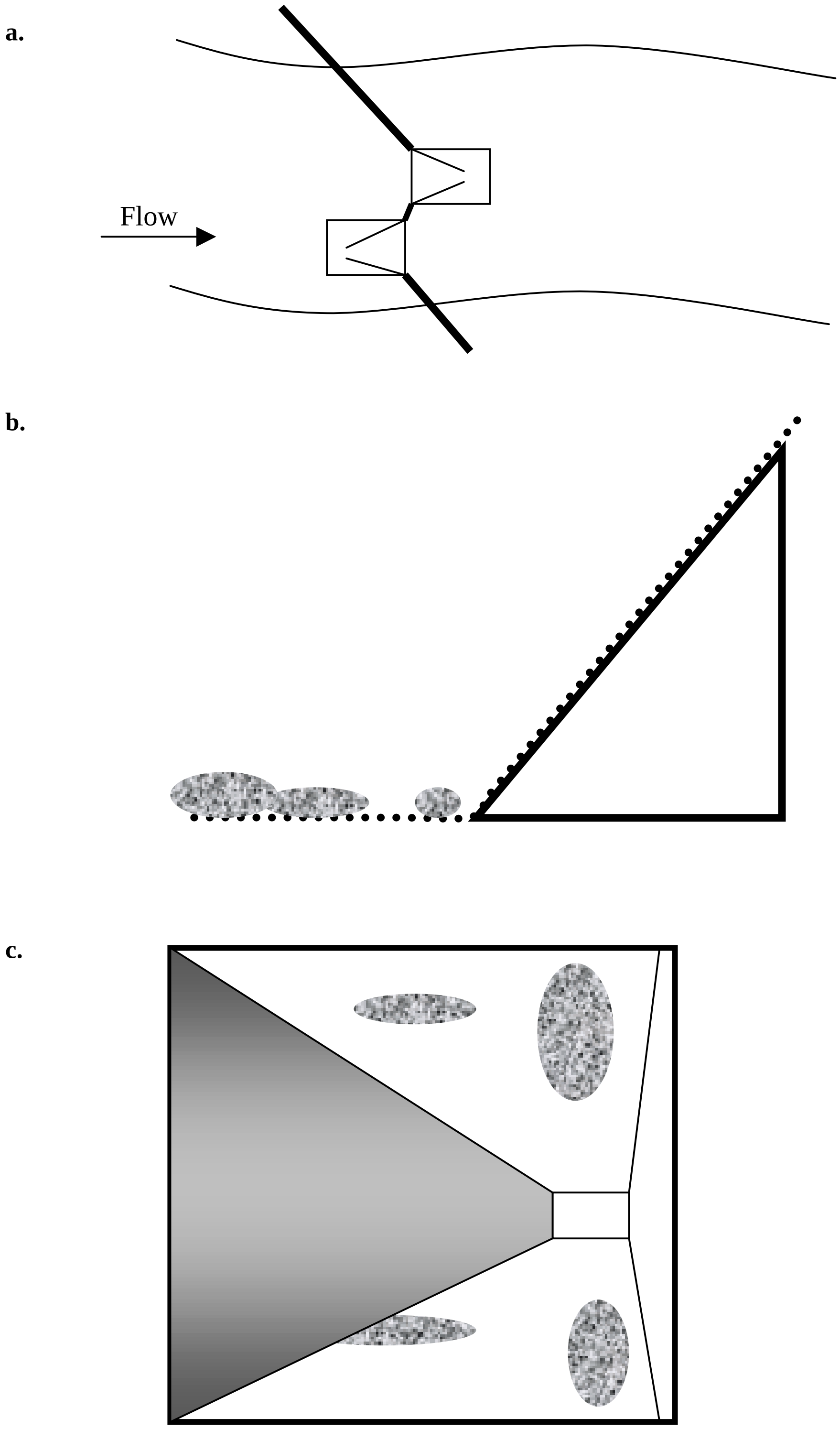

Figure I.1. a. Typical weir configuration. b. Side view of with with mesh attached to aluminum brace and rocks in place to secure mesh to stream bed. c. Top view of trap, grey area represents mesh cone.

during March and April 1999. Fish were captured by electrofishing, given a small fin clip, and placed in the traps. Five to eight fish per day were placed in each trap. The traps were checked the following day to see if any marked fish had escaped or if unmarked fish had entered the trap. Any marked fish that had not escaped were released away from the weir and new fish were placed in the traps for the next trial. A total of 140 fish were used in the escapement trials, ranging from 64 mm-230 mm (mean 114 mm) in total length.

Five weirs were installed in the Staunton River in September 1999. The weirs were placed at approximately 500 m intervals, with two in the debris flow affected area, two in the unaffected area, and one on the border between the affected and unaffected areas of the stream (Figure I.2). The traps were checked once per day. Upon capture in a trap fish were weighed, measured, checked for previous marks, and given a passive integrated transponder (PIT) tag if no old marks were found. Fish <55 mm were marked with fin clips. The weirs were removed from the Staunton River in November 1999.

Weir Data Analysis

Logistic regression was used to estimate daily escapement rates from the upstream and downstream traps and to test for differential escapement between the traps. Total length was included as a covariate in the model. Catch data may be adjusted for escapement rates from upstream and downstream traps using logistic regression results (Gowan and Fausch 1996).

Mark-Recapture

The mark-recapture area spanned a continuous 965 m reach of the Staunton River, with 575 m in the debris flow affected area and 390 m in the unaffected area of the stream (Figure I.3). Trout were first marked for the study in May 1997. Subsequent mark-recapture sessions took place in October 1997, June 1998, October 1998, June 1999, and October 1999.

Before each session the mark-recapture reach was divided into riffle/pool complexes. Each complex was a continuous 10-40 m reach of stream usually encompassing several pools and riffles and terminating at noticeable breaks such as boulder cascades or small waterfalls. A hipchain was used to measure the distance from the downstream end of the mark-recapture reach to the upstream end of each complex. The distance was recorded on a flag hung to mark the end of each riffle/pool complex.

Trout were captured by making a single pass through each complex with an AC backpack electrofishing unit. Fish were checked for marks from previous mark-recapture sessions and their length (mm), weight (g), and location of capture (m) was recorded. Those fish without marks were given a PIT tag and adipose clip. Fish less than 100 mm in length rarely received PIT tags. Fish were returned to the riffle/pool complex from which they were captured.

A recapture-only session was employed in June 2000 to recapture fish that had moved out of the mark-recapture section following previous marking sessions. The recapture only area extended from the confluence with the Rapidan River to the downstream end of the mark-recapture area, a distance of 1400m, and from the upstream end of the mark-recapture area to stream meter 4150, a distance of 1300m. During the recapture only session the stream was broken into approximately 50m riffle-pool complexes and two passes were made through each complex with a backpack electrofishing unit.

*Mark-Recapture Data Analysi*s

A histogram was developed to show distances moved by fish recaptured in their first session after being marked. Because fish marked near the edges of the 965m reach would be less likely to be recaptured than those marked near the center of the reach, I calculated the distance from the upstream end of each riffle/pool complex where a fish was marked to the upstream end of each riffle/pool complex in the entire mark-recapture area. These distances were weighted by the number of fish marked in each complex. This allowed for construction of a distribution of detectable movement distances (Brett Albanese and Charles Gowan, pers. comm.).

Recapture efficiency was calculated by dividing the number of fish recaptured in their first session after being marked by the total number of fish captured during their marking session. In addition, individual identification (PIT tags) made it possible to calculate the number of fish from each marking session that were not recaptured in their first session after being marked but were later recaptured during subsequent sessions. This allowed for calculation of the minimum number of fish that were alive but were not recaptured during the first recapture session for each marking session.

The potential effects of season (spring to fall, fall to spring) recapture period (May 1997-Oct 1997, Oct 1997-June 1998, June 1998-Oct 1998, etc.), section (affected, unaffected), direction moved (upstream, downstream), and change in relative weight (W_r) (Anderson and Neumann 1996) on distance moved by fish recaptured in their first session following marking were investigated using general linear models that assumed a gamma (skewed right) distribution of errors. Model one contained variables for season, section, direction moved, and change in W_r. Model two included variables for recapture period, section, direction moved, and change in W_r. Two models were used because of the highly correlated nature of the season and recapture period variables. The models were applied to the movement data and non-significant variables were excluded in the final analyses. Fish that moved between the affected and unaffected areas of the stream were excluded from analyses (excluded fewer than four fish for each session).

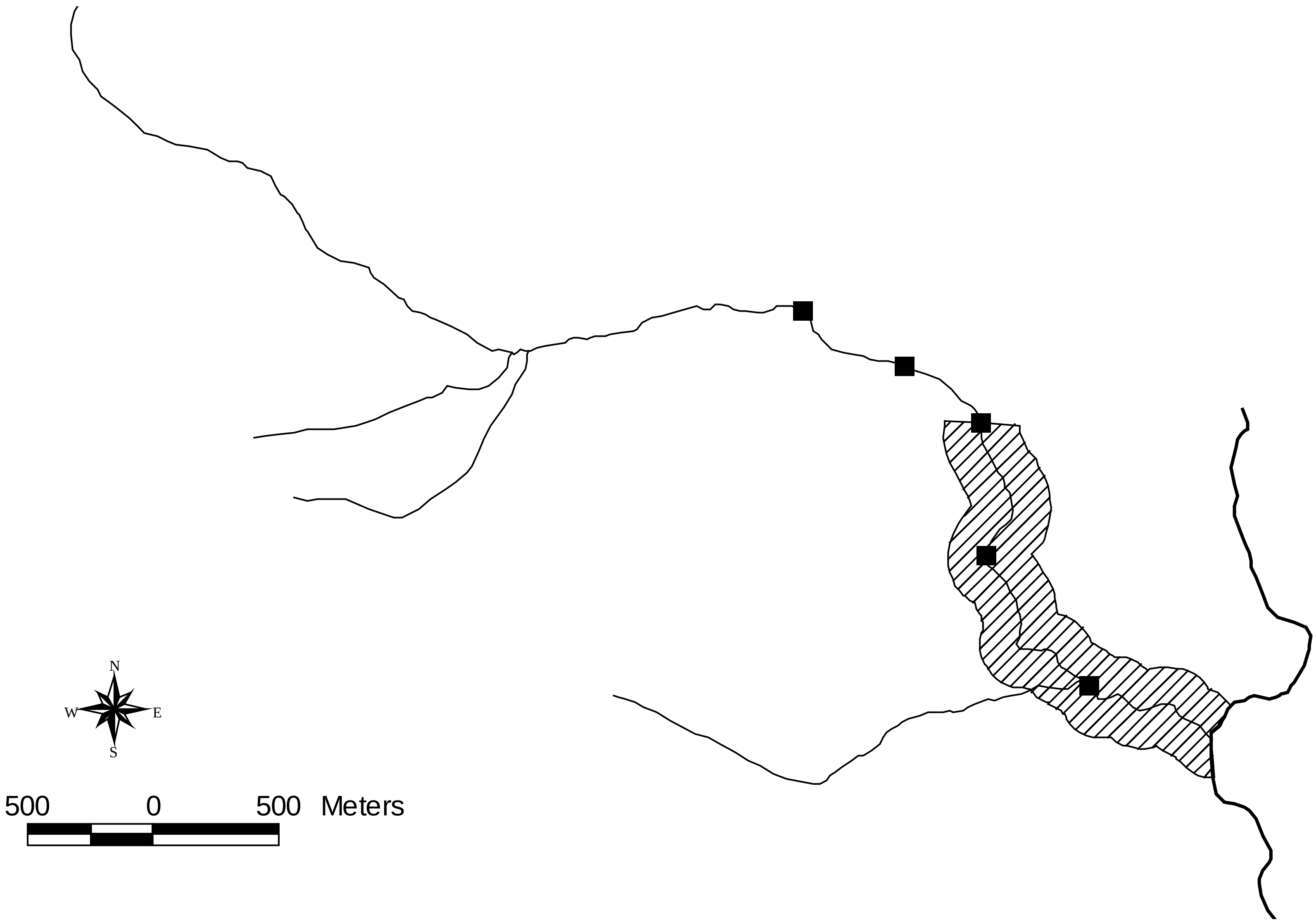

Figure I.2. Location of weirs (squares) on the Staunton River during fall 1999. Patterned area indicates debris flow affected area.

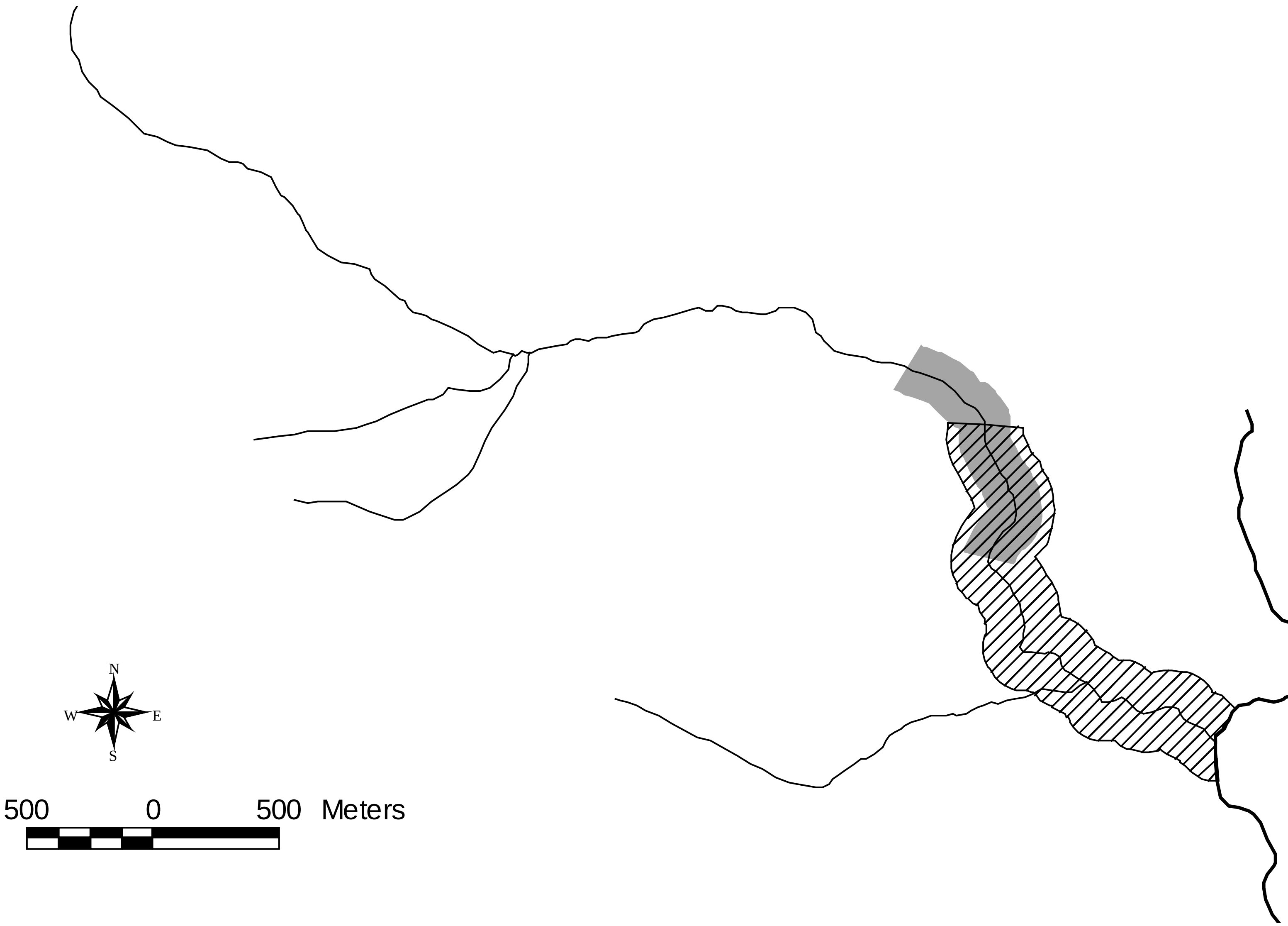

Figure I.3. Location of the mark-recapture study section (shaded area) on the Staunton River, 1997-1999. Patterned area indicates debris flow affected area.

Telemetry

Equipment and Surgical Procedure

Telemetry equipment consisted of a scanning receiver, 3-element folding Yagi antenna, and radio transmitters. The entire transmitter, antenna included, was sealed inside an epoxy capsule (1.7 g, 9 mm x 7 mm x 20 mm). Signals were transmitted in the 149.000-150.000 MHz range at 40 ppm. Battery life for the transmitters averaged 69 days (range 55 – 89).

Fish were captured using a DC backpack electrofishing unit. Laundry baskets placed in the stream served as holding areas before surgery. After 18-24 hours fish were examined to ensure good physical condition. Following examination fish were anesthetized in 200 mg/L MS-222 until they lost equilibrium. They were then measured, weighed, and placed on a portable surgical table which provided a continuous flow of water with 100 mg/L MS-222 across the gills (Courtois 1981). A 20 mm incision was made slightly anterior to the pelvic girdle and lateral to the midventral line with a stainless steel scalpel. A transmitter was inserted into the body cavity through the incision, which was then closed with 2-3 stitches of 3-0 non-absorbable suture material attached to a curved cutting needle. Fish were allowed to regain equilibrium in a bucket of stream water before being returned to their holding basket. The time from initial anesthesia to recovery of equilibrium was less than 15 minutes in most cases. The fish were held in their baskets for 18-24 hours before release at their point of capture.

Fifty-three transmitters were implanted between May and October 1999. Between May and July, 15 fish were implanted with transmitters in the affected area of the stream and 17 were implanted in the unaffected area (Table I.1a). Between September and October, 10 fish were implanted with transmitters in the affected area and 11 fish were implanted in the unaffected area (Table I.1b). Two of the fish implanted during summer were recaptured and their transmitters were replaced for continued tracking during the fall. Five fish were excluded from data analysis due to lack of observations (Table I.1). All implanted fish were captured and released between river meters 590 and 3125 (Figure I.4). Fish averaged 86 g, 215 mm (range 59-163 g, 183-283 mm) at the time transmitters were implanted.

Seasonal Range

Summer (May-August) and fall (September-December) ranges were determined by locating all fish once every 3-7 days. The habitat unit (pool or riffle) occupied was determined from the stream bank by triangulation. In most cases it was possible to determine the occupied unit without approaching the stream closely. Once fish location was determined the distance to the upstream end of the occupied pool or riffle from the confluence with the Rapidan River was recorded. Distances from the confluence to the upstream end of each habitat unit had been recorded during a BVET habitat survey in May 1999. If the occupied unit was a riffle >15 m in length then the exact fish location was recorded. Seasonal range is

Table I.1a. *S. fontinalis* observed with telemetry during summer 1999 in the Staunton River. a) Fish not included in analysis, lack of observation time. b) Fish 742 re-implanted with transmitter 542 during fall. c) Fish 292 re-implanted with transmitter 241 during fall. Fate: battery – transmitter battery died, stream – transmitter found in stream, bank – transmitter found on bank, predator – transmitter found in feces or moving outside of stream.

Frequency (Fish #)	Wt. (g)	Len. (mm)	Date Implanted	Last Observed	Days Observed	# Times Located	Location	Range (m)	Fate
472	125	250	05/27/99	08/10/99	75	17	Unaffected	0	battery
23	80	217	05/27/99	06/30/99	34	9	Unaffected	0	battery
382	112	235	05/27/99	07/28/99	62	14	Unaffected	14	battery
352	96	215	05/27/99	08/03/99	68	15	Unaffected	15	battery
202	94	202	05/27/99	08/07/99	72	16	Unaffected	16	battery
81	80	210	05/27/99	07/06/99	40	9	Unaffected	9	battery
233	87	220	05/27/99	07/23/99	57	13	Unaffected	128	battery
711	87	214	06/17/99	08/25/99	69	17	Unaffected	0	battery
533	93	215	06/17/99	08/27/99	71	16	Unaffected	16	battery
651[a]	74	204	06/17/99	06/30/99	13	4	Unaffected	0	stream
742[b]	78	205	06/17/99	09/12/99	87	18	Unaffected	2.1	battery
360	65	191	07/22/99	09/15/99	55	10	Unaffected	14.9	battery
391	72	203	07/22/99	09/23/99	63	12	Unaffected	31.8	battery
421	61	197	07/22/99	10/19/99	89	19	Unaffected	110.6	battery
451	63	197	07/22/99	09/23/99	63	12	Unaffected	0	battery
481	83	210	07/22/99	09/29/99	69	14	Unaffected	87.1	battery
511	64	204	07/22/99	09/15/99	55	10	Unaffected	0	battery
321	163	283	05/27/99	08/03/99	62	14	Affected	89.8	battery
292[c]	87	219	05/27/99	08/03/99	62	14	Affected	0	battery
261	106	250	05/27/99	08/15/99	72	16	Affected	7.6	battery
413	156	265	05/27/99	08/03/99	62	15	Affected	0	battery
442	92	227	05/27/99	07/06/99	40	9	Affected	0	predator
770	109	226	06/17/99	08/27/99	71	16	Affected	0	battery
681	79	210	06/17/99	08/27/99	71	16	Affected	0	battery
557	89	225	06/17/99	08/27/99	71	16	Affected	0	battery
502	98	225	06/17/99	08/27/99	71	16	Affected	0	battery
622	102	225	06/17/99	08/27/99	47	10	Affected	0	bank
592	99	226	06/17/99	08/27/99	44	7	Affected	33.6	stream
240	107	261	07/22/99	08/27/99	36	8	Affected	0	stream
270[a]	101	247	07/22/99	08/03/99	12	2	Affected	0	stream
302[a]	99	224	07/22/99	08/10/99	19	4	Affected	0	stream
331[a]	85	220	07/22/99	08/10/99	19	4	Affected	0	bank

Table I.1b. *S. fontinalis* observed with telemetry during fall 1999 in the Staunton River. a) Fish not included in analysis, lack of observation time. b) Fish 742 (Table I.1a) re-implanted with transmitter 542 during fall. c) Fish 292 (Table I.1a) re-implanted with transmitter 241 during fall. d) Fish 212 moved from unaffected to affected area and was not included in comparisons of range size between locations. Fate: battery – transmitter battery died, stream – transmitter found in stream, bank – transmitter found on bank, predator – transmitter found in feces or moving outside of stream.

