THE EFFECT OF TURBIDITY ON FORAGING BEHAVIOR OF THE DRIFT-FEEDING MINNOW, ROSYSIDE DACE (CLINOSTOMUS FUNDULOIDES).

by

RICHARD M. ZAMOR

(Under the Direction of Gary D. Grossman)

ABSTRACT

Since the turn of the 20th century, streams in the Southeast have experienced increased sediment loads and turbidity. Unfortunately, the effects of turbidity on non-game fishes are not well understood due to a lack of data. Consequently, I examined the effects of turbidity and seasonal temperatures on foraging success in rosyside dace (Clinostomus funduloides). I conducted experiments in an artificial stream and recorded reactive distance and prey capture success. Turbidity had a strong negative effect on both reactive distance (spring/autumn – p<0.0001, r²=0.96; summer – p<0.0001, r²=0.90) and capture success (spring/autumn – p<0.0001, r²=0.88; summer – p<0.0001, r²=0.70). Rosyside dace had longer reactive distances and greater capture success at spring/autumn temperatures than at summer temperatures. Median effective concentrations for reactive distance (spring/autumn – 9.892 NTU, summer – 9.2245 NTU) indicated that disturbed reaches in the natural range of rosyside dace would cause 50% reductions in their reactive distances 50% of the time. These results indicate that turbidity has immediate negative effects on the foraging success of rosyside dace and establishes that these effects may change seasonally.

INDEX WORDS: capture success, Clinostomus, drift-feeding, foraging success, native stream fish, reactive distance, turbidity, median effective concentration
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RICHARD M. ZAMOR

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RICHARD M. ZAMOR

Major Professor: Gary D. Grossman
Committee: Aaron T. Fisk
Mary C. Freeman

Electronic Version Approved:

Maureen Grasso
Dean of the Graduate School
The University of Georgia
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DEDICATION

This study is dedicated to my parents.
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INTRODUCTION AND LITERATURE REVIEW

Since the turn of the 20th century, the Southeastern United States has experienced increased stream sediment loads primarily caused by increased land disturbances from anthropogenic sources (Tebo 1955, Trimble 1974, Jones et al. 1999). For example, agriculture, silviculture, urbanization, mining, and road construction are often responsible for increased sediment loads, (Waters 1995, Allan et al. 1997, Sutherland et al. 2002, Roy et al. 2003) which can negatively affect fish and macroinvertebrate assemblages (Tebo 1955, Murphy and Hall 1981, Culp et al. 1986, Jones et al. 1999, Shaw and Richardson 2001, Richardson and Jowett 2002, Sutherland et al. 2002). Increases in stream sediment loads can cause particular harm to North American fish biodiversity because these streams provide habitat for up to 500 of the 800 freshwater fish species found on this continent (Walsh et al. 1995, Sutherland et al. 2002). Of these 500 species, nearly 300 are darters (percids) or minnows (cyprinids), with most occupying the highland areas of the Appalachians (Walsh et al. 1995, Sutherland et al. 2002). Increases in sediment and subsequent degradation of stream habitats are likely to be at least partially responsible for the threatened or endangered status of nearly 21% of these species (Walsh et al. 1995, Sutherland et al. 2002).

Anthropogenic sediments are present in streams in two correlated forms: as deposits or suspended materials (Waters 1995). Turbidity is the most common method of quantifying suspended sediment in streams, and represents the amount of light scattered or absorbed by a water sample (Duchrow and Everhart 1971). Typically, turbidity is measured in Nephlometric Turbidity Units (NTU’s). Even small quantities of some sediments can markedly affect
turbidity (e.g., 1g kaolin clay/L water = 689 JTU; Duchrow and Everhart 1971) and hence have immediate effects on stream organisms. However, larger quantities over longer time periods are generally necessary to affect a stream’s biota via deposition (Waters 1995). Consequently, as sedimentation occurs, turbidity will increase prior to changes in substratum composition and is a better indicator of potential habitat degradation.