Frequency (Fish #)	Wt. (g)	Len. (mm)	Date Implanted	Last Observed	Days Observed	# Times Located	Location	Range (m)	Fate
542[b]	71	202	09/29/99	12/01/99	63	15	Unaffected	68.3	battery
801	63	189	10/14/99	12/30/99	77	18	Unaffected	102.6	battery
831	70	198	10/14/99	12/27/99	74	17	Unaffected	80.1	battery
891	92	215	10/14/99	11/20/99	37	7	Unaffected	38.7	unknown
573	68	189	10/14/99	12/27/99	74	17	Unaffected	22.2	battery
212[d]	76	202	10/14/99	01/10/99	88	20	Unaffected	1952.4	battery
272	65	206	10/14/99	11/29/99	46	8	Unaffected	258.5	bank
332	93	217	10/14/99	12/01/99	48	9	Unaffected	22.2	stream
30[a]	67	198	10/14/99	11/06/99	23	4	Unaffected	18.5	bank
62	81	216	10/14/99	12/01/99	48	8	Unaffected	30.7	bank
91	95	220	10/14/99	11/13/99	30	6	Unaffected	339.4	predator
12	59	184	10/14/99	11/20/99	37	7	Affected	37.5	stream
304	70	203	10/19/99	12/20/99	62	14	Affected	15.5	battery
241[c]	95	219	10/19/99	01/03/00	76	18	Affected	40.2	battery
153	71	194	10/19/99	12/30/99	72	17	Affected	528.4	battery
183	83	208	10/19/99	01/14/00	86	20	Affected	75.5	battery
123	78	214	10/19/99	11/29/99	41	6	Affected	23.7	predator
605	72	203	10/19/99	12/16/99	58	13	Affected	93.9	battery
981	62	192	10/19/99	12/23/99	65	15	Affected	94.5	battery
951	88	208	10/19/99	12/27/99	69	16	Affected	39.4	battery
921	59	183	10/19/99	12/27/99	69	16	Affected	28.9	battery

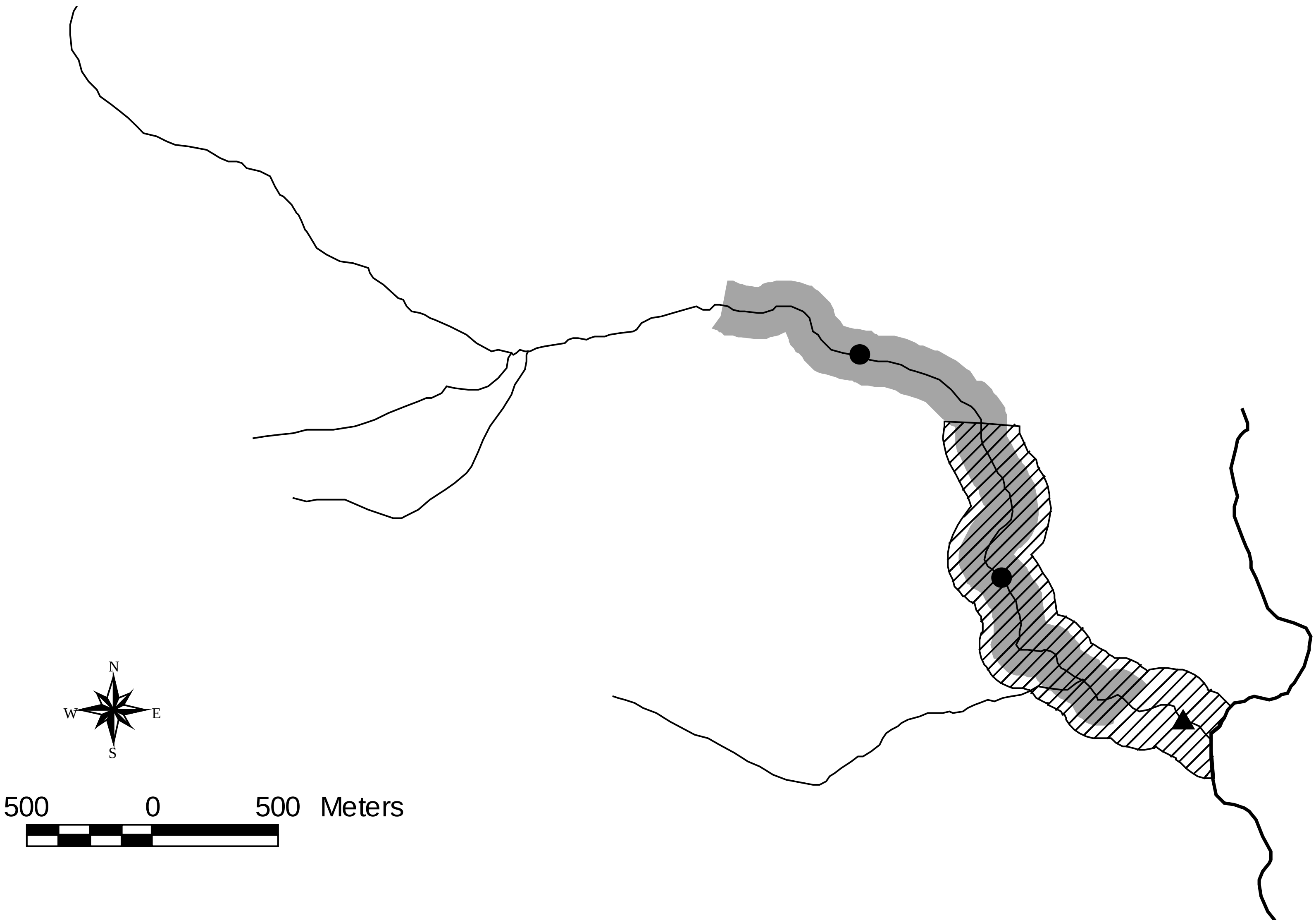

Figure I.4. Location of the summer and fall 1999 radio telemetry study (shaded area) on the Staunton River. Patterned area indicates debris flow affected area. Circles indicate location of temperature loggers. Triangle indicates location of discharge monitoring station.

defined here as the distance between the upstream end of the most upstream unit and the upstream end of the most downstream unit occupied by the fish during the entire season.

Diel Range and Activity

Diel range and activity were simultaneously investigated in the affected and unaffected areas during summer (June-August) and fall (October-November). Groups of 3-7 fish were observed by locating individual fish once every hour over a 24-hour period. Six fish also were observed in the affected area for one 24-hour period in December. Observation began by walking parallel to the stream at a distance that would not disturb the monitored fish until a clear transmitter signal was obtained. The antenna was then held steady and fluctuations in signal strength were recorded for a continuous 3-minute period. Changes in signal strength resulted from changes in orientation of the transmitter to the antenna due to activity by the monitored fish (Clapp et al. 1990).

The occupied pool or riffle was determined by triangulation after activity monitoring was completed. Flags were hung each hour at occupied habitat units. Diel range was measured with a meter tape at the end of the 24-hour period. Diel range is defined here as the distance between the upstream end of the furthest upstream unit and the upstream end of the furthest downstream unit occupied by the fish during the entire 24-hour period.

Discharge and Water Temperature

Discharge was recorded on both an hourly and daily basis by a stream gauge located 140m upstream from the confluence with the Staunton River (Figure 1.4). The gauge is maintained by the Department of Environmental Sciences at the University of Virginia, Charlottesville, VA.

Two Stowaway Tidit (Onset Computer Corporation) temperature loggers were installed in the Staunton River in May 1999. The loggers were installed in the debris flow affected area (stream meter 1120) and the unaffected area (stream meter 2650) (Figure I.4). The loggers recorded water temperature once per hour from May – December 1999, with the exception of the affected area logger which was lost in a flood on September 21 – October 1 and was replaced on November 8 (Appendix A).

Seasonal Range Data Analysis

A rank-based linear model was used to examine seasonal range data. This nonparametric approach does not assume normality and is very robust to nonsymmetrical data distributions (Hettmansperger et al. 1998). The sole assumption of rank based linear models is a continuous distribution of error terms. The approach was used to test for differences in range size by location (affected area, unaffected area) and by season (summer, fall). Interaction between season and location was not significant ($p>0.05$) for the seasonal data, therefore significant differences between seasons and between locations could be examined with the model. Fish weight and length were included as covariates. The range data were log-transformed to decrease variance. Non-significant variables were left in the model to provide spread in data clustered around zero.

Polynomial regression was used to examine relationships between fish movement and water temperature, and discharge. For each 3-7 day period the total distance moved by all fish was summed. This summed distance was divided by the number of fish being monitored over the period as well as by the number of days in the period to give the movement per fish per day. The average water temperature over the period was then calculated. The log of movement per fish per day was plotted vs. average water temperature and a polynomial regression line was fitted to the data. The same procedure was used for discharge.

Diel Range Data Analysis

A rank-based linear model was used to test for differences in diel range size by location (affected area, unaffected area) and by season (summer, fall). Interaction between season and location was significant ($p<0.05$), thus the data were split by season and differences in diel range between the affected and unaffected areas within each season were examined separately. Statistical comparisons between seasons could not be made once the data were split. Fish weight and length were included in the model as covariates. The range data was log transformed to decrease variance. Non-significant variables were left in the model to provide spread in data clustered around zero.

Low occurrence of movement over the diel periods prevented any analyses of relationships between movement and temperature or discharge.

Activity Data Analysis

A generalized linear model assuming a gamma (skewed right) distribution of errors was used to test for differences in activity level by location (affected area, unaffected area) and by season (summer, fall) with water temperature as a covariate. Discharge did not fluctuate within diel periods or within seasons and was not included in the model. Interaction between season and location was significant ($p<0.05$), thus the data were split by season and differences in activity between the affected and unaffected areas within each season were examined separately. Statistical comparisons between seasons could not be made once the data were split.

Results

Recolonization

Age 1+ and YOY *S. fontinalis* were distributed throughout the Staunton River and in Wilson Run during all BVET snorkel surveys prior to June 1995 (Figure I.5). In October 1995, four months post-debris flow, only five age 1+ *S. fontinalis* were found in the debris flow affected area, all within 150m of the upstream end of the debris flow. No YOY were found in the debris flow affected area in October 1995 and no age 1+ or YOY were found in Wilson Run. *S. fontinalis* remained distributed throughout the unaffected area.

Recolonization progressed mostly from upstream to downstream for age 1+ fish between May 1996 and May 1998 (Figure I.5). Age 1+ fish were found up to 400m downstream of the upstream end of the affected area in May 1996. By October 1996 they were commonly found 800m downstream of the upstream end of the debris flow. Age 1+ fish were found in the lower 200m of the affected area in May 1996 but were not common there until October 1996. The June 1997 and October 1997 surveys revealed that age 1+ *S. fontinalis* were common in all but a 200-250m reach beginning approximately 200m upstream from the confluence with the Rapidan River. Age 1+ *S. fontinalis* were common throughout the Staunton River by May 1998. When the difference between the furthest downstream distance between each snorkel survey was divided by the number of months between each survey, the age 1+ recolonization rate was 49 m/month ($\sigma = 29$). The period from October 1997 to May 1998 was excluded from this calculation because of overlap between upstream and downstream colonizers during that period.

YOY *S. fontinalis* were common up to 200m and 500m downstream of the upstream end of the affected area in May and October 1996, respectively. YOY were not found in the lower 200m of the stream until June 1997. The June and October 1997 surveys revealed that YOY were absent from a 600m reach beginning approximately 200m upstream from the confluence with the Rapidan River. YOY fish were distributed throughout the Staunton River by May 1998. No YOY or age 1+ fish were observed in Wilson Run after October 1995, although no formal surveys were performed there. In total it took 2.5-3 years for both YOY and 1+ *S. fontinalis* to recolonize the debris flow affected area.

Weirs

Estimated escapement rates were not significantly different for the downstream (1.4%) and upstream (8.6%) traps ($p=0.08$). Trap escapement did not vary by total length ($p=0.97$).

The weirs failed to produce any usable movement data. The weirs were breached several times by high flows and all five were destroyed by a flood in September, 1999. Three sets of weirs were re-installed in mid-October but leaves plugged trap entrances and caused the weirs to be breached on a daily

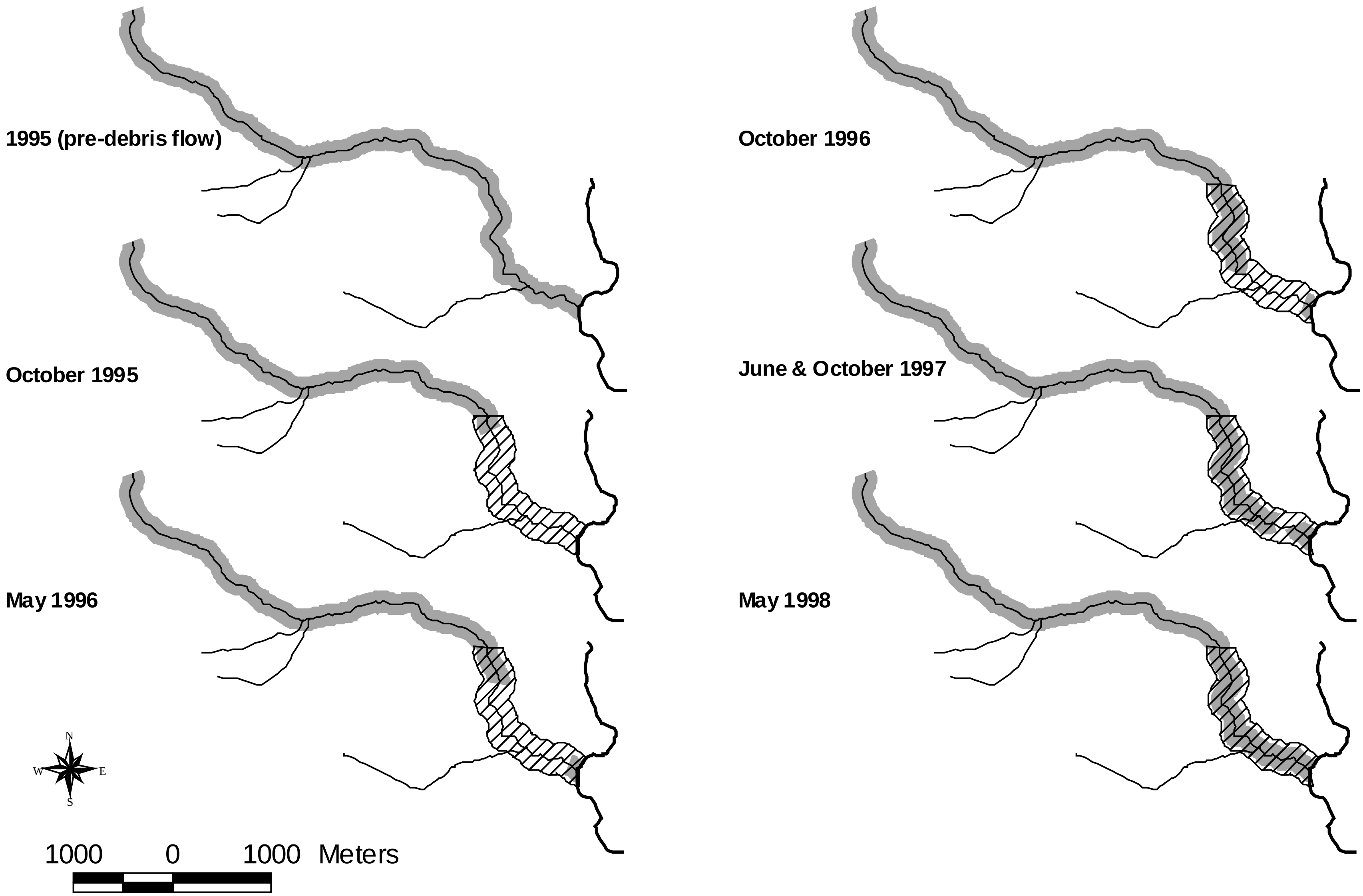

Figure I.5. Recolonization of the debris flow affected area of the Staunton River by age 1+ *S. fontinalis*, 1995-1998. Shaded areas indicate sections of the stream occupied by S. fontinalis. Patterned area indicates debris flow affected area.

basis. *S. fontinalis* with radio transmitters moved past one set of weirs on a regular basis without being caught.

While the weirs did not provide movement data they did capture a wide variety of species. In addition to *S. fontinalis*, brown trout (*Salmo trutta*), sculpin (*Cottus spp.*), torrent suckers (*Thoburnia rothoeca*), blacknose dace (*Rhinichthys atratulus*), and several species of salamanders, snakes, crayfish, frogs, and toads were captured during weir testing in Barbour's Creek. *S. trutta* were not known to inhabit Barbour's Creek. It was particularly interesting to note a downstream pulse of pickerel frog (*Rana palustris*) movement during the first week in April in Barbour's Creek and consistently high catches of crayfish (species not identified) moving upstream and downstream in both Barbour's Creek and Staunton River during spring, summer, and fall (weirs were not in place during winter). More crayfish were captured than any other species, including *S. fontinalis* in both Barbour's Creek and Staunton River.

Mark-Recapture

Fish that moved less than 100m upstream or downstream made up 91% of *S. fontinalis* recaptures (Figure I.6a). The maximum detected upstream and downstream movements were 410m and 873m, respectively. Construction of a distribution of detectable movements revealed that the probability of detecting movements of 100m upstream or downstream was relatively high and decreased steadily towards zero as distance moved increased (Figure I.6b).

Recapture percentages ranged from 15%-27% for the first recapture session after each marking session (Table I.2). Recapture percentage increased for every recapture session following the first, as fish that were not caught in previous sessions were recaptured. Percentage of fish recaptured increased from 22% during the first recapture session to 47% when all four recapture sessions following the first were included for fish marked in June 1997. The number of fish known to be alive at the time of the first recapture session could be calculated by adding the number of fish recaptured in the first session to the additional number of recaptures. The percent missed could be calculated by dividing the number of additional recaptures by the number of fish known to be alive at the time of the first recapture session. Using these calculations, a *minimum* of 54% of the number of fish confirmed to have been alive at the time of the initial recapture session were not recaptured for fish marked in June 1997. This was the minimum percent missed as it does not include fish that left the mark-recapture area, remained but were never caught, or lost their PIT tags.

During the June 2000 recapture only session, 17 PIT tagged fish were recaptured outside of the mark-recapture area. Fish from all marking sessions were captured with the exception of June 1997. Fish were captured up to150m upstream and 1100m downstream from the mark-recapture area, with 15 of the 17 recaptured downstream (Table I.3).

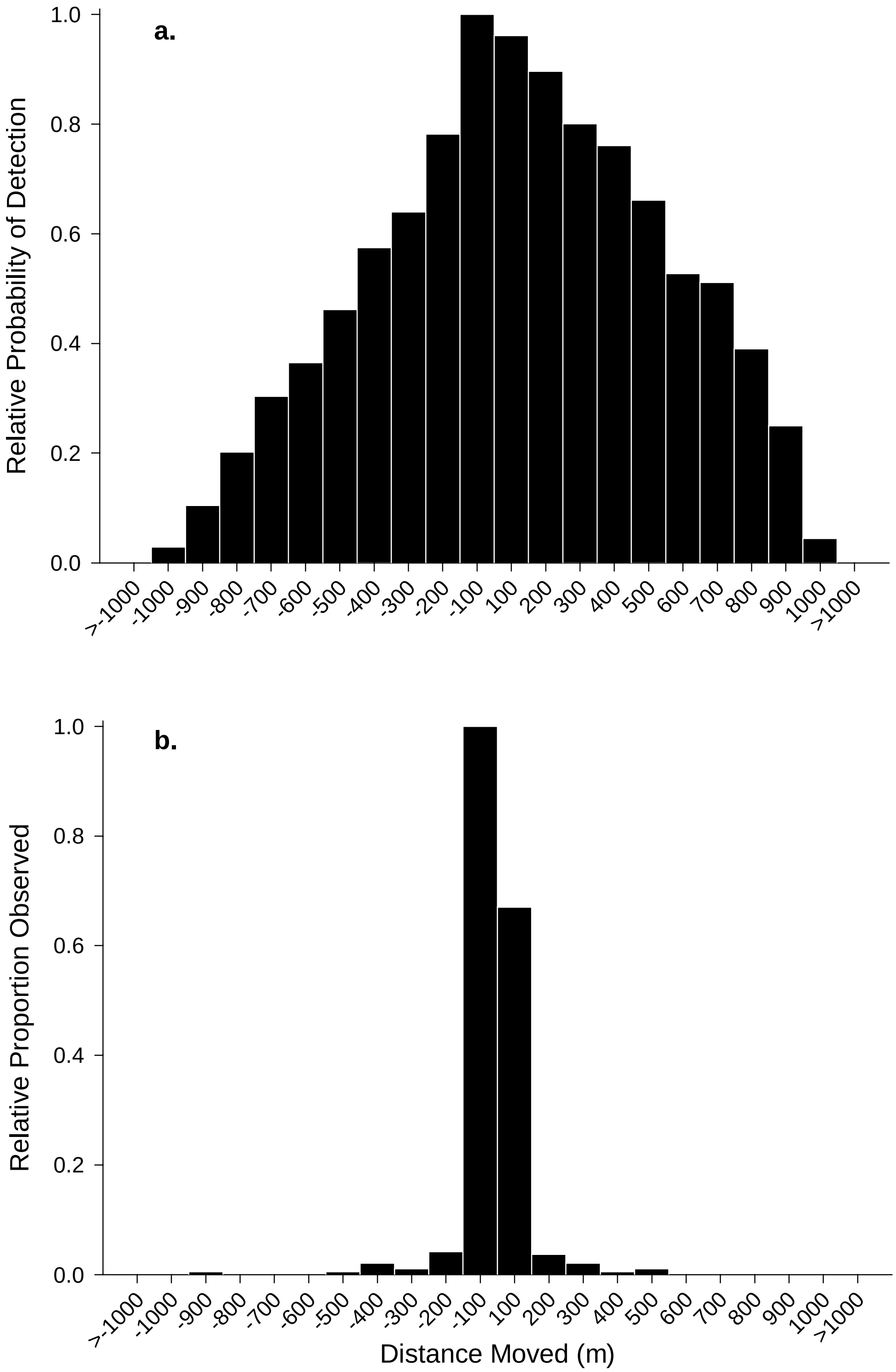

Figure I.6. a. Distribution of detectable movement distances weighted for number of fish marked in each riffle-pool complex between 1997 and 1999 in the Staunton River. b. Movements detected during the 1997-1999 mark-recapture sessions in the Staunton River (n=347).

Table I.2. Summary of *S. fontinalis* mark-recapture data for the Staunton River, SNP, May 1997-June 1999. Percentages indicated in parentheses. Minimum percent missed indicates the percentage of fish confirmed to have been alive but not recaptured in the first session after marking, = (total recaptures – 1st session recaptures) / total recaptures. Percent missed reported as a minimum as it does not account for fish that left the mark-recapture area, remained but were never captured, or lost their PIT tag.

Date Marked	# Marked	1st Session Recaptures	Total Recaptures	Minimum % Missed	# of Recapture Sessions
May-97	83	18 (22)	39 (47)	54	5
Oct-97	316	61 (19)	99 (31)	38	4
Jun-98	268	72 (27)	87 (33)	17	3
Oct-98	301	51 (17)	56 (19)	8	2
Jun-99	438	64 (15)	64 (15)	0	1
Total	1406	266 (19)	335 (24)	--	--

Table I.3. PIT tagged *S. fontinalis* recaptured outside of the mark-recapture (m-r) area on the Staunton River in June 2000. Location: DS = downstream of the m-r area, US = upstream of m-r area. Number of times recaptured indicates number of times recaptured inside the m-r area. Distance indicates the number of meters upstream or downstream from the m-r section the fish was captured.

Fish #	Location	Date Marked	# Times Recaptured	Years Since Last Recaptured	Distance (m)
1	DS	Oct-97	0	2.5	1100
2	DS	Oct-97	2	1.5	1000
3	DS	Oct-97	1	1.5	650
4	DS	Oct-97	0	2.5	100
5	DS	Jun-98	0	2.0	700
6	DS	Jun-98	0	2.0	350
7	DS	Jun-98	1	1.5	350
8	DS	Jun-98	2	0.5	50
9	DS	Oct-98	0	1.5	600
10	DS	Oct-98	0	1.5	200
11	US	Oct-98	1	1.0	150
12	DS	Jun-99	0	1.0	200
13	DS	Jun-99	0	1.0	100
14	DS	Jun-99	0	1.0	100
15	DS	Oct-99	0	0.5	550
16	US	Oct-99	1	0.5	50
17	DS	Jun-00	0	0.0	50

Results from the gamma regressions were not consistent between the models. Model one was not significant for any of the included variables ($p>0.05$). Model two was significant for section and recapture period ($p<0.05$), indicating differences in distances moved between affected and unaffected areas and between recapture periods. Wr and direction moved were not significant ($p>0.10$) and were excluded from the final model. Pairwise comparisons between recapture periods revealed that there was no consistent significant difference between spring to fall and fall to spring movements.

Telemetry

Seasonal Ranges

The median seasonal ranges for summer affected, summer unaffected, fall affected, and fall unaffected were 0m, 14m, 39m, and 68m, respectively (Figure I.7). Interaction between season and location was not significant ($p > 0.30$), therefore it was possible to test for effects of season and location on seasonal range size. No significant difference was found between affected and unaffected areas ($p = 0.058$), whereas a highly significant difference was found between summer and fall ($p = 0.0003$). None of the covariates, weight, length, or transmitter life, were found to be significant ($p > 0.400$). The maximum ranges for summer affected, summer unaffected, fall affected, and fall unaffected were 130m, 90m, 530, and 340m, respectively. The largest range detected was 1950m, a fish that moved from the affected to the unaffected area of the stream during fall.

The linear term in the temperature regression model was not significant ($p>0.10$), but the negative quadratic term was significant ($p<0.05$) indicating activity was greatest at moderate temperature levels. The regression showed that movement peaked at temperatures between 8C and 10C and that distance moved decreased rapidly as temperatures approached 18C (Figure I.8a). There was a large amount of variability in the data not explained by the model ($r^2 = 0.40$).

Both the linear and negative quadratic terms in the discharge regression model were significant ($p<0.01$) indicating that activity increased as temperature increased and also peaked at moderate levels. The regression showed that movement peaked at discharges between $.25m^3/s$ and $.31m^3/s$ and that distance moved was minimal at high and low discharges (Figure I.8b). A large amount of variability in the data was not explained by the model ($r^2 = 0.25$).

Diel Ranges

The median diel ranges for summer affected, summer unaffected, fall affected, and fall unaffected were 0m, 0m, 12m, and 0m, respectively (Figure I.7). Interaction between season and location was significant ($p < 0.0001$), therefore it was not possible to test for effects of season and location simultaneously. When the data were split into separate models for summer and fall significant differences in ranges were found between affected and unaffected areas for fall ($p=0.03$). The summer data could not be interpreted due to the large number of range sizes equal to zero. The covariates, length and weight,

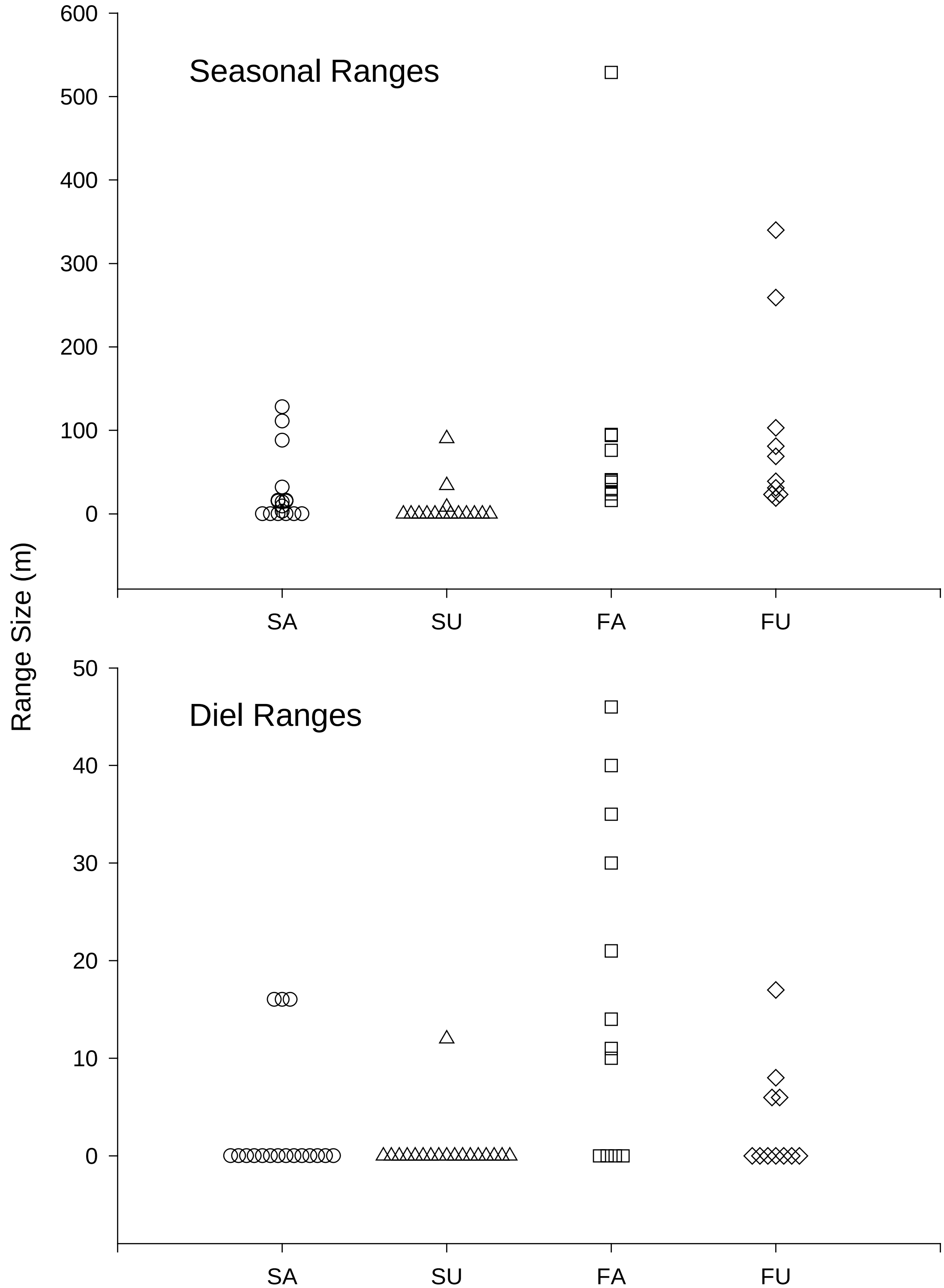

Figure I.7. Seasonal (n=47) and diel (n=58) range sizes for *S. fontinalis* as observed with radio telemetry during summer and fall 1999. SA = summer, affected area; SU = summer, unaffected area; FA = fall, unaffected area; FU = fall, unaffected area.

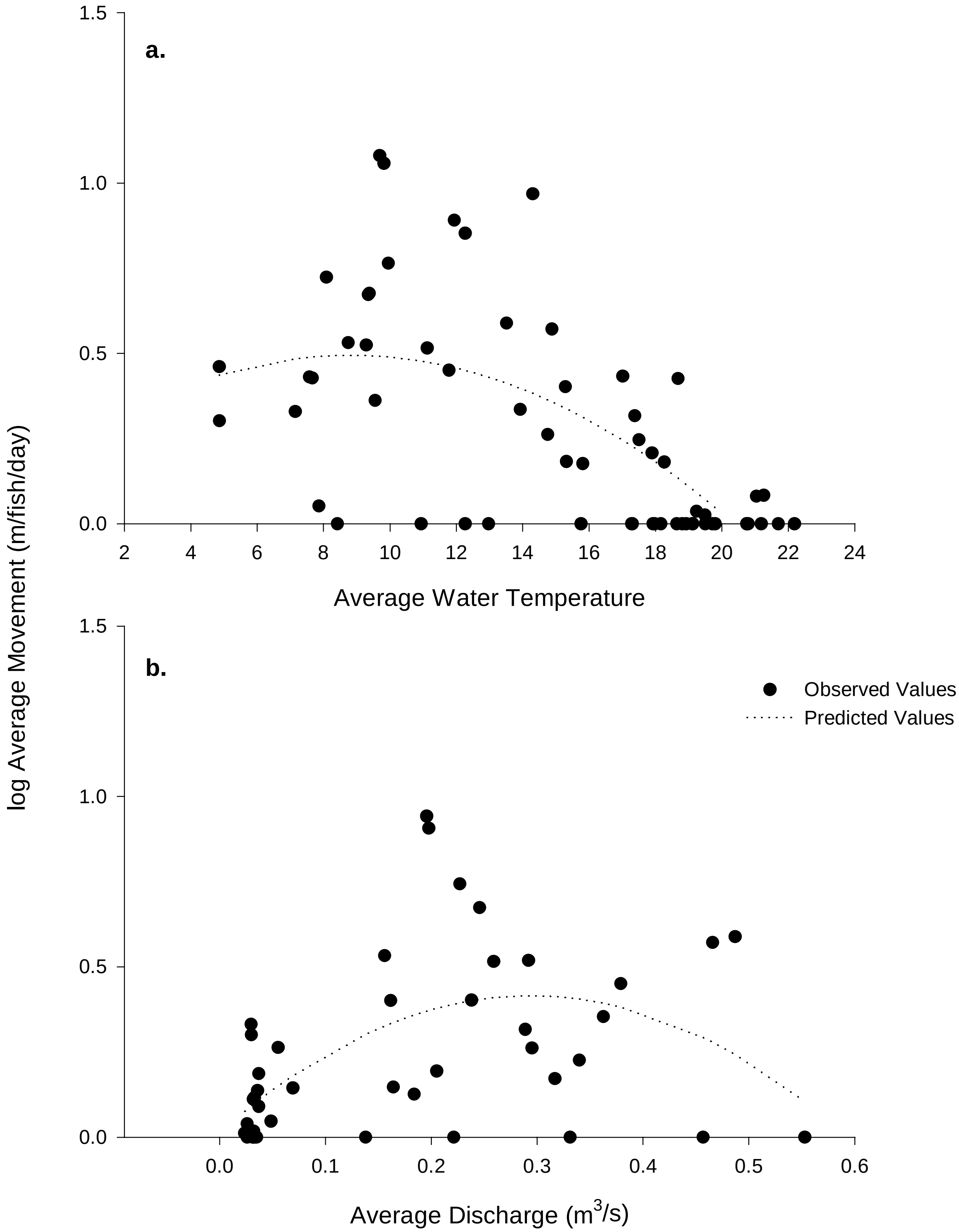

Figure I.8. Polynomial regression results for average movement vs. a. water temperature (C) (r^2=0.40), and b. discharge (r^2=0.25) for *S. fontinalis* in the Staunton River as observed using radio telemetry during spring and fall 1999.

were not significant for fall ($p>0.10$). Although statistical comparisons could not be made between seasons, range size appeared to increase during the fall. The maximum diel ranges for summer affected, summer unaffected, fall affected, and fall unaffected were 16m, 11m, 45m, and 16m, respectively. Diel observation of six fish during December in the affected area yielded a median range of 0m, and a maximum range of 0m. Fish did not consistently move during the night or the day during any season.

Due to the lack of movements during diel observations it was not possible to test for correlation between average movement and temperature or discharge.

Activity

Interaction between season and location was significant ($p<0.001$), therefore it was not possible to test for the effects of season and location simultaneously. When the data were split into two models significant differences in activity level were found between locations for both summer ($p<0.001$) and fall ($p<0.001$). Average predicted values indicated that activity was higher in the unaffected area in both seasons. Although seasons could not be compared statistically, there was a notable change in the diel activity pattern and level from summer to fall (Figure I.9). Fish displayed a crepuscular activity pattern with peaks near dawn and dusk during summer. There was a singular sustained peak in activity during fall between hours 1100 and 1700 and the peak activity level was much higher than during summer. During the summer, temperature had a significant linear ($p=0.01$) and negative quadratic ($p<0.01$) effect on activity. Temperature did not have a significant effect on activity during fall ($p>0.07$).

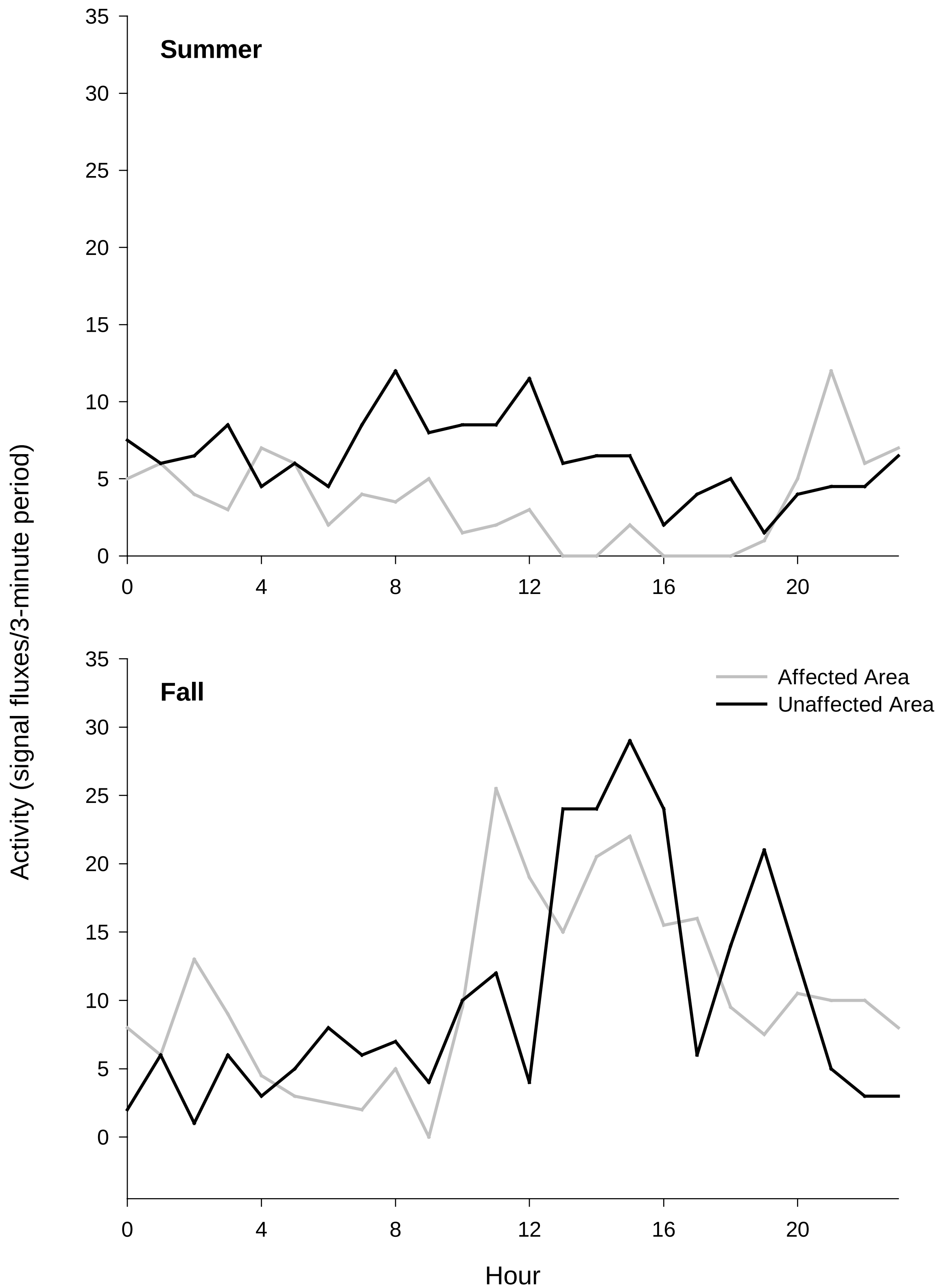

Figure I.9. Diel activity patterns for *S. fontinalis* in debris flow affected and unaffected areas of the Staunton River as observed with radio telemetry during summer and fall 1999. Hour 0 = 12:00AM.

Discussion

Recolonization

Elimination of *S. fontinalis* from the lower 1.9km of the Staunton River in June 1995 presented a rare opportunity to examine recolonization following a natural catastrophic disturbance. Both YOY and age 1+ *S. fontinalis* recolonized the stream well within the 1-4 year recolonization times reported for salmonids following experimental defaunation and natural catastrophic disturbances (Detenbeck et al. 1992, Lamberti et al. 1991, Phinney 1975, Propst and Stefferud 1997, Swanson et al. 1998). The recolonization pattern observed (mainly from upstream to downstream) is not surprising for three reasons, 1) the existence of a 20m long, steep bedrock cascade 200m upstream from the confluence with the Rapidan River, 2) the status of *S. fontinalis* population in the Rapidan River before and after the debris flow, and 3) the existence of a debris flow unaffected population upstream of the affected area. Although the bedrock cascade is not a barrier to fish movement (*T. rothoeca* and *Nocomis leptocephalus* have both been found upstream of the cascade since the debris flow), it is an impediment to movement (Adams et al. 2000), especially during base flow. Portions of the Rapidan River both upstream and downstream of its confluence with the Staunton River were affected by debris flows and *S. fontinalis* were certainly depleted, if not eliminated from the Rapidan River near the confluence. The population in the debris flow unaffected area was immediately available for recolonization and by October 1995 had already recolonized the upper 150m of the debris flow affected area.

The recolonization rate and pattern observed in the Staunton River has important implications in management of disturbed *S. fontinalis* populations. The speed of recolonization demonstrates the resiliency these populations have to catastrophic events. This ability to quickly repopulate decimated reaches should preclude the use of stocking to supplement similarly impacted populations in the future. It was suggested that the Staunton River be stocked following the debris flow to speed the recovery of the fish population. Stocking would have likely taken place were it not for the identification of the Staunton population as genetically unique within SNP (Poompuang et al. 1997). In this case stocking would have cost time and money, possibly destroyed the genetic integrity of the population, and would not have greatly decreased population recovery time.