Sediment deposition can adversely affect lotic ecosystems in many ways, such as loss of habitat diversity due to the filling of riffles with sediment or direct smothering of benthic organisms (Berkman and Rabeni 1987, Waters 1995, Jones et al. 1999). It may also reduce larval fish and egg survival due to degraded physical conditions (e.g., decreased dissolved oxygen) or increased predation when refuges between gravel particles are filled (Waters 1995; (Dilts, 1999).

**The effect of turbidity on drift-feeding fishes**

Given that turbidity is an important management concern in Southeastern lotic systems, we need a better understanding of its mode of action. Unfortunately, there are few studies on the effects of turbidity on non-game fishes, which comprise the majority fishes in both the Southeast and the United States as a whole.

Based mostly on studies of game species, it is clear that turbidity may affect fish both physiologically and behaviorally. Negative physiological responses to turbidity include reduced growth rates, reduced disease resistance, impaired vision, gill fouling, reduced clutch size, and increased stress (Sykora et al. 1972, Redding et al. 1987, Sigler et al. 1991, Newcombe and MacDonald 1991, Newcombe and Jansen 1996, Burkhead and Jelks 2001,
Shaw and Richardson 2001). Altered behavioral responses include changes in spawning behavior such as fewer spawns and delayed onset of spawning, resulting in reduced reproductive success (Burkhead and Jelks 2001), microhabitat shifts to avoid increased turbidity, such as moving closer to the surface (Swenson and Matson 1976), and reductions in foraging efficiency. Turbidity primarily reduces foraging efficiency via reductions in reactive distance (i.e. the distance at which a fish first orients to a prey item) (Vinyard and O’Brien 1976, Berg and Northcote 1985, Barret et al. 1992, Gregory and Northcote 1993, Sweka and Hartman 2001a, Sweka and Hartman 2003), capture success (Berg and Northcote 1985, Boehlert and Morgan 1985, Breitburg 1988, Reid et al. 1999, Sweka and Hartman 2001a, Sweka and Hartman 2003), and reduced feeding rates (Gardner 1981, Redding et al. 1987, Gregory and Northcote 1993, Benfield and Minello 1996, Robertis et al. 2003). However, other behavioral effects such as increased searching for prey (Gradall and Swenson 1982, Sweka and Hartman 2001b) and altered prey selection patterns have been observed (Breitburg 1988, Gregory and Northcote 1993, Reid et al. 1999, Rowe et al. 2003).

Studies of the effects of turbidity on fish foraging generally focus on two factors: reactive distance and prey capture success. Fish exhibit a stereotypical orientation behavior upon detection of a potential prey item (Confer and Blades 1975). Once a fish detects a prey item, it aligns its head with the prey item, which improves its visual accuracy (Confer and Blades 1975, Vinyard and O’Brien 1975). After orientation, the fish intercepts the prey. Reactive distance is the distance between the fish’s snout and a given prey item when the fish first orients toward the prey (Confer and Blades 1975 Vinyard and O’Brien 1975). Reactive

A second method of examining the effects of turbidity on drift-feeding fishes is to quantify its effect on prey capture. However, the effects of turbidity on prey capture are difficult to interpret because there is no standard methodology for such studies. For example, results can be reported in terms of feeding rate (prey consumed·min⁻¹·fish⁻¹) (Gregory and Northcote 1993, Benfield and Minello 1996, Rowe et al. 2003), percent fewer prey items consumed (Breitburg 1988), probability of detection (Sweka and Hartman 2001a, 2003), or capture and consumption of individual prey items (Berg and Northcote 1985, Robertis et al. 2003). Hence, it is difficult to derive generalities from these studies. An additional methodological problem for drift-feeding species is that many turbidity studies are conducted in tanks without current (Gregory and Northcote 1993, Rowe et al. 2003) as opposed to flowing water systems which better simulate the natural environment of drift-feeders (Sweka and Hartman 2001a, 2001b, 2003). Finally, many studies have used unrealistic turbidity levels (e.g. Gregory and Northcote 1993 levels up to 810 NTU, Rowe et al. 2003 levels up to 320 NTU) or sediments not commonly found in aquatic habitats (e.g. bentonite and kaolin clay, Robertis et al. 2003). These methodological problems render extrapolation to the natural environment problematical (Duchrow and Everhart 1975; Waters 1995).