Recolonization of the affected area also provided information of the movement capabilities of *S. fontinalis*. Since the debris flow occurred in June 1995, reproduction within the debris flow affected area could not have accounted for any age 1+ fish found there before the June 1997 survey. All 1+ fish found in the debris flow affected area during the October 1995, May 1996, and October 1996 surveys had to have moved there from outside of the section. Age 1+ fish were found up to 400m downstream of the upstream end of the affected area in May 1996 (Figure I.5). By October 1996 they were commonly found

800m downstream of the upstream end of the debris flow and some were located >1km downstream, indicating that age 1+ *S. fontinalis* are capable of moving several hundreds meters in less than one year.

Information on YOY movement capabilities could also be obtained from mark-recapture data. Movement must account for the difference between YOY location in May and October 1996 since *S. fontinalis* spawns only once per year, during the fall, and spring surveys were intentionally scheduled after YOY swim-up. Spring and fall locations of YOY differed by 300m in 1996, indicating that YOY may be capable of moving of several hundred meters during the summer, however YOY fishes may also be displaced during high discharges (Harvey 1987). A flood triggered by hurricane related storms during September 1996 may have moved YOY downstream between the May and October 1996 surveys.

It is important to note that rapid recolonization depended on a nearby source population. Given the creation of the bedrock cascade and the loss of fish in the Rapidan River near the confluence with the Staunton River it is unlikely that the population would have recovered as quickly in the absence of an upstream source. Rapidan River fish would have likely eventually repopulated the Staunton River, but an alternative source population is not a luxury shared by all streams in SNP. Many streams that support *S. fontinalis* populations within SNP flow outside of the Park where deforestation, development, and agriculture often create unsuitable habitat conditions. The *S. fontinalis* populations in many of these streams are essentially isolated because of the habitat conditions outside of the Park. Should a debris flow or other catastrophic event eliminate the *S. fontinalis* population from such a stream, it is unlikely that it would be quickly, if ever, recolonized. If this scenario were encountered transplanting fish from other streams may become a viable option. The source of fish for transplantation should be from within close proximity, preferably within the same watershed, as populations within different areas of the Park have been shown to differ genetically and the extent of the variation is not yet fully known (Poompuang et al. 1997).

The results of the recolonization study and the rapid recovery of the fish population (Chapter 2) may seem to indicate that removal of trees from the riparian area was not detrimental to the fish population and imply that removal of trees from riparian areas during activities such as logging and housing development is acceptable. However there are some very basic differences between natural and anthropogenic disturbances that need to be considered. While natural disturbances tend to be localized short-term events, anthropogenic disturbances can be more chronic, having effects over a longer period of time and over a greater area, which can greatly increase population recovery time (Detenbeck et al. 1992). Although clearing of riparian areas may initially increase stream production, chronic impacts such as increased sediment input, decreased LWD inputs, and loss of seasonal habitats decrease fish production in the long-term (Murphy et al. 1986). The fact that the debris flow was a localized event and that the forest in the entire watershed around the debris flow affected area was intact helped to minimize the impact of the event on the fish population and stream as a whole and will facilitate recovery of the riparian zone.

Weirs

Estimated escapement rates from the weirs were relatively low and escapement was not significantly different between upstream and downstream traps as has been found in past studies (Gowan and Fausch 1996). It is encouraging that both the present study and Gowan and Fausch (1996) did not find that escapement varied by fish length. More research is needed in alternative designs, configurations, and escapement from weirs as it is likely that escapement varies by trap design and weir configuration.

The weirs were not functional in high gradient streams during times of high flows or high debris transport. The wings and traps often became plugged, backing up water and causing the weirs to be breached. Weir installation, maintenance, and checking were extremely labor intensive. A trail along the stream afforded only foot access and weir components were transported via backpack. Weir installation in remote areas without the use of vehicles to transport components is not recommended. Care should be taken not to install weirs in areas with beaver activity as beavers repeatedly chewed holes in one weir. Careful consideration should be made as to materials used to construct weirs. While the use of aluminum frames made this design particularly strong, it was also particularly expensive (approx. $600 per weir). An extremely high water event can destroy a very large investment in a very short time, as was the case here.

Mark-Recapture

The initial mark-recapture results seemed to support the notion that *S. fontinalis* are sedentary. However, upon closer inspection 1) 9% of the recaptured fish moved >100m, 2) only 19% of fish were recaptured in the first session after being marked, and 3) the distribution of detectable movement distances indicated that long distance movements were relatively unlikely to be detected using this study design. In addition, marked fish were recaptured outside the mark-recapture area in June 2000. These revelations had several implications in the interpretation of the mark-recapture data.

The percentage of fish moving more than 100 m was likely an underestimate given the low relative probability of detecting long distance movements. If a correction factor could be developed and applied to the data it would have boosted the proportion of recaptures that moved long distances and decreased the proportion that moved short distances (Albanese and Gowan, pers. comm.). Furthermore, coupling low probability of detecting a long distance movement with low recapture rates pushed the probability of detecting long distance movements even lower. When these factors were taken into consideration it became apparent that this study design was capable of providing large amounts of data only for fish that moved very little. This confounded attempts to find effects of season, location, W_r, or direction moved on movement because all the movement data were clustered near zero. It would be difficult to correct for the biases introduced by this study design, especially given the low recapture rate.

Low recapture rates were particularly troubling since not capturing fish could have meant they died, moved out of the study area, or remained in the study area but were not recaptured. Past studies typically blamed low recapture rate on mortality, while sampling efficiency and movement were seldom considered factors (Gowan and Fausch 1994). However our data indicated that a minimum of 54% of the fish that were confirmed to have been alive were not recaptured at the time of the first recapture session (Table I.2, May 1997). Fish that were alive but not captured during their first recapture session could have been in the mark-recapture reach and not captured or have moved outside of the mark-recapture area and returned at some later time.

One may suspect that the low recapture rate was due to the use of a single pass during the mark-recapture sessions. During the June 2000 recapture only session first pass capture efficiency was assessed by making two passes through each riffle/pool complex upstream of the mark-recapture area with effort equal to that used during the 1997-1999 mark-recapture study. On average, 73% of the estimated number of fish in each riffle/pool complex were captured during the first pass (Table I.4). In comparison, Kruse et al. (1998) had an 83% first pass capture rate in a high gradient trout stream with lower population density than the Staunton River (Table I.4).

Inference about the proportion of fish that had moved out of the mark-recapture reach during the first recapture session for fish marked in June 1997 may be made from the first pass capture rate information. Of the 39 fish known to have been alive during the first recapture session in October 1997, 18 were captured in the mark-recapture reach. If I assumed that this represented 73% (the first pass capture rate) of the marked fish present in the reach, then 18 divided by 0.73, or 25, marked fish were present in the reach at that time. This means that 39 minus 25, or 14, marked fish were outside of the mark-recapture reach making them unavailable for recapture in October 1997. A closer examination of the mark-recapture data revealed that in May 1997, 14 fish were marked within 100 m of the upstream or downstream end of the mark-recapture section (Table I.5). It is plausible that a number of marked fish moved out of the mark-recapture reach before the October 1997 session and returned at a later date. The major implication is that recapture efficiency and fish movement both played roles in low recapture rates during the mark-recapture study.

Table I.4. Population estimates in individual riffle/pool complexes for age 1+ trout from two-pass depletions performed during the recapture only session in June 2000 on the Staunton River and Kruse et al. (1998). Only complexes upstream of the mark-recapture reach were used to assess first pass capture efficiency for the Staunton River study. N = population estimate, performed using methods outlined in Kwak (1991), na indicates that population estimate could not be made because there was not a depletion in the second pass. Rate = Pass #1/N, and indicates proportion of fish captured on first pass.

Present Study					Data from Kruse et al. 1998				
	Pass #					Pass #			
Complex	1	2	N	Rate	Complex	1	2	N	Rate
1	20	2	22	0.90	1	6	2	9	0.67
2	35	1	36	0.97	2	16	4	21	0.75
3	36	9	48	0.75	3	13	1	14	0.92
4	45	5	51	0.89	4	7	3	12	0.57
5	26	3	29	0.88	5	2	4	na	na
6	5	4	25	0.20	6	3	0	3	1.00
7	43	3	46	0.93	7	1	1	na	na
8	19	1	20	0.95	8	14	4	20	0.71
9	25	7	35	0.72	9	13	5	21	0.62
10	32	5	38	0.84	10	4	0	4	1.00
11	44	12	61	0.73	11	21	5	28	0.76
12	28	8	39	0.71	12	1	0	1	1.00
13	38	12	56	0.68	13	19	2	21	0.89
14	32	9	45	0.72	14	18	5	25	0.72
15	32	5	38	0.84	15	10	4	17	0.60
16	32	4	37	0.88	16	35	4	40	0.89
17	17	15	145	0.12	17	44	5	50	0.89
18	20	8	33	0.60	18	11	2	13	0.82
19	22	6	30	0.73	19	12	1	13	0.92
20	19	6	28	0.68	20	5	0	5	1.00
21	20	2	22	0.90	21	2	0	2	1.00
22	50	6	57	0.88	22	11	0	11	1.00
23	29	5	35	0.83	23	5	2	8	0.60
24	21	14	63	0.33	24	5	0	5	1.00
25	29	12	49	0.59	25	30	9	43	0.70
26	15	5	23	0.67	26	16	2	18	0.88
Average				0.73	27	7	0	7	1.00
					28	14	0	14	1.00
					29	37	13	57	0.65
					30	16	7	28	0.56
					Average				0.83

Table I.5. *S. fontinalis* marked within 100 m of the upstream or downstream end of the 965 m long mark-recapture section in June 1997 but not recaptured in October 1997. Years between captures indicates time from May 1997 to recapture of the fish inside the mark-recapture reach.

Fish #	Mark Location (m)	Recapture Location (m)	Years Between Captures
1	12	12	1
2	12	12	1
3	33	33	1
4	33	190	1
5	65	65	1
6	900	900	1.5
7	910	965	2.5
8	910	910	3
9	920	940	1
10	940	955	1.5
11	940	955	2
12	940	940	1
13	955	955	1.5
14	955	965	1.5

Although this evidence is admittedly tenuous (based on assumptions such as similar recapture rates in June 1997 and June 2000), I believe it provides evidence that mortality is not the only factor affecting low recapture rates in mark-recapture studies. Furthermore, recapture of large numbers of fish in sessions subsequent to their original recapture session and of several individuals outside of the study area in spring 2000 reduced the influence of mortality in the low recapture rates. While mortality certainly contributed to low recapture rates, our data indicated that movement outside of the study area and sampling inefficiency can play larger roles than have been commonly acknowledged.

This mark-recapture study was originally designed to compare growth of fish in the debris flow affected and unaffected areas (Chapter 2), not to examine movement. However, since fish were individually identifiable (PIT tags) and since location of capture was recorded, fish movement could be examined with what amounted to a classic mark-recapture study design. The results of the present study reflect the limitations of mark-recapture studies as outlined in Gowan et al. (1994), and highlight the importance of study design consideration, of maximizing recapture effort, and of proper interpretation of results from similarly flawed study designs.

Telemetry

Seasonal Range

The telemetry study design allowed for examination of the effect of season and of location on the movement and activity of *S. fontinalis* at seasonal and diel scales. Finding smaller ranges during the summer than the fall was not surprising given the findings of past studies, however the difference in range size between seasons was striking. For example, the two fish that were originally tagged during the summer but were recaptured and re-implanted during the fall had summer ranges near 0m, however during the fall they increased their total ranges to 68m and 145m and both occupied new areas of the stream following fall (Figure I.10). This demonstrates the danger in excluding spawning movements in assessment of range (e.g. Gerking 1959). Although summer range may be very small, fall range may be hundreds of meters, to say nothing of winter or spring movements or the sum of movements over a lifetime. This also demonstrates the need for new technology to extend the time that individual fish can be tracked.

Fish moved least at extremes in temperature and discharge and most at more moderate levels, however, low r^2 values indicated a weak relationship. This indicated that although fish were more capable of movements at moderate flows and temperatures, other factors may have ultimately triggered movement. A potentially confounding factor to the interpretation of the flow and temperature data is that moderate flows and temperatures were only experienced during the fall (spawning season). During summer 1999 low flows and high temperatures associated with drought conditions were the norm.

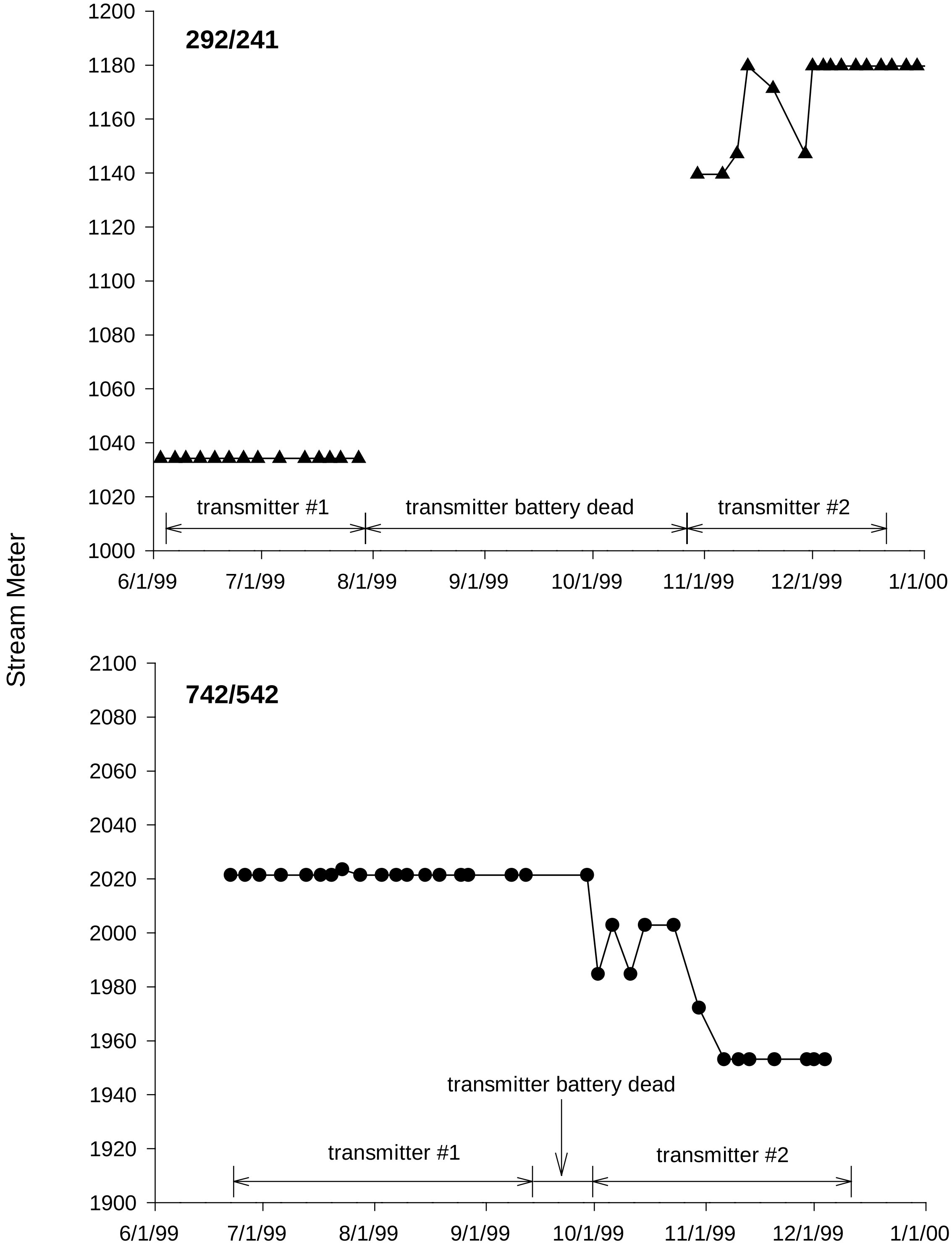

Figure I.10. Range sizes as determined by radio telemetry for two *S. fontinalis* that were originally implanted during summer 1999 and were recaptured and implanted with a second radio transmitter during fall 1999. Numbers on figures indicate the two transmitter numbers for each fish (Table 1a&b). Symbols represent dates that fish were located.

Movement data from summers with moderate flows and temperatures and falls with extremes in discharge and temperature need to be gathered to validate the observed relationships.

Diel Range

Because locating fish once every 3-7 days may not adequately describe range size (Young et al. 1997), examination of diel range was necessary. Median diel movements of less than 15m and a maximum diel movement of less than 50m indicated that daily ranges were typically small. However, it should be noted that most large increases in range size were not made gradually. Long distance movements were often made quickly, within 3-4 days, not as a series of smaller movements over several days or weeks. Despite the fact that some of the fish monitored during diel periods were the same ones that had seasonal ranges up to 1.9km, no fish was observed making a long distance movement during diel observations.

Although the diel range models prevented statistical comparison between seasons diel range size and movements appeared to increase during the fall. Diel ranges in the affected and unaffected areas during the fall were statistically different, but it is doubtful that the difference between ranges of 0m and 16m, which represented the difference between staying in one habitat unit or occupying two adjacent habitat units over a 24-hour period, was biologically significant.

Activity

Significant differences in activity were found between locations and different activity patterns were observed between seasons. *S. fontinalis* are generally regarded as diurnal, but may have peaks in activity near dawn and dusk. A crepuscular pattern was observed during the summer but during the fall the pattern changed, with a single, more sustained peak in late afternoon. Several fish, including some implanted with transmitters were observed spawning during late afternoon. Although activity patterns were similar between locations within a season, activity levels were significantly higher during both seasons in the unaffected area. Controlled experiments are needed to determine if differences in activity levels as measured in this manner could have a significant biological effect (variation in growth, mortality, etc.) (Boisclair and Legget 1989, Marchand and Boisclair 1998).

Temperature had an effect on activity levels during the summer but did not have an effect during the fall. Spawning seemed to be the factor driving activity level during fall whereas feeding and energy conservation were likely the primary factors during summer. During the summer fish were least active when water temperatures were highest, in late afternoon. During the fall temperatures were more homogenous throughout the day and fish were most active in late afternoon. Spawning during the brightest period of the day is logical if these brightly colored fish use visual cues during activities such as mate selection (Rowland et al. 1995).

Significant differences were found between the affected and unaffected areas of the stream for fall diel range and summer diel activity level. When making comparisons about the mean or median, activity levels and range sizes were generally less for the affected area of the stream than the unaffected area. However, for both seasonal and diel movements the maximum range size was always greater in the affected area. Past movement studies have tended to focus on the average movement of fish or fish populations. Maximum movements become equally important when examining topics such as recolonization or metapopulation dynamics.

Conditions created by the debris flow in 1995 appeared to still be having little effect on activity and diel movement of *S. fontinalis* in 1999. The magnitude of the differences between locations was minimal when compared to the differences in activity and range observed between seasons. It seems unlikely that the slight differences in activity and diel range size between locations will have any immediate or lasting effect on the *S. fontinalis* population of the Staunton River.

Comparison of Methods

Lack of movement data from the weirs prevented comparison with mark-recapture or telemetry results, but the mark-recapture and telemetry data both provided estimates of seasonal movements. A fish marked in May and recaptured in October provided an estimate of spring/summer range, and a fish marked in October and recaptured in May provided an estimate of fall/winter range. A comparison of range sizes between all fish recaptured during the mark-recapture study and telemetered fish indicated that larger range sizes could be detected with telemetry (Table I.6). While 54% of fish in the mark-recapture had ranges of 0-10m, only 35% of telemetered fish had 0-10m ranges (over all seasons and locations). Furthermore, only 10% of mark-recapture fish compared with 25% of telemetered fish had seasonal ranges greater than or equal to 90m. However, fish size has been shown to play a role in salmonid movements (Young 1994, Adams et al. 2000) and mark-recapture fish ranged from 100-260mm in length, whereas telemetered fish ranged from 183-283mm. When mark-recapture fish >183mm in length were compared to telemetered fish the distribution of range sizes became very similar (Table I.6). Although this may seem to validate the use of mark-recapture for examining range size, several factors that limit the effectiveness of mark-recapture should be considered.

In addition to the several limitations discussed previously, mark-recapture studies are limited in their ability to detect the magnitude of long distance movements. This is highlighted by comparing the maximum range size detected for fish >183mm using mark-recapture (309m) and radio telemetry (1950m). In addition, it took two and a half years and six mark-recapture sessions to obtain a sample size comparable to the telemetry study, which took less than one year. A direct comparison of range sizes obtained with mark-recapture and telemetry during summer 1999 was not possible because zero fish >183mm were recaptured in fall 1999. Transmitter battery life limitations associated with telemetry also

Table I.6. Comparison of *S. fontinalis* range sizes as observed using mark-recapture (M-R) and radio telemetry techniques in the Staunton River. Percentages indicated in parentheses. M-R >183mm indicates fish >183mm (i.e. similar size to telemetry fish) that were recaptured during the mark-recapture study.

Range Size (m)	M-R All Fish	Cumulative %	M-R >183mm	Cumulative %	Telemetry	Cumulative %
0-10	188 (54)	54	21 (35)	35	17 (35)	35
11-20	42 (12)	66	12 (20)	55	6 (13)	48
21-30	33 (10)	76	6 (10)	66	5 (10)	58
31-40	16 (5)	81	2 (3)	69	6 (13)	71
41-50	10 (2)	83	2 (3)	72	0 (0)	71
51-60	9 (2)	85	3 (5)	75	0 (0)	71
61-70	10 (3)	88	2 (3)	81	1 (2)	73
71-80	2 (1)	89	1 (1)	83	2 (4)	77
81-90	1 (0)	89	0 (0)	83	2 (4)	81
91-100	2 (1)	90	1 (1)	85	2 (4)	85
>100	34 (10)	100	9 (15)	100	7 (15)	100
Sum	347	--	59	--	48	--

hamper the comparison. While fall/winter ranges for mark-recapture fish included movement between October and the following June, battery life limited assessment of range using telemetry to October through December.

Weirs, mark-recapture, and telemetry all have advantages and disadvantages for studying movement. Without careful planning, well defined study goals, and cautious interpretation of results each method can introduce biases (Gowan and Fausch 1996, Gowan et al. 1994). For example, spacing of weirs and location and timing of recapture sessions in a mark-recapture study determine the spatial scale at which movement can be detected. Spatial scale is not problematic in a well designed telemetry study, however, as highlighted earlier telemetry studies are currently limited to a relatively short temporal scale and to only a portion of the total fish population – those large enough to accommodate a transmitter.

Logistics and finances also are major considerations, as all three methods require a high amount of effort and monetary commitment. Weir and telemetry studies require nearly daily maintenance but very few person days, whereas mark-recapture studies focus effort on just a few days but with larger crews. Weir and telemetry studies involve mostly of up-front costs for acquisition of materials and equipment, whereas costs for mark-recapture studies can be spread out over a longer period of time, especially if multiple mark-recapture sessions are planned. Selection of the proper method(s) given research goals and study design is essential to minimizing the biases and logistic and financial concerns associated with each technique.

Challenging the RMP

The present study provides evidence that *S. fontinalis* may be more mobile than previously believed, however does it represent a challenge to the RMP? This depends on how the RMP is interpreted: does it represent a numeric threshold that can be challenged with *a priori* assumptions and statistical tests, or does it represent a mindset that adult stream resident fish are sedentary? If the RMP is thought of in terms of thresholds then we first must decide how to define movement or mobility and what magnitude of movement over a given period of time by individuals or the population as a whole qualifies as a challenge.

Based on the bounds of an arbitrarily chosen threshold, movement of 80% of a population >30m over a 24-hour period, movement of 30% of a population >2km over a year, or movement of 2% of a population several kilometers between streams over the course of a lifetime may or may not represent a challenge to the RMP. Gowan et al. (1994) stated that according to the RMP the range size for adult *S. fontinalis* is approximately 20m, but did not speculate as to what proportion of the population would have to exceed this range over a given time period for the paradigm to be rejected. Setting such *a priori* thresholds is certainly problematic, if not impossible, especially for lesser studied species. Such arbitrarily chosen thresholds are not necessary when movement studies are designed to address a specific question regarding the importance of movement in the research and management of fish.

Experiments should be designed to investigate the roles movement can play at the necessary scale given research or management objectives. For example, understanding movement or activity over diel periods is important in bioenergetics modeling, whereas movement of a population over a year is more important in management of catch and release fisheries adjacent to harvestable waters, and movement of individuals between populations is important in establishing evolutionarily significant units or studying metapopulation dynamics. Studies designed to address specific concerns about fish movement and activity inherently recognize that they are important factors in the biology and ecology of fishes, and not only eliminate problems associated with *a priori* thresholds, but trivialize the RMP as a mindset.

Past movement studies have tended to focus attention on the average movements of individual fish or fish populations. This in part led to the perpetuation of the RMP and the belief that fish populations could be managed as isolated groups in localized areas. Long distance movements and movements of small numbers of fish within the population were discounted as abnormal and unimportant.

The RMP mindset has implications in the management of stream resident salmonids such as the population described in the present study. The headwater reaches and smaller tributaries of streams such as the Staunton River often go dry during drought years and contain relatively poor habitat (low pool:riffle ratio) when compared to areas further downstream and in the mainstem, yet these areas often contain small numbers of fish (or their progeny) that recolonized the reach following the last low flow year. These areas are unstable (subject to drying) in the short term, but their decreased drainage area makes them less subject to the catastrophic floods and debris flows that the mainstem and larger tributaries may experience every 50-100 years (Hack and Goodlet 1960, Lamberti et al. 1991). Movement of fish, even small numbers of fish, into these areas provides an essential source for recolonization and facilitates population persistence in the long term. In addition, long distance movements of individuals from outside of debris flow affected streams into debris flow affected areas can provide an additional source for recolonization. The important implication is that managers must account for movement and provide for movement, even if it is for a relatively small number of individuals, if populations are to persist in these highly variable environments.

Chapter Two:

Effects of a Catastrophic Flood and Debris Flow on Abundance, Density, and Growth of *S. fontinalis* in the Staunton River, Shenandoah National Park, VA.

Introduction

Salmonid populations often live in highly variable environments, which can lead to fluctuations in population abundance and density (Hunt 1969, Reeves et al. 1995, Marschal and Crowder 1996). While it is likely that populations in such environments have adapted to moderate and fairly predictable disturbances, unpredictable, extreme, episodic events can overwhelm individuals leading to elimination of a portion of the population (Reice et al. 1990). Catastrophic events such as debris flows are rare (50-200 year intervals) and pre-event data, knowledge of the timing and severity of the event, and ability to assess effects at sufficient spatial and temporal scales are rarer yet (Hack and Goodlet 1960, Lamberti et al. 1991, Sousa 1984).

Studies examining recovery from catastrophic events have indicated that salmonid populations are resilient to disturbance. Typically, disturbed populations return to pre-event densities within three years (Lamberti et al. 1991, Propst and Stefferud 1997, Swanson et al. 1998) unless the event created unsuitable habitat conditions (Elwood and Waters 1969). When habitat after a catastrophic event is suitable, abundance and density can actually rebound beyond pre-event levels (Lamberti et al. 1991, Thorpe 1994).

Catastrophic events not only cause an immediate and dramatic reduction in abundance and density but also can have effects on growth of fish through impacts on riparian areas. Debris flows can remove trees from the immediate riparian area, opening the tree canopy and increasing solar radiation, primary production, and ultimately the abundance and biomass of macroinvertebrates (Lamberti et al. 1991, Murphy et al.1986, Keith et al. 1998). Decreased density of the fish population in combination with increased food supply creates the potential for increased growth of individuals that recolonize and occupy impacted areas (Sousa 1984), and indeed this phenomenon has been observed in salmonid populations (Lennon 1961, Elwood and Waters 1969, Lamberti et al. 1991, Letcher and Terrick 1998, Swanson et al. 1998).

The Staunton River debris flow presented a unique opportunity to examine the effects of a natural catastrophic event on an *S. fontinalis* population. Pre-event abundance and density data existed (Newman 1996) and knowledge of the timing and severity of the event was available (Karish et al. 1997). Use of both BVET and mark-recapture techniques for a period of four years post-event provided information at spatial and temporal scales rarely examined in the past.

The objective of this study was to examine effects of the June 1995 debris flow on abundance, density and growth of *S. fontinalis*. BVET fish and habitat surveys performed between 1993 and 1999 provided total stream pre- and post-event distribution, abundance, and density data for YOY and age 1+ *S. fontinalis*, and post-event abundance and density data for debris flow affected and unaffected areas. A post-event mark-recapture study provided average length-at-age data for YOY and age 1+ *S. fontinalis* and individual growth data for age 1+ *S. fontinalis* in the affected and unaffected areas between 1996 and 1999.

Methods

Population Abundance and Density

S. fontinalis population abundances were estimated using the basin-wide visual estimation technique (BVET) fish survey (Dolloff et al. 1993). Abundance estimates were made for the total stream in the spring and fall of each year between 1993 and 1999. The total stream BVET data was split into two data sets to calculate abundance for the affected and unaffected areas separately for all surveys after June 1995. BVET fish surveys were performed using a combination of visual and electrofishing methods.

During the BVET fish surveys every 5th pool and every 10th riffle from the confluence to the upper limits of habitability by fish (stream meter 6300) were snorkeled and the number of YOY and 1+ fish in each habitat unit was recorded. Throughout this thesis age 1+ refers to all fish older than YOY. The snorkel counts were calibrated by electrofishing a sub-sample of the units that were snorkeled. Three passes were made through each sub-sample unit using a backpack electofishing unit. The number of YOY and 1+ fish captured in each pass was recorded. The estimated number of fish in each unit was calculated using methods presented in Kwak (1991). Abundance of YOY and 1+ fish in pools and riffles was calculated using methods described in Dolloff et al. (1993).

S. fontinalis population density was estimated by dividing the estimated abundance (as calculated above) by estimated stream area. Stream area was estimated using the BVET habitat survey (Dolloff et al. 1993). During the habitat surveys each habitat unit from the confluence to the upper limits of habitability by fish was measured with a hipchain and the average width of each unit was estimated visually. All habitat units were classified as either pool or riffle (Bisson et al. 1982). Riffles included bedrock slides and step-pool cascades (Dolloff et al. 1993) which made up the majority of riffle habitat in the Staunton River. The actual width of every 5th pool and 10th riffle was measured to calibrate the visual estimates. Stream habitat surveys were made in June 1993, October 1995, and June 1999 (Table II.1).

Population densities for June 1993-May 1995 were calculated using the June 1993 habitat survey. The October 1995 habitat survey was used to calculate post-debris flow densities for October 1995 and May 1996. Densities between October 1996 and November 1999 were calculated using the 1999 habitat survey, as a flood in September 1996 may have caused changes in stream area following the October 1995 habitat survey. Density estimates were made for the debris flow affected and unaffected areas of the stream in addition to the total stream for all surveys following the June 1995 debris flow.

Table II.1. Area (m^2) of the Staunton River in pools, riffles, and total area for the debris flow affected, unaffected and total stream sections in 1993, 1995, and 1999 as estimated using BVET techniques. 95% confidence intervals in parentheses.

Year	Habitat Type	Affected	Unaffected	Total Stream
Jun-93	pools	3570 (±707)	6018 (±588)	9199 (±1822)
	riffles	5383 (±367)	11011 (±1032)	16083 (±1487)
	total	8954 (±546)	17029 (±1196)	25283 (±1041)
Oct-95	pools	2634 (±772)	7222 (±439)	9905 (±876)
	riffles	3409 (±240)	6690 (±635)	10048 (±671)
	total	6044 (±792)	13912 (±761)	19953 (±1095)
Jun-99	pools	3279 (±279)	5136 (±1637)	8565 (±1700)
	riffles	4409 (±1092)	9778 (±1028)	14196 (±1284)
	total	7689 (±1101)	14187 (±1929)	22762 (±2134)

Fish Growth

Average length-at-age and growth of individual fish was examined using data from a post-debris flow mark-recapture study. The mark-recapture area spanned a continuous 965 m reach of the Staunton River, with 575 m in the debris flow affected area and 390m in the unaffected area of the stream (Figure I.3). Trout were first marked for the study in October 1996. Subsequent mark-recapture sessions took place in May 1997, October 1997, June 1998, October 1998, June 1999, and October 1999. Before each session the mark-recapture reach was divided into riffle/pool complexes. Each complex was a continuous 10-40 m reach of stream usually encompassing several pools and riffles and terminating at noticeable breaks such as boulder cascades or small waterfalls. Trout were captured by making a single pass through each complex with an AC backpack electrofishing unit. Fish were checked for marks from previous mark-recapture sessions and their length (mm), weight (g), and location of capture (affected or unaffected area) was recorded. Those fish not having marks were given a PIT tag and adipose clip. Fish less than 100 mm in length (i.e. YOY) rarely received PIT tags. Fish were returned to the riffle/pool complex from which they were captured.

Data from all fish captured during the mark-recapture sessions were used to test for significant differences in average length-at-age between ages (YOY and 1+), sections (affected and unaffected), and dates (Oct 96, Jun 97, Oct 97, etc.) The length data were log-transformed to minimize the effects of unequal variance. When a generalized linear model was applied to the data, significant interaction between date and section and date and age were found ($p<0.001$). The model was split by date, allowing for comparison between sections and between ages within each date. Statistical comparisons between dates could not be made once the model was split.

Data from individually marked (PIT tagged) fish recaptured in the first session after they were marked were used to test for differences in change in length and weight between affected and unaffected sections of the stream and between periods (see Table II.4 for periods). When generalized linear models were applied to the data, significant interaction between date and section and date and age were found ($p<0.001$), thus differences between periods could not be examined. The data were divided into ten groups representing the affected and unaffected sections of the stream during each period. A Levene's test rejected that the variances between the groups were the same ($p=0.003$), thus a Welch's ANOVA was employed to examine for significant differences between groups. Welch's ANOVA assumes unequal variance between groups. The ANOVA results revealed significant differences between the ten groups ($p=0.0001$) and two-sample t-tests were used to examine for significant differences between affected and unaffected sections within each period.

Relationships between change in length and density and change in weight and density were investigated using linear regression. The median change in length and weight from time t_1 to time t_2 (as calculated from mark-recapture data) was plotted vs. density at time t_1 (as calculated from BVET data) and a line was fitted to the data.

Results

Population Abundance and Density

Density (Table II.2) and abundance (Appendix B) estimates varied widely for both YOY and age 1+ *S. fontinalis* before and after the June 1995 debris flow. The most precise estimates (smallest confidence intervals) were obtained for YOY and age 1+ in pools; therefore these data were used to test for trends in density. The highest density estimates for both age groups came after the debris flow (Figures II.1, II.2). The density of both age groups decreased from spring to fall for the affected area in every year following the debris flow, whereas there was no consistent trend in the unaffected area.

Fish Growth

Significant differences in YOY and age 1+ average length-at-age were found between the affected and unaffected areas of the stream from June 1997 to October 1998 (Table II.3). The October 1996 data could not be examined due to significant interaction between age and section. Over the period from June 1997-October 1998, the average lengths of fish in the affected area were consistently higher than those in the unaffected area. Although differences between years could not be compared statistically, there was a decreasing trend in average lengths from 1996 to 1999 for YOY in fall (Figure II.3), age 1+ in spring, and age 1+ in fall (Figure II.4).

Significant differences (α=0.05) in change in length and weight between the affected and unaffected sections were found during periods one and two (Table II.4). Changes in length and weight were greater for the affected area than the unaffected area during these periods (Figure II.5). Although differences between periods could not be statistically compared, there was a decreasing trend in change in length and weight from period 1 to period 5. Changes in length and weight over summer/fall periods (1 and 3) were lower than those during the following winter/spring periods (2 and 4).

The linear regressions for median change in length ($p<0.0001$) and weight ($p=0.008$) vs. density were significant. As density increased both change in length (r^2=0.97) and change in weight (r^2=0.91) decreased (Figure II.6).

Table II.2. Population density (fish/100m^2) by age and habitat type for *S. fontinalis* in the Staunton River, 1993-1999 as estimated using BVET techniques. Estimates for affected and unaffected sections are not available before the June 1995 debris flow. 95% confidence intervals in parentheses.

		1+		YOY	
Date	Section	Pool	Riffle	Pool	Riffle
Jun-93	Total Stream	39 (±25)	13 (±24)	5 (±4)	4 (±8)
Oct-93	Total Stream	10 (±9)	2 (±4)	12 (±5)	2 (±4)
May-94	Total Stream	3 (±5)	2 (±3)	2 (±2)	1 (±1)
Oct-94	Total Stream	7 (±5)	3 (±4)	8 (±8)	3 (±4)
May-95	Total Stream	23 (±9)	3 (±4)	8 (±5)	4 (±6)
Oct-95	Affected	0 (±0)	0 (±0)	0 (±0)	0 (±0)
	Unaffected	6 (±6)	6 (±6)	4 (±4)	1 (±1)
	Total Stream	5 (±4)	4 (±4)	3 (±3)	1 (±1)
May-96	Affected	0 (±4)	0 (±0)	6 (±3)	0 (±0)
	Unaffected	16 (±6)	3 (±8)	33 (±41)	9 (±14)
	Total Stream	11 (±4)	2 (±6)	24 (±25)	6 (±10)
Oct-96	Affected	0 (±0)	2 (±6)	0 (±0)	0 (±0)
	Unaffected	18 (±6)	3 (±6)	14 (±13)	3 (±4)
	Total Stream	13 (±3)	2 (±4)	11 (±7)	2 (±3)

Table II.2. continued.

Date	Section	1+		YOY	
		Pool	Riffle	Pool	Riffle
Jun-97	Affected	17 (±18)	8 (±12)	47 (±27)	57 (±44)
	Unaffected	15 (±6)	2 (±5)	29 (±23)	9 (±3)
	Total Stream	16 (±8)	4 (±5)	37 (±15)	28 (±16)
Oct-97	Affected	2 (±11)	2 (±3)	7 (±8)	4 (±15)
	Unaffected	24 (±46)	4 (±5)	57 (±30)	16 (±21)
	Total Stream	14 (±22)	3 (±3)	35 (±19)	13 (±14)
Jun-98	Affected	62 (±55)	21 (±11)	48 (±67)	15 (±17)
	Unaffected	52 (±76)	na	0 (±0)	0 (±0)
	Total Stream	52 (±31)	9 (±5)	34 (±34)	9 (±9)
Oct-98	Affected	14 (±10)	0 (±2)	45 (±17)	16 (±15)
	Unaffected	78 (±31)	0 (±0)	26 (±26)	4 (±20)
	Total Stream	36 (±19)	10 (±8)	15 (±22)	10 (±4)
Jun-99	Affected	36 (±16)	0 (±0)	34 (±27)	36 (±8)
	Unaffected	40 (±34)	9 (±13)	5 (±19)	8 (±8)
	Total Stream	52 (±16)	3 (±7)	35 (±13)	9 (±12)
Nov-99	Affected	21 (±6)	2 (±2)	11 (±6)	3 (±7)
	Unaffected	22 (±23)	1 (±4)	11 (±21)	2 (±8)
	Total Stream	22 (±10)	2 (±2)	11 (±7)	3 (±5)

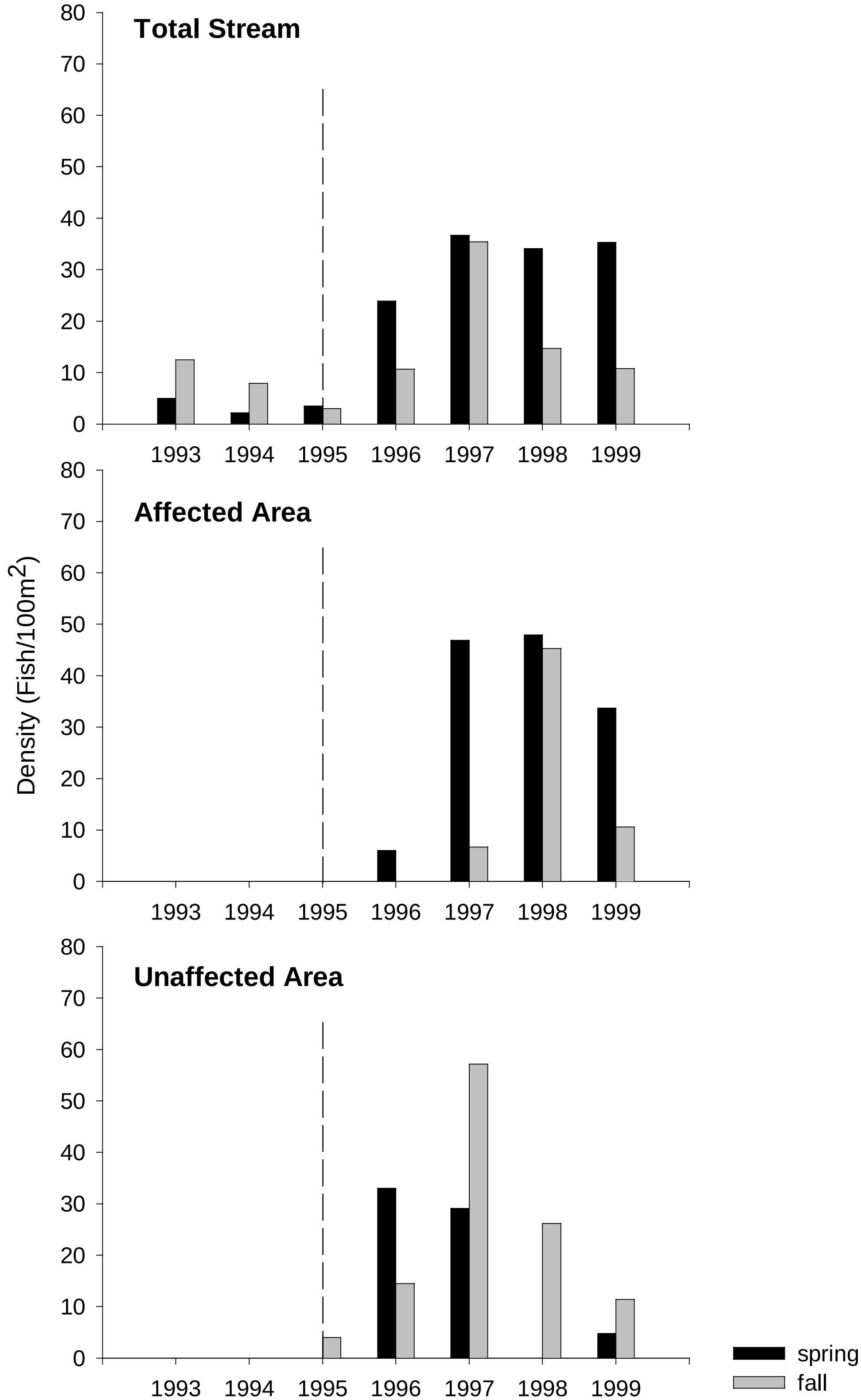

Figure II.1. Density of YOY *S. fontinalis* in pools in the Staunton River by season in the total stream, affected area, and unaffected area as estimated using BVET techniques from 1993-1999. Dashed line indicates date of debris flow.