Reductions in prey capture success, feeding rates, or probability of detection in drift-feeding species at high turbidities has been shown for coho salmon (30 and 60 NTU) (Berg and Northcote 1985), chinook salmon (370 – 810 NTU) (Gregory and Northcote 1993), and brook trout (25 – 40 NTU) (Sweka and Hartman 2001a). However, Gregory and Northcote (1993)
also observed increased or unchanged feeding rates at intermediate turbidities (35 – 150 NTU). Gregory and Northcote (1993) suggested that these behaviors were due to perceptions of reduced predation risk at intermediate turbidities by juvenile Chinook salmon. Thus, foraging motivation may change with increases in turbidity. However, this may also be due to increased visibility of prey under turbid conditions (Robertis et al. 2003) resulting in increased visual acuity and increased prey consumption. These may also be the result of a “tank effect” allowing fish to encounter prey in a different way than seen in a flowing system. Therefore, the general trend suggests decreased foraging success of fishes with increased turbidity; however, more studies need to be completed for non-salmonid drift-feeding fishes.

Suspended sediment also may negatively affect drift-feeding salmonids by: 1) reducing selection for larger prey (Rowe et al. 2003), 2) restricting searching behavior to non-turbid portions of the water column or substratum (Gregory and Northcote 1993, Rowe et al. 2003), and 3) disrupting feeding hierarchies (Berg and Northcote 1985). A possible explanation for these behavioral changes is reductions in the foraging capability of the fish (Berg and Northcote 1985, Gregory and Northcote 1993, Rowe et al. 2003). The presence of these responses in non-game fishes is unknown due to a paucity of research.

Temperature is another factor that may influence reactive distance and capture success. To my knowledge the effects of temperature on reactive distance in fishes is unknown. However, increased temperatures may: 1) increase prey consumption and/or attack rate (Persson 1986, Bergman 1987, Johnson and Mathias 1994, Taniguchi et al. 1998), 2) decrease handling times (Persson 1986, Bergman 1987, Taniguchi et al. 1998), 3) increase capture success (Persson 1986, Magoullick and Wilzbach 1998, Taniguchi et al. 1998), 4) decrease or have no
effect on capture success (Schmidt and O’Brien 1982, Persson 1986, Bergman 1987, Johnston and Mathias 1994, Taniguchi et al. 1998). Temperature increases produce increased capture success in roach (12, 15, 18, and 21ºC, Persson 1986), brook and rainbow trout (13 and 18ºC, Magoullick and Wilzbach 1998) and creek chub (range of treatments from 3 – 26ºC, Taniguchi et al. 1998). However, increased temperature did not affect capture success of perch (12, 15, 18, and 21ºC, Persson 1986; range of treatments from 4 – 20ºC, Bergman 1987), ruffe (4 – 20ºC, Bergman 1987), or walleye (15, 18.5, and 22ºC, Johnston and Mathias 1994) and produced decreased capture success for arctic grayling (5, 10, and 15ºC, Schmidt and O’Brien 1982) and brook and brown trout (3 – 26ºC, Taniguchi et al. 1998). The differences seen in capture success for grayling appeared to be due to reduced escape capabilities of prey at low temperatures (Schmidt and O’Brien 1982) and the higher temperatures in Tanaguchi et al.’s (1998) study were at or above the upper thermal limit for trout survival and likely reduced their ability to feed.