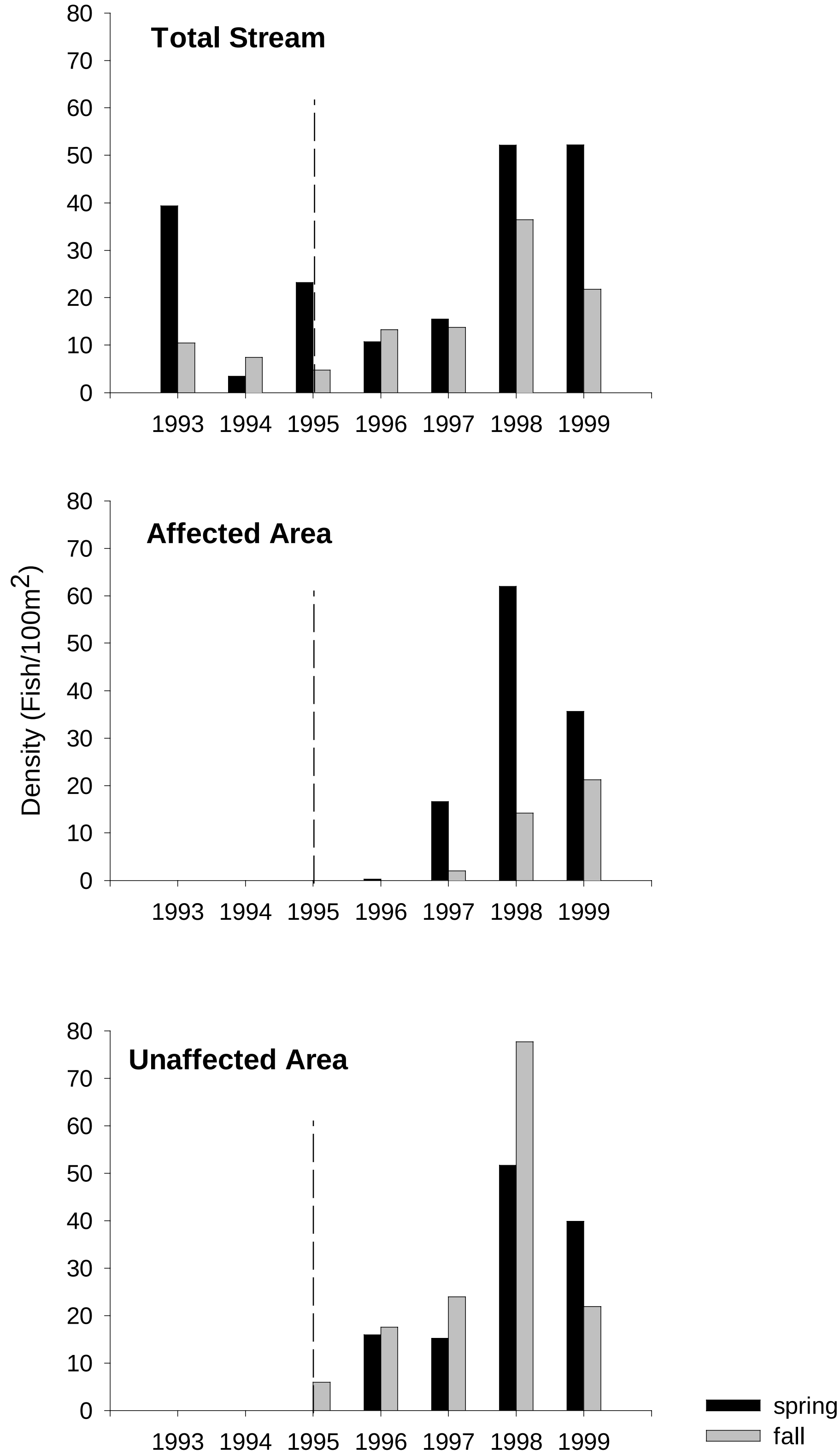

Figure II.2. Density of age 1+ *S. fontinalis* in pools the Staunton River by season in the total stream, affected area, and unaffected area as estimated using BVET techniques from 1993-1999. Dashed line indicates date of debris flow.

Table II.3. Average length-at-age by age and section as determined from mark-recapture data for *S. fontinalis* in the Staunton River, 1996-1999. Sample size in parentheses. Asterisk indicates significant difference between affected and unaffected areas ($p<0.001$), from generalized linear model. Affected and unaffected sections not compared in October 1996 due to interaction between section and age.

	YOY		1+	
Date	Affected	Unaffected	Affected	Unaffected
Oct-96	110 (135)	99 (88)	166 (13)	207 (18)
Jun-97	49 (216)	46* (339)	167 (167)	154* (128)
Oct-97	92 (577)	78* (684)	184 (95)	166* (128)
Jun-98	63 (166)	59* (56)	146 (486)	131* (363)
Oct-98	78 (272)	76* (65)	143 (415)	128* (346)
Jun-99	57 (493)	59 (149)	143 (427)	139 (213)
Oct-99	73 (220)	73 (66)	138 (186)	133 (106)

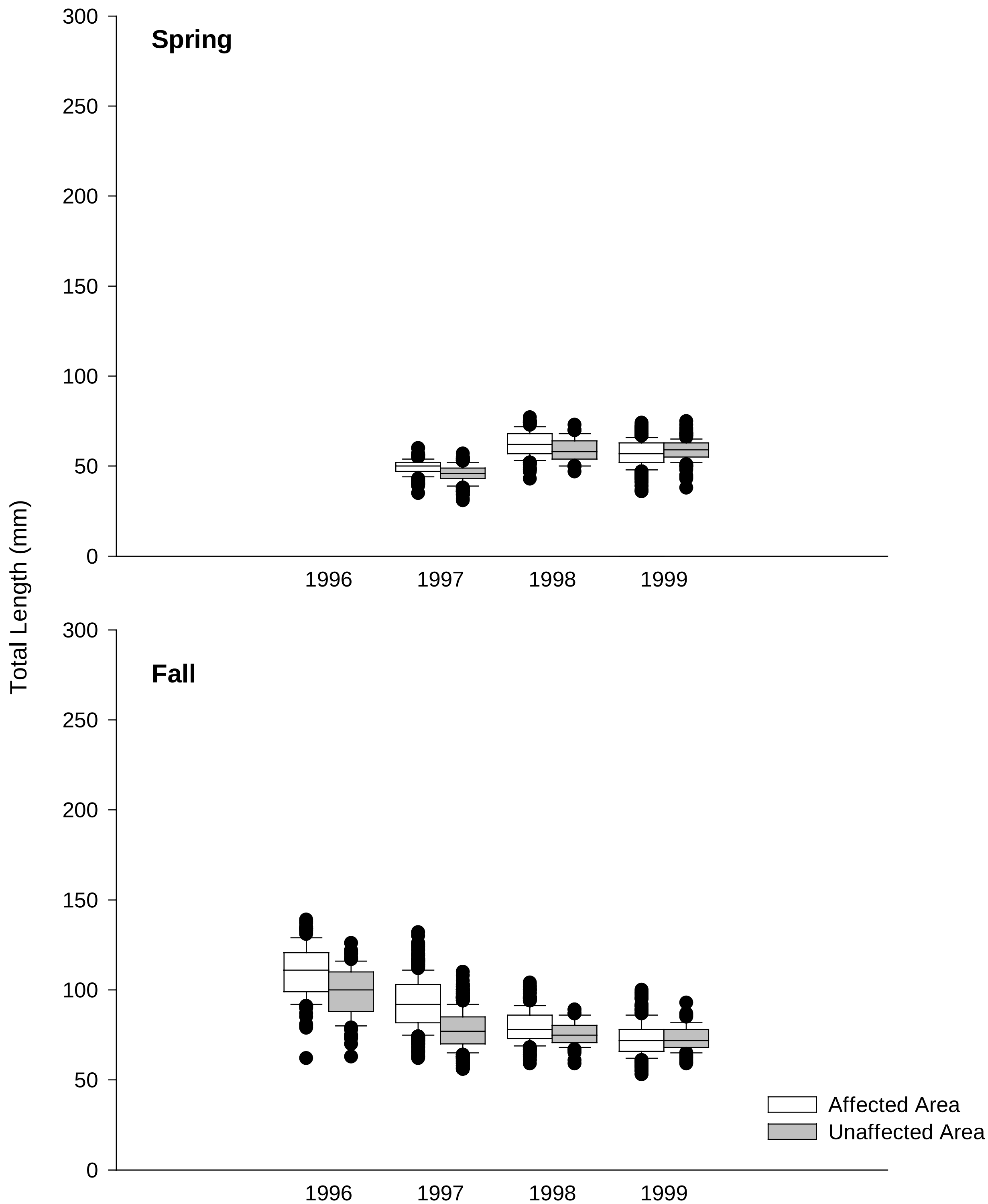

Figure II.3. Length-at-age of YOY *S. fontinalis* in Staunton River in spring and fall, from mark-recapture data. Boxplots represent 25th and 75th percentiles and median value. Whiskers represent 10th and 90th percentiles. Closed circles represent entire range of data.

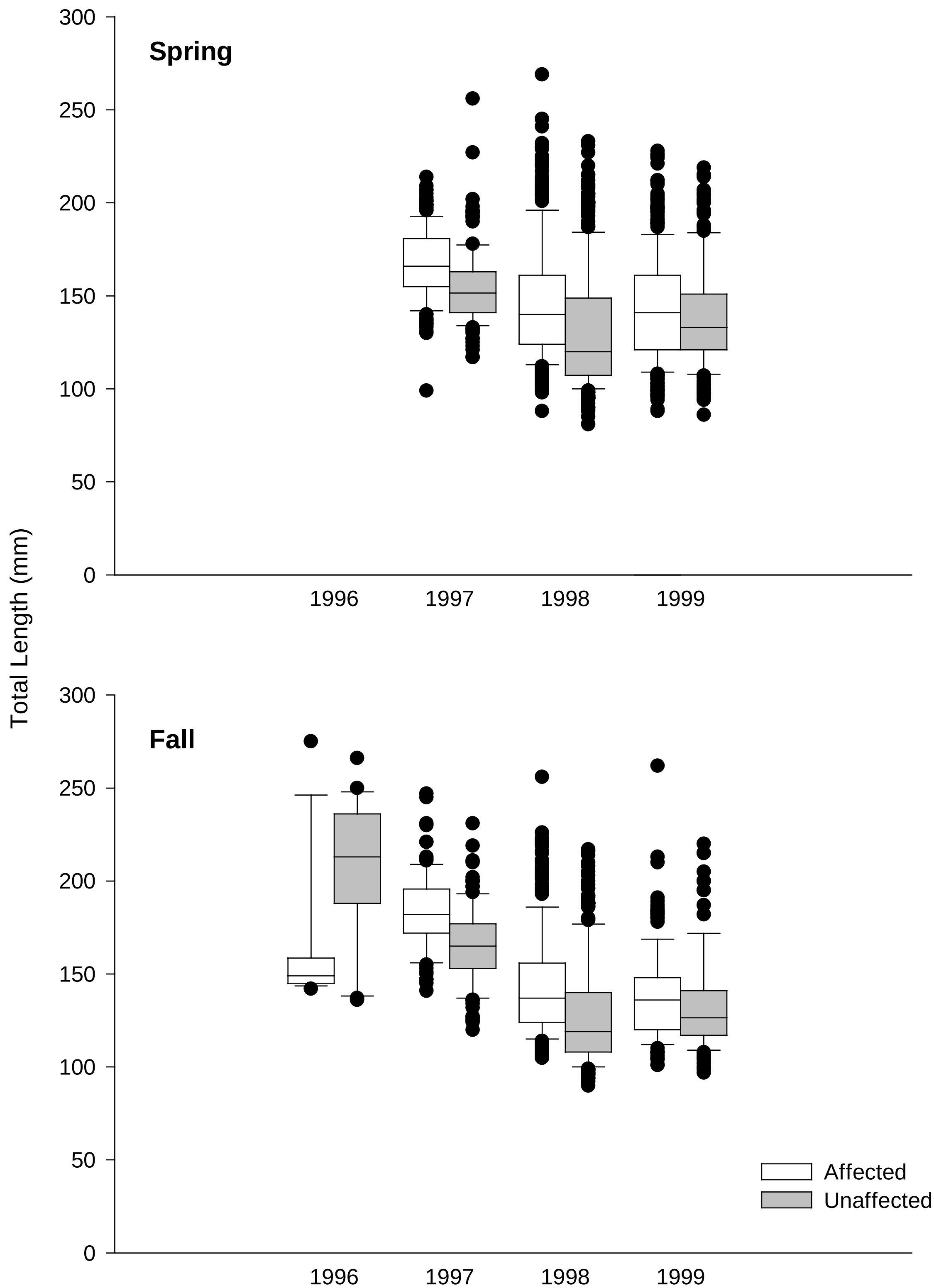

Figure II.4. Length-at-age of age 1+ *S. fontinalis* in Staunton River in spring and fall, from mark-recapture data. Boxplots represent 25th and 75th percentiles and median value. Whiskers represent 10th and 90th percentiles. Closed circles represent entire range of data.

Table II.4. Mean change in length (mm) and weight (g) for individual *S. fontinalis* marked with PIT tags and recaptured in the first period after marking. Sample size in parentheses. Asterisk indicates significant difference between affected and unaffected area, two-sample t-test, α =0.05

		Length (mm)		Weight (g)	
Period	Date	Affected	Unaffected	Affected	Unaffected
1	Jun-97 to Oct-97	18 (11)	11* (8)	14 (11)	4* (8)
2	Oct-97 to Jun-98	27 (48)	11* (22)	17 (48)	10* (22)
3	Jun-98 to Oct-98	0 (70)	0 (23)	-5 (70)	-7 (23)
4	Oct-98 to Jun-99	8 (63)	9 (24)	3 (63)	4 (24)
5	Jun-99 to Oct-99	0 (39)	-2 (29)	-3 (39)	-4 (29)

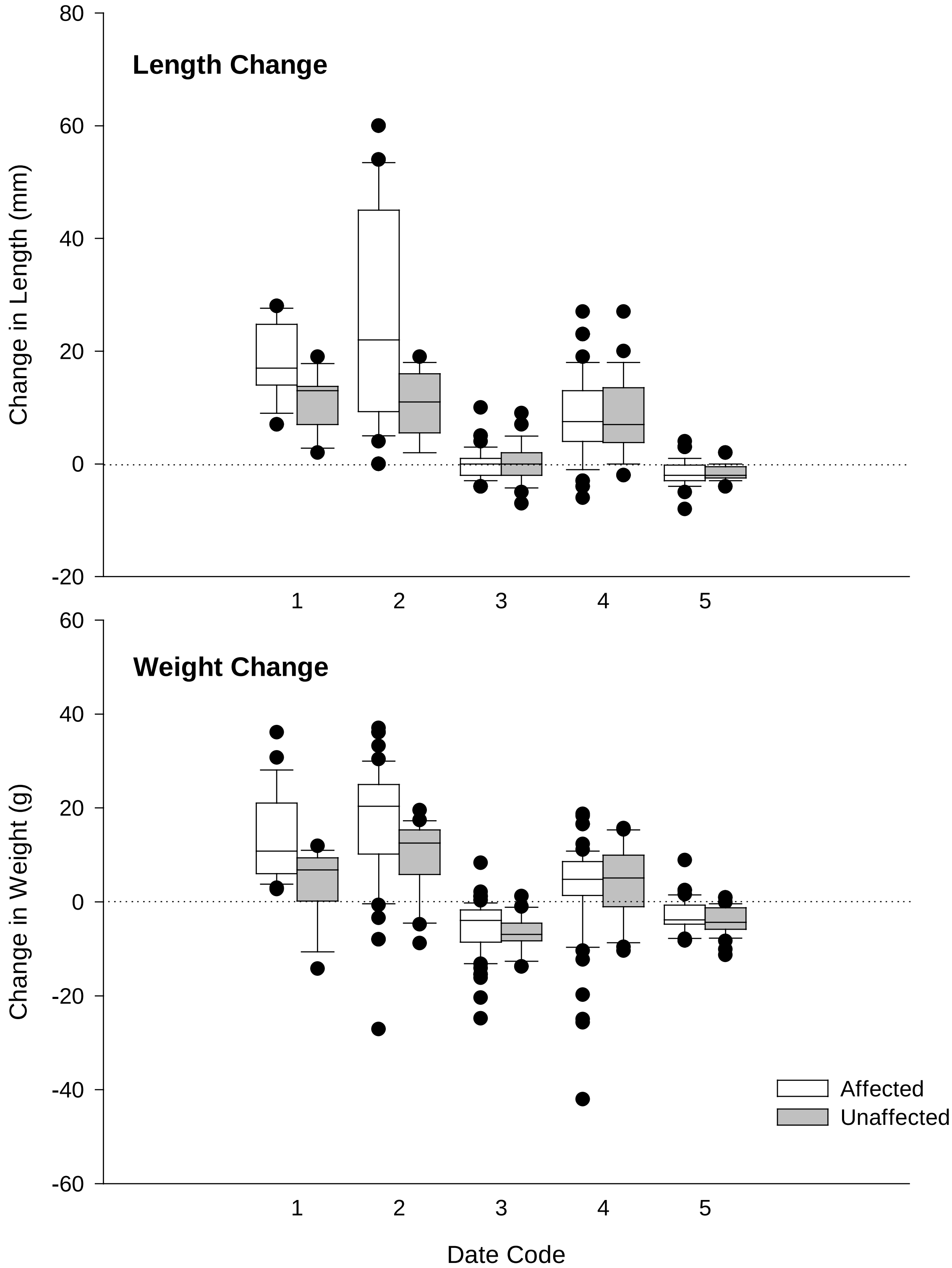

Figure II.5. Change in length and weight of age 1+ *S. fontinalis* recaptured in the Staunton River during the mark-recapture study. Boxplots represent the 25th and 75th percentiles and median value. Whiskers represent the 10th and 90th percentiles. Closed circles represent the entire range of data. See Table II.4 for definition of date codes.

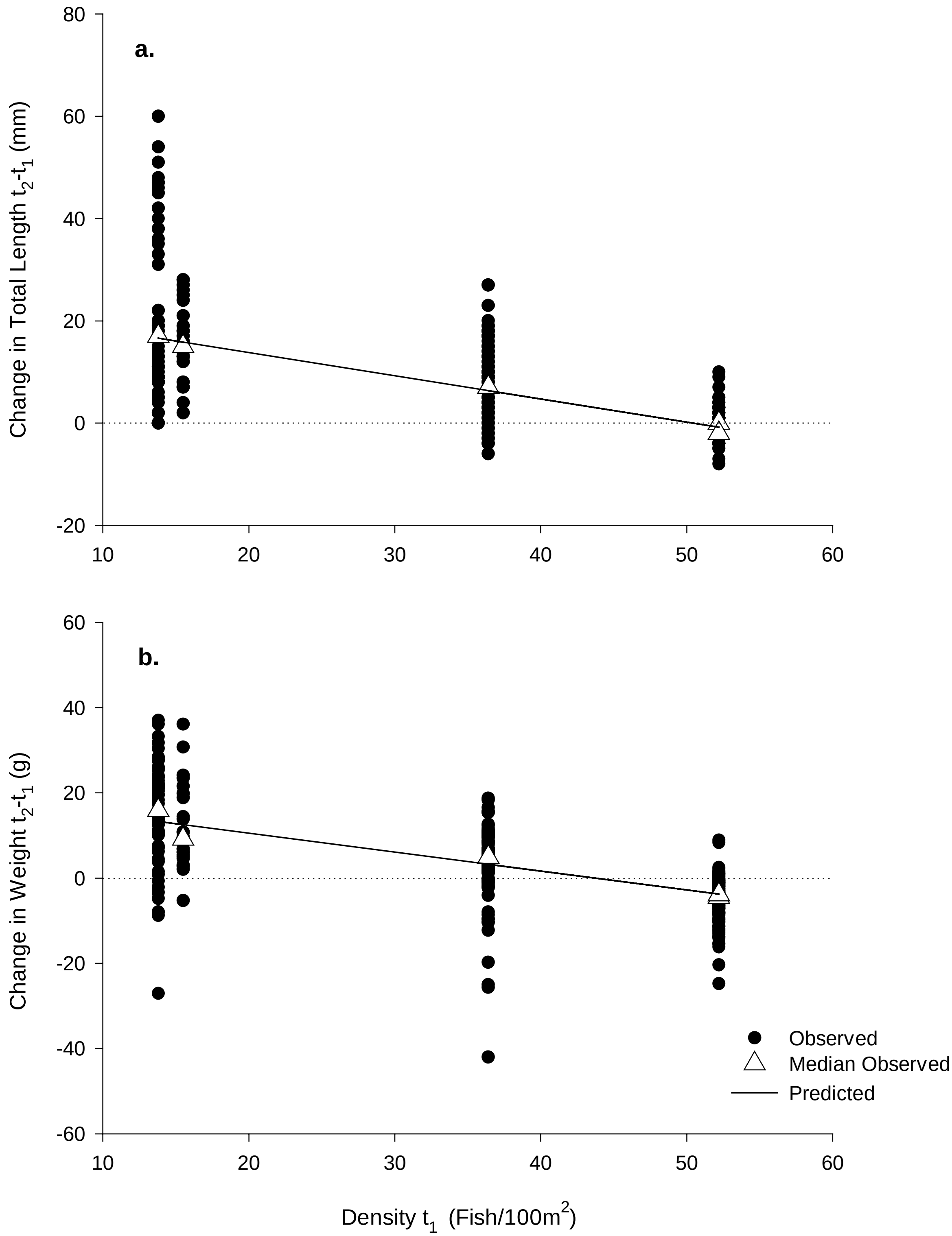

Figure II.6. Change in a. individual fish length ($p<0.0001$, $r^2=0.99$, and b. individual fish weight ($p=0.008$, $r^2=0.91$) vs. population density for age 1+ *S. fontinalis* in the Staunton River. Closed circles represent the entire range of data, triangles represent median change, solid line represents linear regression, and dotted lines represent zero change.