**Study Species: Rosyside Dace**

The rosyside dace is a species of special concern in North Carolina (NCWRC 2004, Grossman et al. 2002). Rosyside dace are members of the mid-water column habitat guild and occupy microhabitats over a range of substrate types, in deeper areas of the stream, with relatively high focal point velocities, but far from both shelter and the substratum. They are native to streams of the Little Tennessee River drainage located in the highlands of western North Carolina (Grossman and Freeman 1987, Grossman and Ratajczak 1998, Grossman et al. 1998) and typically breed between May and August with gonadal recrudescence being positively correlated with photoperiod (DeHaven et al. 1992). Rosyside dace feed opportunistically on the
drift, primarily consuming aquatic macroinvertebrates and terrestrial insects (Stouder 1990) occupying microhabitats at focal point velocities that maximize net energy gain (Hill and Grossman 1993, Grossman et al. 2002). This is important because if turbidity negatively affects foraging success then it may also negatively affect microhabitat availability and microhabitat choice. Because rosyside dace are drift-feeders, they are likely to be negatively affected by increases in turbidity. Previous examinations of capture success of rosyside dace at summer and winter temperatures revealed that capture success was largely a function of velocity and distance between a fish and its prey rather than temperature, although at summer temperatures fish had slightly higher capture success rates at longer distances and higher velocities (Hill and Grossman 1993). Therefore, investigations of how turbidity affects foraging success at different seasonal temperatures are also important to understanding how turbidity could affect microhabitat selection.

My primary objective in the following study was to determine the effects of turbidity on reactive distance and capture success in rosyside dace at both spring/autumn and summer temperatures. Specifically, I tested the following null hypotheses: (1) reactive distance and prey capture success are not affected by increased turbidity levels, and (2) relationships between turbidity, prey capture, and reactive distance are not affected by seasonal temperatures.

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CHAPTER 1

THE EFFECT OF TURBIDITY ON FORAGING BEHAVIOR OF THE DRIFT-FEEDING MINNOW, ROSYSIDE DACE.

ABSTRACT

North American streams are experiencing increased suspended sediment loads, but unfortunately, the effects of suspended sediment on non-game fishes are not well understood. We examined the effects of suspended sediment (i.e., turbidity) on reactive distance and prey capture success at spring/autumn (12°C) and summer (18°C) temperatures for rosyside dace (Clinostomus funduloides) in an artificial stream. Experimental turbidities ranged from 0-56 NTU’s at each of the seasonal temperatures and sample sizes for each turbidity ranged from 5-10 fish. Each fish was tested at one randomly selected turbidity. Non-linear regression detected a strong negative relationship between turbidity and reactive distance at both spring/autumn (p<0.0001, r²=0.96) and summer (p<0.0001, r²=0.90) temperatures. Turbidity also had a strong negative effect on capture success at both spring/autumn (p<0.0001, r²=0.88) and summer (p<0.0001, r²=0.70) temperatures. Two-way ANOVA showed significant turbidity and temperature effects in all cases (p<0.0001). Median effective concentrations for reactive distance (spring/autumn – 9.9 NTU’s, summer – 9.2 NTU’s) indicated that some monitored streams within the natural range of rosyside dace have turbidities levels that would produce a 50% reduction in the maximum reactive distance of rosyside dace 50% of the time. These results suggest that turbidity negatively affects foraging behavior of rosyside dace at even low to intermediate levels (>9 NTU) and that these effects may vary seasonally.

INTRODUCTION

Increased suspended sediment loads in North American streams pose an important management concern for fisheries biologists (Barrett et al. 1992, Sutherland et al. 2002, Walsh et
al. 1995, Waters 1995). In general, most of these increases can be attributed to increased land disturbances from agriculture, urbanization, and vacation home construction (Tebo 1955, Waters 1995, Sutherland et al. 2002, Roy et al. 2003). Increased sediment loads in streams negatively affect both fish and macroinvertebrate assemblages (Tebo 1955, Murphy and Hall 1981, Culp et al. 1986, Jones et al. 1999, Shaw and Richardson 2001, Richardson and Jowett 2002, Sutherland et al. 2002), and may be linked to the threatened/endangered status of approximately 21 percent of the almost 300 darter (Percidae) or minnow (Cyprinidae), species in the Southeastern United States (Walsh et al. 1995; Sutherland et al. 2002).