Discussion

Population Abundance and Density

The *S. fontinalis* population of the Staunton River responded quickly and vigorously following the June 1995 debris flow. In the stream as a whole, YOY abundance and density exceeded pre-debris flow levels within 1 year and age 1+ abundance and density exceeded pre-debris flow levels within 2.5 years. Abundance and density of both YOY and age 1+ fish remained above pre-debris flow levels for each observation period after exceeding pre-debris flow levels. Further study is currently underway to determine if population levels will remain elevated (U.S. Forest Service, unpublished data).

Future fluctuations in abundance and density are likely given that *S. fontinalis* are known to have highly variable survival because of the unstable nature of the streams they occupy (Elwood and Waters 1969, Hunt 1969, Marschall and Crowder 1996, Reice et al. 1990). The unstable nature of stream environments has led to the assertion that stream fish populations may essentially be in a constant state of recovery from disturbance (Reice et al. 1990). Given that *S. fontinalis* evolved in such systems, the rapid response of the population following the debris flow should not have been surprising and suggests a mechanism that has developed for responding to catastrophic events.

While natural disturbances and the fluctuations in abundance they cause may seem to harm populations in the short-term, natural stream disturbance/recovery cycles are essential for maintaining productive salmonid habitat in the long-term (Reeves et al. 1995). For example, the number of salmonid eggs surviving to emergence from gravel is known to increase as the amount of fine substrate materials in redds decreases (Reice et al. 1990, Kondolf 2000). In the Staunton River the proportion of pools with a dominant substrate of sand, silt, or clay decreased following the flood and debris flow (Chapter 3). The flood and debris flow provided unembedded spawning substrate with decreased fine substrates, a combination that should benefit *S. fontinalis* reproductive capabilities in the Staunton River. However, these conditions do not guarantee increased fish abundance, as factors such as those described below may affect YOY and age 1+ survival and produce population fluctuations.

While the population as a whole increased following the debris flow, differing patterns were observed in the affected and unaffected areas. From spring to fall, YOY and age 1+ abundance and density decreased each year in the affected area, whereas they increased from spring to fall for the majority of observations in the unaffected area. While movement from the affected to the unaffected area could help to explain the different patterns, a movement study found little movement between the affected and unaffected areas during summer (Chapter 1). During the fall, discharge was generally lower and fish size was larger than in the spring, likely increasing both snorkeling and electrofishing efficiency. This would explain the pattern observed in the unaffected but not the affected area.

Lower survival in the affected area, especially for YOY fish, seems a more likely explanation. Higher mortality in the affected area could offset increased sampling efficiency during the fall. Increased mortality of YOY in the affected area would also explain why density of 1+ fish in the affected area was not higher than in the unaffected area despite higher densities of YOY in the affected area each spring after 1996 (Figures II.1 and II.2). Increased density of YOY salmonids has often been observed when tree canopies over streams are opened (Murphy et al. 1986, Keith et al. 1998), however the density increase can be negated through increased mortality if suitable seasonal habitats are lost (Murphy et al. 1986, Lambert et al. 1991). Water temperatures often neared the 25C lethal limit for *S. fontinalis* (Biro 1998) during summer 1999 in the affected area while remaining 2-4C cooler in the unaffected area (Appedix A).

Fish Growth

Conditions created by the debris flow may have continued effects on abundance and density, however other effects appeared shorter lived. Differences in average length-at-age and change in length and weight of individuals between the affected and unaffected areas became negligible 3-4 years post-debris flow. While increased growth rates have been observed for YOY and age 1+ salmonids occupying disturbed areas (Elwood and Waters 1969, Lamberti et al. 1991, Letcher and Terrick 1998), decreased growth rates over time have not been documented. In these studies increased growth was generally attributed to increased water temperature, increased food supply, and/or decreased competition due to low population density following the event. It does not seem likely that increased water temperatures would benefit a salmonid population in Virginia. Data on the abundance of macroinvertebrates in the affected and unaffected areas of the stream were being collected by the USGS-BRD but were not available at the conclusion of the present study and it was beyond the scope of the study to investigate differential consumption rates between the affected and unaffected area. However, a strong negative relationship was found between growth and density, which could support either the increased food supply or decreased competition hypotheses.

Density-dependent growth of *S. fontinalis* has been observed in field (Hunt 1969) and laboratory studies (Marchand and Boisclair 1998), and has been examined with modeling (Marschall and Crowder 1996). Whether increased growth at low density is the result of increased food consumption or decreased agonistic interactions remains a point of debate (Marchand and Boisclair 1998). That a relationship between density and growth existed in the Staunton River population was not as surprising as the strength of the relationship. The post-event data provided a rare look at growth during periods of extremely high and low densities for the Staunton River. Further study is needed to examine the strength of the density dependence, especially at moderate population densities.

Growth of individual fish not only decreased as density increased but also showed seasonal trends. Growth of fish during summer/fall (June-October) periods was lower than during the following winter/spring (October-June) period (Table II.4, Figure II.5). Past studies indicate that *S. fontinalis* display the most growth between February and June and show little growth between June and October (Bridges 1972, Meyer 1990, Shetter 1937, Whitworth and Strange 1983). The current study indicates that *S. fontinalis* typically grow most during winter/spring, although under certain conditions (very low density) may show substantial growth during summer/fall.

Natural disturbances often create dramatic changes in the landscape and cause sudden changes in fish populations leading biologists to describe the events as 'catastrophic'. The present study suggests that these events can be little more than a minor setback given proper conditions for recovery (e.g. viable source population, intact habitat). The rate at which YOY and age 1+ abundance and density recovered following the debris flow demonstrated the resiliency of *S. fontinalis* populations to natural disturbances. This resilience should preclude the immediate use of stocking as a method to supplement impacted populations. The only management action taken following the debris flow on the Staunton River was to close the fishery until the population recovered to sufficient levels. However, the population likely would not have recovered as quickly had a source population not been available upstream of the debris flow.

Increased growth immediately following the debris flow and its relationship to density seems to indicate that suppressing population size could alter the fishery by providing fewer, but faster growing and ultimately larger fish. This management approach has been examined for lentic *S. fontinalis* with similar results to the present study. Donald and Alger (1989) found that reducing the number of fish in a population increased fish growth. However they noted that the increase in size would likely be of minimal importance to anglers. The same could be said of the present study. When density was lowest, in 1997, mean length-at-age for 1+ fish ranged from 154mm to 184mm (Table II.4). When density was highest, in spring 1998 and spring 1999, means ranged from 128mm to 143mm. During the entire period the maximum fish size remained near 250mm (Figure II.6). It is doubtful that changes in the population size structure related to density would substantially affect the quality of fishing for the typical angler.

Although the differences in growth observed over time or between affected and unaffected areas may be of little importance to the typical angler they are of importance to ecologists. The observed increase in growth, particularly in YOY, immediately following the flood could decrease age-at-maturity or increase fecundity (Hutchings 1996) and suggests a mechanism that has developed to allow the population to recover quickly from severe episodic disturbances. While the observed changes in the population provided information on recovery in the short-term, further study is needed to uncover long-term effects on abundance, density, and growth. Long-term data will provide a clearer picture of how disturbances shape the populations that occupy variable environments.

Chapter Three:

Changes in Instream Habitat Immediately Following and Four Years After a Catastrophic Flood and Debris Flow on the Staunton River, Shenandoah National Park, VA.

Introduction

Physical disturbance to stream habitats is a common effect of flooding. Floods can alter large woody debris (LWD) loadings, substrate composition, and channel morphology (Dolloff et al. 1994). In some cases these changes can create unsuitable habitat conditions for fish species occupying impacted areas (Elwood and Waters 1969). When heavy precipitation occurs in mountainous areas water in the stream channel can combine with water, soil, LWD, and rocks from landslides, creating a debris flow that can be transported down the stream channel at up to 10m/s (Hack and Goodlet 1960, Lamberti et al. 1991, Swanson et al. 1998).

Debris flows can drastically alter both stream channel and riparian zone. Movement of a slurry of water, soil, rocks, and LWD through a stream channel can remove trees and vegetation from riparian areas, alter LWD distribution, scour the channel to bedrock, and deposit new substrate materials (Gecy and Wilson 1990, Lamberti et al 1991, Swanson et al. 1998). Changes in the stream channel and the surrounding landscape immediately following extreme flooding and debris flows can be dramatic, however lack of pre-event data often precludes any meaningful assessment of initial impacts on instream habitat (Lamberti et al. 1991, Dolloff et al. 1994) or longer-term changes due to channel altering flows following the events.

The objective of this study was to document instream habitat changes following the June 1995 debris flow on the Staunton River. Habitat in the Staunton River was inventoried using BVET techniques in 1993, and post-event BVET surveys took place in 1995 and 1999. The data from these three surveys provided a rare opportunity to examine changes in channel morphology, LWD loading, and substrate composition both immediately and four years after a catastrophic event.

Methods

Habitat in the Staunton River was inventoried using visual estimation methods described in Hankin and Reeves (1988). In this study, surface area, maximum depth, LWD loading, and substrate were estimated in all habitat units from the confluence with the Rapidan River to the upper limits of habitability by fish (stream meter 6300). Habitat units were limited to pools and riffles as described in Bisson et al. (1982). Riffles included bedrock slides and step-pool cascades (Dolloff et al. 1993) which comprised the majority of riffles in the stream.

Habitat surveys were performed by two-person crews, with individual #1 responsible for estimating habitat features and individual #2 responsible for measuring habitat unit length and recording data. The crew began at the downstream end of the first habitat unit and progressed upstream with individual #1 identifying habitat units by type (pool or riffle), and estimating the width of each unit, and individual #2 measuring the length of each unit. The width of 20% of the pools and 10% of the riffles was measured with a meter tape to calibrate visual estimates. Stream area estimates were calculated following the methods outlined in Dolloff et al. (1993). Maximum depth of each unit was estimated by individual #1 using a wading staff marked at 5cm intervals.

Dominate substrate (substrate covering greatest percentage of stream bottom) and LWD loading also were estimated for each habitat unit. Substrates were divided into seven classes: organic debris, small diameter substrate (included clay, silt, sand), gravel, pebble, cobble, boulder, and bedrock. LWD included all pieces >1m in length and 5cm in diameter. LWD was assigned to one of seven size classes based on length and diameter (Table III.1).

Differences in discharge can affect estimates of habitat features (Herger et al. 1996, Hildebrand et al. 1999). During the 1993 and 1999 habitat surveys discharge on the Staunton River was monitored with a stream gauge located 140m upstream from the confluence with the Rapidan River (Figure I.4). Discharge during the 1995 survey was calculated by correlation from data collected at nearby gauging stations, as the 1995 debris flow had destroyed the Staunton River gauge. Personnel from the Department of Environmental Sciences at the University of Virginia, Charlottesville, VA, maintain the gauge.

Table III.1. Classification of large woody debris in the Staunton River.

	Diameter (cm)		
Length (m)	5 to 10	11 to 50	over 50
1 - 5	Class 1	Class 2	Class 3
Over 5	Class 4	Class 5	Class 6

Results

Discharges during the 1993 (0.14 m^3/s) and 1995 (0.16 m^3/s) habitat surveys were not significantly different ($p>0.30$), however the 1993 and 1995 surveys were significantly different from the 1999 survey ($p<0.001$), when drought conditions resulted in an average discharge of .03 m^3/s. In past studies, decreased discharge led to increased number of habitat units, decreased average surface area, and decreased total stream area (Herger et al. 1996, Hilderbrand et al. 1999). In the present study the opposite effect was observed between 1995 and 1999, indicating that any effect decreased flow may have had on estimates was overridden by large changes in channel morphology.

The total number of habitat units increased by more than 50% in 1995, immediately following the flood and debris flow (Table III.2). The increased number of units was reflected in a 50% decrease in average surface area for both pools and riffles in debris flow affected and unaffected areas of the stream. Maximum depth of habitat units also decreased following the debris flow, most obviously in the debris flow affected area where they decreased by more than 25%. Total number of habitat units, average surface area, and maximum depth had all returned to near pre-event levels by 1999.

Total stream area and area in pools and riffles was not significantly different during the 1993 and 1999 habitat surveys (Figure III.1). Significant differences in riffle area and total stream area were found when the 1995 survey was compared with both the 1993 and 1999 surveys. Total pool area was not significantly different between any of the surveys.

More than 25% of pre-event habitat units had substrates dominated by small diameter substrate particles with other size classes evenly distributed through the remainder of habitat units. Large size class cobbles and boulders (size 4&5) dominated post-event substrates (Figure III.2). Substrate composition in the debris flow affected and unaffected areas were nearly identical during all three surveys. In 1999, the percentage of habitat units with a dominant substrate of cobble decreased substantially as the percentage with gravel substrate increased more than three-fold, returning to pre-event levels.

Overall, the amount of LWD per kilometer increased in both the debris flow affected and unaffected areas of the stream in 1995 (Figure III.3). Small diameter LWD (sizes 1&4) showed marked increases in both sections, as did large diameter LWD (sizes 3&6), with the exception of size 6 in the unaffected area. The amount of medium diameter LWD (sizes 2&5) per kilometer did not increase in 1995.

In 1999, the amount of LWD per kilometer continued to increased in the debris flow unaffected area, but decreased in the debris flow unaffected area (Figure III.3). Small (sizes 1&4) and large (sizes 3&6) diameter LWD decreased substantially in the debris flow affected area, whereas medium diameter

LWD increased. The amount of LWD per kilometer remained approximately the same or increased for all size classes except for short, large diameter LWD (size 3) in the debris flow unaffected area.

Table III.2. Number of habitat units, estimated average surface areas of individual pools and riffles, and average maximum depths in the Staunton River before and after the June 1995 debris flow. Affected section in 1993 represents the lower 1.9km of the stream. Total is pools and riffles combined.

	Pools			Riffles			Total		
Section	1993	1995	1999	1993	1995	1999	1993	1995	1999
					Number of habitat units				
Affected	107	176	105	86	121	76	193	297	181
Unaffected	211	328	190	178	259	166	389	587	356
Total Stream	318	504	295	264	380	242	582	884	537
					Average surface area of individual habitat units (m^2)				
Affected	43.0	22.0	38.6	75.9	36.9	55.7	118.9	58.9	94.3
Unaffected	21.8	18.0	23.7	53.7	25.8	60.0	75.5	43.8	83.8
Total Stream	29.0	19.4	29.0	60.9	29.4	58.7	89.9	48.8	87.7
					Depth (cm)				
Affected	70.9	52.4	58.0	34.2	24.8	35.5	105.2	77.2	93.5
Unaffected	64.5	56.6	57.6	23.6	26.5	30.8	88.1	83.1	88.5
Total Stream	66.7	55.1	57.7	27.1	25.9	32.3	93.7	81.1	90.1

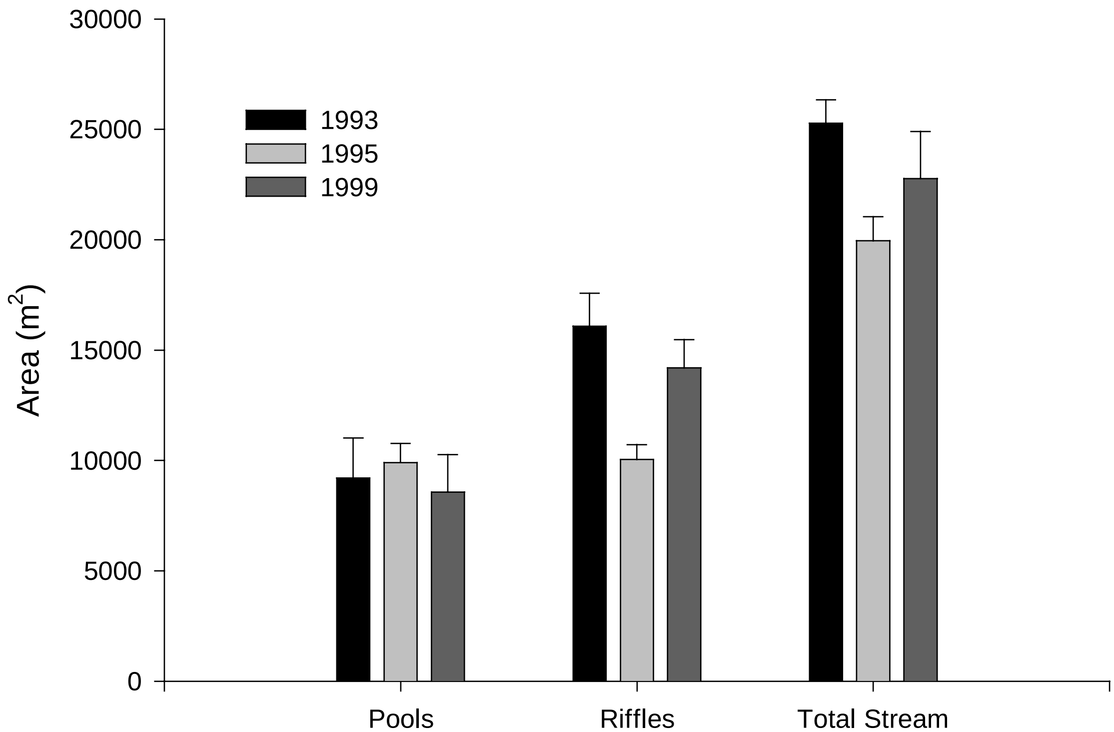

Figure III.1 Total areas (and 95% confidence intervals) of habitats before (1993), immediately following (1995), and four years after (1999) a flood and debris flow on the Staunton River.

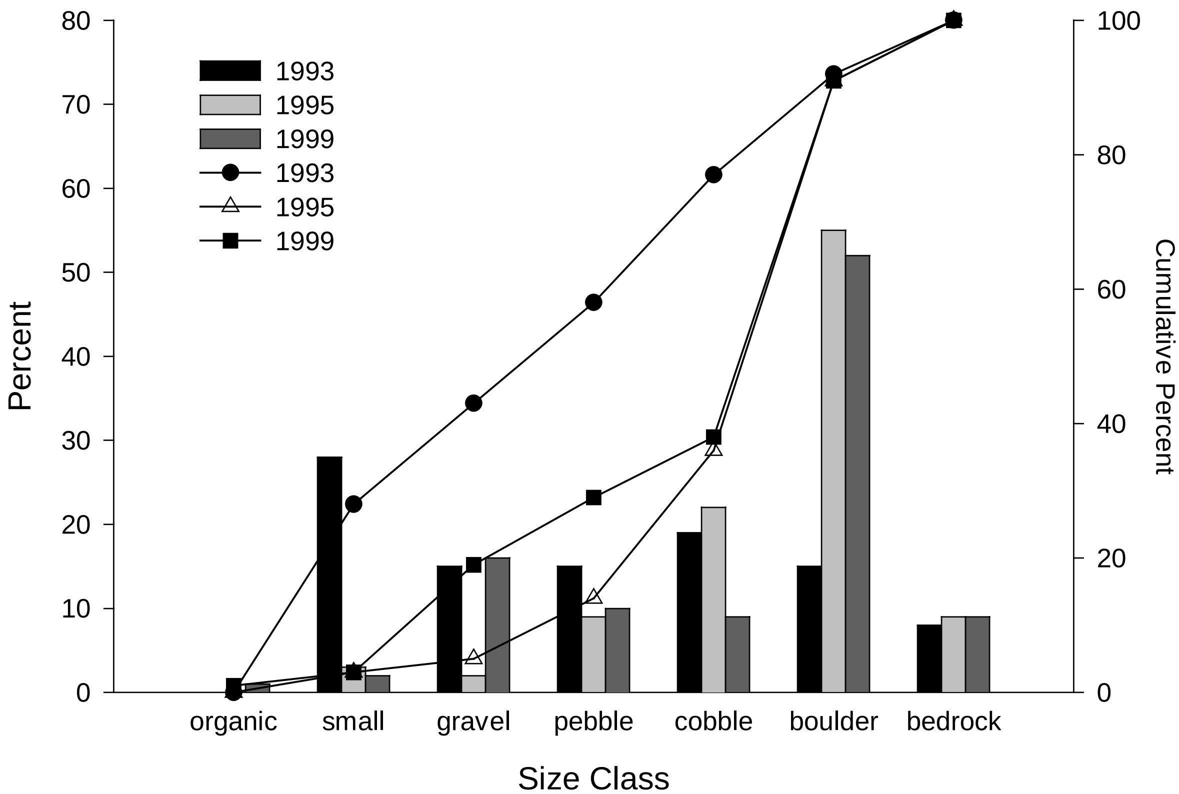

Figure III.2. Percentages (bars) and cumulative percentages (lines) of occurrence of dominant substrates in the Staunton River before (1993), immediately following (1995), and four years after (1999) the June 1995 flood and debris flow.

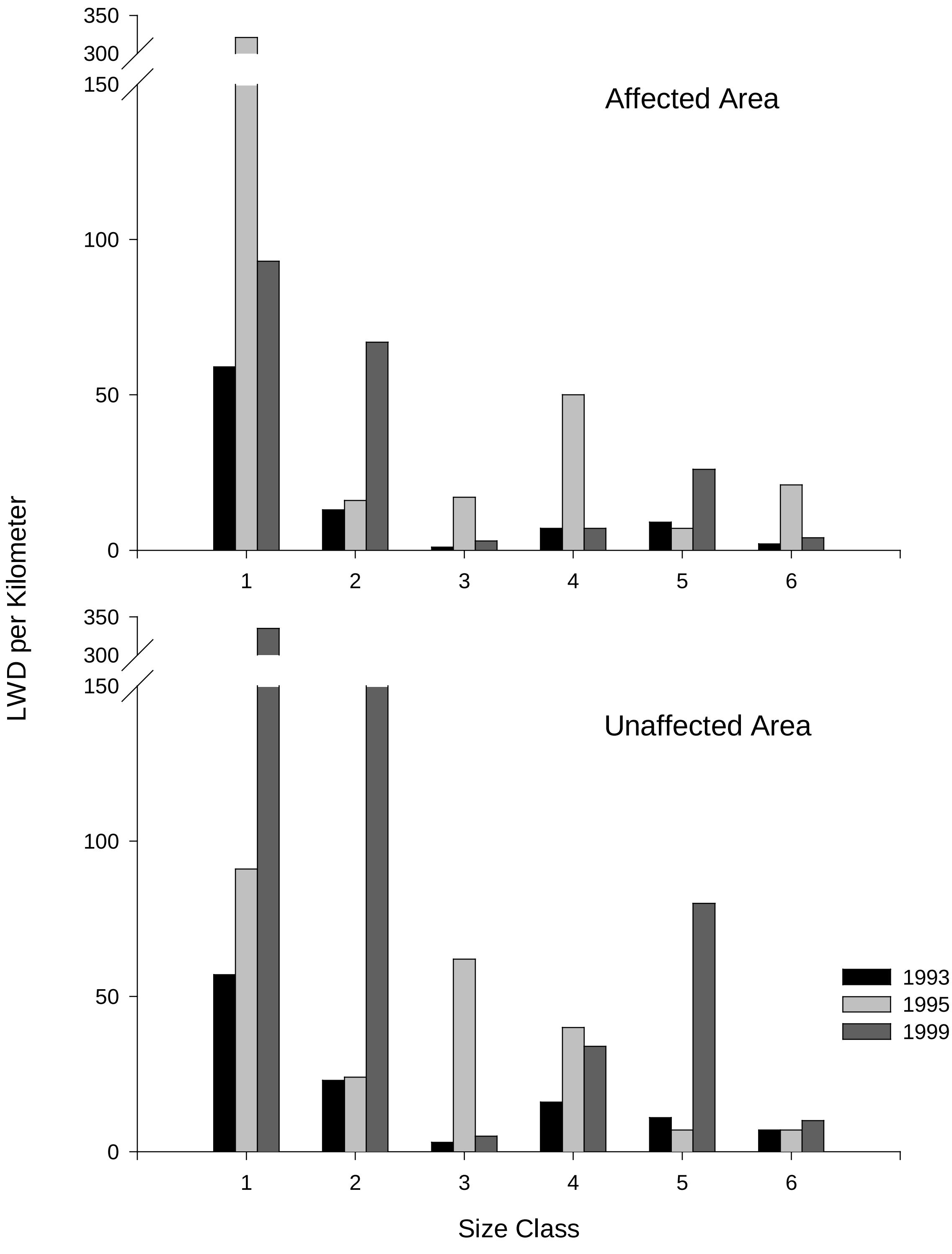

Figure III.3. Large woody debris per kilometer in debris flow affected and unaffected areas of the Staunton River before (1993), immediately following (1995), and four years after (1999) the June 1995 flood and debris flow. Affected area in 1993 represents the lower 1.9km of the stream.

Discussion

Persons performing the 1995 BVET habitat survey reported that the stream channel was highly disordered, making it difficult to distinguish the main channel (M. K. Underwood, pers. comm.). The highly disordered nature of the channel was reflected in the increased total number of habitat units and decreased average surface area during the 1995 survey. The immediately post-event Staunton River consisted of a large number of small surface area pools, which served to break riffles into small sections, effectively decreasing their average surface area also. Decreased total surface area of riffles in 1995 was also the result of the disordered channel. Water flowed in several small channels between pools instead of through a single main channel, resulting in a decreased estimate of total riffle area and total stream area in 1995. By 1999, the stream channel had become more organized, with a single main channel in most areas, and the total number of habitat units, average surface area, and total stream area had returned to pre-event levels.

The reorganization of the stream channel between 1995 and 1999 demonstrated the importance of discharge events in channel adjustments (Reice et al. 1990). Information on the discharge levels needed to move substrates of differing sizes in the Staunton River is sparse, however several high discharge events were experienced in the four years after the flood and debris flow (Table III.3). The two largest events were in 1996 (>8.50 m^3/s) and 1998 (>2.83 m^3/s). The June 1995 flood was estimated at 7.45 m^3/s, although this is known to be an underestimate (K. Reinhardt, University of Virginia, pers. comm.) and base flow ranged between 0.03m^3/s and 0.16m^3/s. The high discharge events between 1995 and 1999 altered channel morphology and returned several of the measured variables to near pre-event conditions.

Floods and debris flows have the ability to scour streams to bedrock (Swanson et al. 1998) and change dominant substrate size (Dolloff et al. 1994, Elwood and Waters 1969). The increase in dominant substrate size observed in the Staunton River following the debris flow was not surprising given that smaller size substrate particles are more easily transported downstream during high water events (Reice et al. 1990). The shift towards large substrate sizes could have been perceived as detrimental if smaller size substrate used for spawning became limited (Staunton River *S. fontinalis* typically spawn on gravel or pebble substrates). Within three years of the flood and debris flow *S. fontinalis* population abundance and density in the Staunton River recovered to pre-event levels (Chapter 2), indicating that suitable spawning habitat was not a limiting factor. In fact, the flood may have benefited *S. fontinalis* by decreasing the embeddedness of spawning substrates and removing fines (smooth substrate materials). By 1999 particle sizes had begun to shift towards pre-event (smaller) particle sizes. This trend will likely continue as smaller size particles enter the stream and embed larger substrates.

Table III.3. High discharge events on the Staunton River between the 1995 post-flood habitat survey and the 1999 habitat survey. Base flows were between 0.03 m^3/s and 0.16 m^3/s over the period.

m^3/s	cfs	occurrences
0.71 - 1.42	26 - 50	12
1.42 - 2.83	51 - 100	4
> 2.83	> 100	2

Observed changes in LWD loadings were controlled by riparian zone changes following the flood and debris flow. The majority of small diameter LWD that was gained by the debris flow affected area in 1995 was washed out of the stream channel during high flow events before 1999. Elimination of trees from the immediate riparian zone of the debris flow affected area decreased input of these size classes of LWD. The source of small diameter LWD remained intact in the riparian zone of the debris flow unaffected area, where loading of small diameter wood increased in 1999. The unaffected area could serve as a source of these size classes for the affected area as high flow events wash small diameter LWD downstream.

In contrast to small diameter LWD, medium diameter wood increased in the debris flow affected area. As stream banks gave way during the debris flow many medium diameter trees fell across, but were suspended above the stream channel. In the four years following the debris flow stream banks continued to erode and many of the suspended trees broke and fell into the stream channel, providing a source of medium diameter LWD. Medium diameter LWD increased at a greater rate in the debris flow unaffected area from 1995-1999, as the riparian zone served as a source of both standing trees and previously downed trees for LWD inputs. The majority of trees standing in the Staunton River riparian zone are medium diameter (pers. obs.). While this will provide a continued source of LWD for the debris flow unaffected area, direct inputs to the affected area are likely to decrease over the next two decades.

Large diameter LWD made up a small proportion of the total LWD load during each survey. Much of the land around the Staunton River was cleared for subsistence farming in the early 1900's before the land was acquired by SNP. Forests must typically be allowed to mature for more than 100 years before large diameter LWD inputs become significant (Murphy and Koski 1989, Dolloff et al.1994). Despite its relative scarcity, large diameter wood showed an increase in 1995, followed by a decrease in 1999. While shorter pieces (size 3) could have conceivably been washed out of the stream, longer pieces (size 6) should have been more persistent (Hilderbrand et al. 1998). The loss of size 6 LWD in 1999 may be in part explained by channel reorganization and the significantly lower discharges during the 1999 survey. Much of the size 6 LWD accumulated in large log jams on the shore, near the stream banks during the debris flow. As the stream channel became more organized between 1995 and 1999 it could have moved far enough away from some of these log jams to prevent their inclusion in the 1999 habitat survey. Low water levels during the 1999 survey would have compounded this effect.

Low input of LWD in the debris flow affected area may seem worrisome based on results of past studies relating low amounts of LWD with decreased habitat diversity, low number of pools, and decreased salmonid density (Fausch and Northcote 1992, Flebbe and Dolloff 1995). While LWD can create a large number of pools through scour or damming of the channel in some streams (Harmon et al. 1986), the 1999 habitat survey indicated that no pools (0 of 105) in the debris flow affected area were formed exclusively by LWD, 5% (5 of 105) were formed by a combination of LWD and rock, and the remainder (95%) were formed by rock alone. Despite higher input of LWD, similar results were found in the debris flow unaffected area where 1% (1 of 190) pools were formed by LWD exclusively, 6% (11 of 190) were formed by a combination of LWD and rock, and the remainder (94%) were formed by rock alone.

As discussed earlier, much of the Staunton River watershed was cleared in the early 1900's before it was acquired by Shenandoah National Park. Although the stream has a high load of LWD in comparison to other streams in the area (USFS CATT, unpublished data), the load consists mostly of small diameter pieces typical of second growth forests that do not have the capability of maintaining their position in the channel and forming pools (Murphy and Koski 1989, Reeves et al. 1995). It is unlikely that LWD has played a significant role in habitat formation in the Staunton River for the last 75 years. Although LWD did not appear to play a significant role in pool formation in the Staunton River, the number of pools/km and fish density (Chapter 2) were similar to those of similarly sized streams flowing through old growth forest where LWD did play a significant role in habitat formation (see Flebbe and Dolloff 1995 for comparison). Flebbe and Dolloff (1995) indicated that in some streams where LWD did not contribute significantly to habitat formation, large cobbles and boulders could provide a suitable surrogate, as appears to be the case in the Staunton River.

In addition to its possible effects on LWD estimates in the present study, changes in discharge have significantly affected estimates of total number of habitat units, average unit surface area, and total stream area in past studies (Herger et al 1996, Hildebrand et al. 1999). In these studies, decreased discharge led to increased number of habitat units, decreased average surface area, and decreased total stream area. In the present study discharge was significantly lower in 1999 than 1995, however the total number of habitat units *decreased*, and average surface area and total stream area *increased*. Any effect decreased flow may have had on estimates was overridden by large changes in channel morphology.

One would suspect that changes in discharge in a disorganized stream with no distinguishable main channel would affect the stream, and thus instream measurements, differently than changes in discharge in an organized stream with one main channel. For example, it may take less change in discharge to cause changes in a disordered channel since water is spread between several small channels. Hildebrand et al. (1998) and Herger et al. (1996) made comparisons within a single stream at different discharge levels and found marked changes in many instream measurements, however they did not speculate on the combined effects of changes in channel morphology and changes in discharge on instream measurements. This combined effect needs to be examined if we are to fully understand results such as those discussed above.

The June 1995 flood and debris flow caused immediate and dramatic changes in the appearance of the Staunton River stream channel and surrounding landscape. Overall, these effects on the stream channel were short-lived with many characteristics at or returning to pre-event levels within four years. Instream habitat differences between the debris flow affected and unaffected areas were minimal and would not likely differentially influence movement and activity (Chapter 1), or density (Chapter 2) of *S. fontinalis* in the Staunton River.

Changes in the riparian zone appeared to have the greatest potential for affecting the fish population after the debris flow. Removal of trees from the riparian zone increased solar radiation to the stream channel, likely increasing production of aquatic invertebrates (Murphy et al. 1986, Keith et al. 1998) and potentially leading to increased growth of fish in the debris flow affected area. Differential growth between affected and unaffected areas was initially observed but was not evident four years post-event. Fish growth appeared to be highly density dependent over that period (Chapter 2). Any differences in growth attributable to increased production in the debris flow affected area will disappear as the stream channel becomes shaded by the riparian tree canopy within the next 10 years (Hornbeck and Kochenderfer 2000).

Synthesis

The long-term impacts of the flood and debris flow on the Staunton River's *S. fontinalis* population appear to be minimal. The debris flow affected area was recolonized and pre-event abundance was reached and exceeded within three years. Stocking was not necessary to facilitate recovery of the population. The only management action that was taken was to close the fishery for two years to allow the population to regain numbers. Four years after the flood and debris flow fish growth, movement, and activity were nearly identical in the affected and unaffected areas of the stream. Long term monitoring will be necessary to determine if the *S. fontinalis* population can maintain its currently high density levels and if fish growth will remain strongly related to population density.

The most obvious and dramatic changes to the Staunton River following the flood and debris flow were to the instream habitat and riparian zone. Instream habitat was initially changed by the event, however, within four years most characteristics were at or returning to pre-event levels. No habitat manipulations were necessary to bring about changes in habitat conditions. The longest lived effect will likely be on LWD inputs in the debris flow affected area. It will be decades before direct inputs of LWD in the affected area return to pre-event levels. Despite low inputs of LWD the trout habitat in the affected area should remain in good condition as boulders create most pools and overhead cover in the Staunton River. Changes in LWD inputs during future habitat surveys will reflect changes in riparian zone characteristics.

Ultimately, humans have the greatest potential for causing long-term and permanent changes to fish populations and habitat following extreme flooding and debris flows. For example, immediately following the debris flow on the Staunton River tree seedlings began to appear in the riparian zone of the debris flow affected area. It was soon noticed that many of the early colonizers, and the fastest growing ones, were the exotic species *Ailanthus* and *Pawlonia*. Furthermore, defoliated areas along the Rapidan River near the Park boundary were rapidly overgrown over by kudzu, yet another introduced species. It will be years before we know the full impact that these exotics will have on SNP and the Staunton River.

Existence of a nearby source population kept the role of humans at a minimum in the recovery of the *S. fontinalis* population of the Staunton River. Had an viable source population not existed in the headwaters of the stream following the debris flow, the Staunton River would have had to depend on the Rapidan River as a source of fish for recolonization. Although population recovery would have likely been slower, it would eventually have occurred. An alternative source population is not a luxury common to streams of SNP. Once most streams leave the protection of the Park habitat quality degrades so quickly and severely that each stream's population is essentially isolated. Should an event similar to the Staunton River debris flow eliminate fish such a stream, transplanting of fish from other streams may become the only option for recovery. Given the genetic variability discovered between SNP streams thus far and the unknown extent of this variability, stocking is not a desirable option. Isolation of *S. fontinalis* populations in headwater reaches of streams decreases their probability of persistence in these highly variable stream environments.

Fish populations in high gradient streams, and in fact the streams themselves are to a large extent shaped by extreme flow events. On a geologic timescale extreme floods and debris flows are common, and although they may initially seem devastating the present study lends itself to the credo; ‘It’s only catastrophic if you’re in the way’. Our desire to immediately return streams to their pre-event condition must be tempered with patience to allow fish populations and habitat to recover naturally.

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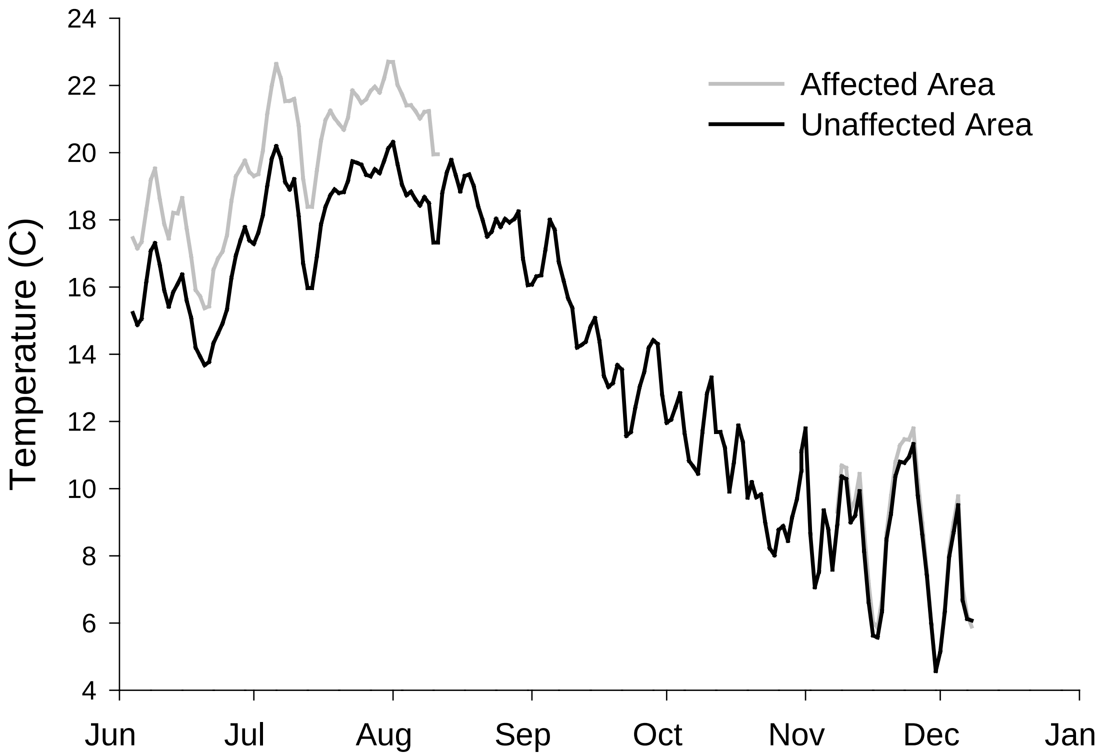
Appendices



Appendix A. Daily average water temperatures in debris flow affected and unaffected sections of the Staunton River June – December, 1999. Note that the affected area logger was not functioning September 21 – October 1.

Appendix B. Population abundance by age and habitat type for *S. fontinalis* in the Staunton River, 1993-1999 as estimated using BVET techniques. Estimates for affected and unaffected sections are not available before the June 1995 debris flow. 95% confidence intervals in parentheses.

		1+		YOY	
Date	Section	Pool	Riffle	Pool	Riffle
Jun-93	Total Stream	3621 (±2281)	2058 (±3935)	455 (±337)	705 (±1250)
Oct-93	Total Stream	962 (±805)	279 (±723)	1146 (±438)	257 (±609)
May-94	Total Stream	317 (±432)	383 (±453)	200 (±192)	138 (±138)
Oct-94	Total Stream	684 (±431)	511 (±564)	727 (±705)	441 (±715)
May-95	Total Stream	2136 (±867)	481 (±716)	321 (±481)	715 (±895)
Oct-95	Affected	0 (±0)	0 (±0)	0 (±0)	0 (±0)
	Unaffected	427 (±457)	420 (±408)	288 (±311)	60 (±59)
	Total Stream	484 (±429)	444 (±378)	307 (±288)	63 (±58)
May-96	Affected	12 (±99)	0 (±0)	153 (±67)	0 (±0)
	Unaffected	1166 (±455)	222 (±564)	2356 (±2995)	573 (±945)
	Total Stream	1082 (±398)	213 (±583)	2428 (±2497)	555 (±982)
Oct-96	Affected	0 (±0)	109 (±246)	0 (±0)	0 (±0)
	Unaffected	904 (±320)	268 (±580)	743 (±692)	251 (±407)
	Total Stream	1136 (±297)	348 (±580)	913 (±634)	223 (±380)

Appendix B. continued.

		1+		YOY	
Date	Section	Pool	Riffle	Pool	Riffle
Jun-97	Affected	544 (±596)	334 (±523)	1537 (±895)	2493 (±1951)
	Unaffected	781 (±332)	240 (±517)	1493 (±1192)	887 (±339)
	Total Stream	1330 (±721)	617 (±713)	3144 (±1262)	3945 (±2253)
Oct-97	Affected	66 (±350)	70 (±136)	219 (±252)	180 (±658)
	Unaffected	1231 (±2381)	345 (±462)	2937 (±1538)	1569 (±2027)
	Total Stream	1178 (±1864)	410 (±420)	3034 (±1598)	1790 (±1931)
Jun-98	Affected	2031 (±1789)	944 (±501)	1571 (±2211)	679 (±750)
	Unaffected	2654 (±3888)	na	0 (±0)	0 (±0)
	Total Stream	4468 (±2677)	1217 (±730)	2919 (±2887)	1246 (±1323)
Oct-98	Affected	466 (±327)	20 (±74)	1485 (±571)	723 (±649)
	Unaffected	3992 (±1608)	0 (±0)	1344 (±1355)	349 (±1911)
	Total Stream	3118 (±1614)	1474 (±1124)	1258 (±1927)	1349 (±579)
Jun-99	Affected	1169 (±541)	0 (±0)	1104 (±885)	1577 (±371)
	Unaffected	2050 (±1726)	901 (±1257)	245 (±977)	734 (±764)
	Total Stream	4472 (±1338)	396 (±981)	3026 (±1150)	1235 (±1642)
Nov-99	Affected	696 (±203)	89 (±86)	347 (±193)	150 (±314)
	Unaffected	1128 (±1159)	122 (±418)	584 (±1082)	167 (±815)
	Total Stream	1865 (±840)	220 (±259)	922 (±622)	425 (±707)

Vita

Craig N. Roghair was born to Barry and Janet Roghair on May 17, 1975. Craig was raised in Richland Center, WI and graduated from Richland Center High School in 1993. Following high school he attended the University of Wisconsin-Stevens Point where he majored in Water Resources and Biology with an emphasis in Fisheries and Limnology. He graduated from UWSP in December 1997 and began his graduate school career at Virginia Polytechnic and State University in January 1998. Craig was brought to Virginia Tech to examine several aspects of recovery of the Staunton River and its brook trout population from a catastrophic flood and debris flow under the guidance of Dr. C. A. Dolloff.