Turbidity, the amount of light scattered or absorbed by a sample of water, is the most common metric for quantifying suspended sediment in streams (Duchrow and Everhart 1971). Turbidity measurements may offer a more sensitive index of sediment effects on stream fishes, because even small changes in suspended sediment may change turbidity levels, whereas larger quantities generally are necessary to produce changes in the substratum (Waters 1995). The effects of turbidity, however, on non-game fishes (i.e. the majority of fishes in North America), are not well understood. Nonetheless, studies of game/commercial species and one cyprinid show that turbidity negatively affects foraging success (salmonids - Berg and Northcote 1985, Redding et al. 1987, Barret et al. 1992, Gregory and Northcote 1993, Sweka and Hartman 2001a), reproductive success, (cyprinid - Burkhead and Jelks 2001), and microhabitat selection (coregonid - Swenson and Matson 1976).

Most researchers have quantified the effects of turbidity on fish foraging by measuring reactive distances and prey capture success (Berg and Northcote 1985, Barret et al. 1992, Sweka and Hartman 2001a), although there are few extant studies that deal with non-game
fishes. Reactive distance is the distance between the fish’s snout and a given prey item when
the fish first orients toward the prey (Confer and Blades 1975; Vinyard and O’Brien 1975).
Drift-feeding fish exhibit stereotypical orientation behavior when prey is sighted, with the fish
immediately aligning its head with the prey item (Confer and Blades 1975; Vinyard and
O’Brien 1975). This behavior improves visual accuracy and capture success (Confer and
Blades 1975 Vinyard and O’Brien 1975). After orientation, fish typically move to intercept the
prey (Confer and Blades 1975).

The relationship between increased turbidity and prey capture success also is not well
established, because investigators have used varying research methodologies. For example,
capture success has been measured in different ways (prey consumption rate - Gregory and
Northcote 1993, Rowe et al. 2003; percent fewer prey consumed - Breitburg 1988; probability
of detection - Sweka and Hartman 2001a, 2003; capture and consumption of individual prey -
Berg and Northcote 1985; Robertis et al. 2003), non-flowing experimental systems have been
used for drift feeders (Gregory and Northcote 1993, Rowe et al. 2003), non-naturally occurring
substances (bentonite, kaolin) have been used as turbidity agents (Robertis et al. 2003) and
some turbidity treatments may be extremely high (810 NTU’s - Gregory and Northcote 1993;
320 NTU’s - Rowe et al. 2003). Nevertheless, high turbidities decrease either prey capture
success or feeding rate in the following drift-feeding species: coho salmon (Berg and
Northcote 1985, Redding et al. 1987), chinook salmon (Gregory and Northcote 1993),
steelhead (Redding et al. 1987), and brook trout (Sweka and Hartman 2001a).

Given the lack of information on the effects of turbidity on foraging behavior of non-
game fishes, we assessed the effects of turbidity on reactive distance and prey capture success
of rosyside dace (*Clinostomus funduloides*), a drift-feeding minnow of special concern in North Carolina (NCWRC 2004). We also determined whether temperature and fish size affected reactive distance and capture success at different turbidities. To our knowledge, there is little or no information on how seasonal temperature change or fish size affect turbidity-foraging behavior relationships.

**METHODS**

**Study Species**

The rosyside dace is a native species of special concern in North Carolina (NCWRC 2004, Grossman et al. 2002). Rosyside dace are members of the mid-water column habitat guild (sensu Grossman et al. 1998) and occupy deeper microhabitats with higher focal point velocities and varying substrata far from both shelter and the substratum. In the southern Appalachian region, rosyside dace typically breed between May and August and gonadal recrudescence is positively correlated with photoperiod (DeHaven et al. 1992). Rosyside dace are opportunistic drift-feeders, primarily consuming aquatic macroinvertebrates and terrestrial insects (Stouder 1990). In Coweeta Creek, NC, rosyside dace occupy focal point velocities (i.e. microhabitats) that maximize their rate of net energy gain, which presumably increases individual fitness (Hill and Grossman 1993, Grossman et al. 2002). Given their biology and previous research on distribution and abundance (Jones et al. 1999, Scott and Helfman 2001), it is likely that rosyside dace will be affected by increased turbidities.
**Experimental Stream**

We conducted experiments in a 3.0 x 0.75 x 1.0m plexiglas artificial flume (Figure 1). Flows within the flume were generated using electric trolling motors, and were recirculated via a false bottom (Fig 1). A collimator composed of baffles covered with net mesh maintained semi-laminar flow within the flume, and bounded the upstream and downstream ends of the “test” chamber (Fig 1). The dimensions of the test chamber of the flume were 2.2 x 0.75 x 0.5m (Fig 1). We used a four cm layer of pebbles (~1-50mm) as substratum. The flume was marked with a grid on the back wall and a scale on front wall even with the false bottom facing the observer. We surrounded the flume with a black plastic blind to prevent disturbance of fish during observation. Experimental conditions simulated habitats where fish are found in the natural environment, i.e. depth (45cm), pH (~7), and flow (12 cm/s) (Grossman and Freeman 1987, Grossman and Ratajczak 1998, Grossman et al. 1998). We have previously shown that rosaside dace preferentially occupy this velocity to maximize their rate of net energy gain in Coweeta Creek (Grossman et al. 2002). We conducted experiments at two seasonal temperatures: spring/autumn – 12ºC and summer – 18ºC (Grossman et al. unpublished data). Seasonal photoperiods also were used (i.e., 12L:12D for spring/autumn, 14L:10D for summer).

**Experimental Procedures**

We collected adult rosaside dace for experiments via electrofishing and seining from Coweeta Creek, a fifth-order tributary stream of the Little Tennessee River. Fish collected for summer temperature trials were collected from July 2004 to February 2005, whereas fish for spring/autumn temperature trials were collected from February 2005 to May 2005. Fish used in
experiments showed no visible physical or behavioral alterations from collection, and this represented the vast majority of fish collected (i.e., ~95%). As a prophylactic, we treated fish with a general antibiotic, Kanamycin upon introduction to a holding tank that simulated conditions in the experimental flume. We acclimated fish for at least four days prior to experimental trials. During both the holding period and experiments, we fed fish miniature mealworms (*Tenebrio molitor*, length - 8-10mm, mass ~0.02g). We used mealworm prey for a variety of reasons including: 1) their resemblance to natural prey and ready acceptance by rosyside dace, 2) ease of visibility during experiments, and 3) previous use in other studies (Barret et al. 1992). After acclimation to the holding system, we removed test fish to a similar holding tank for a 48h period of starvation to ensure that all test individuals were in similar condition and would feed readily during trials.

We used commercially purchased red clay to vary turbidity levels within the flume because it is a naturally occurring suspended sediment in the Southeastern United States (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. personal communication). A paint mixer powered by an electric drill, mixed water from the artificial flume with red clay in a five-gallon bucket to create a clay/water suspension. We allowed this mixture to settle for 15 minutes before adding the suspension to the flume. We measured turbidity in trials with a Hach Model 2100P turbidity meter and took three measurements immediately before and after each trial to ensure that the test turbidity was maintained throughout the trial. We also took a 1L sample (four 250ml subsamples) from different positions of the tank during one trial) at each turbidity level to derive a relationship between turbidity and total suspended solids (TSS) for both spring/autumn and summer temperatures using non-linear regression.
For trials, we first placed an arbitrarily selected fish in the test chamber and randomly selected a turbidity level. For summer temperatures, we measured reactive distances and prey capture success for 85 individual fish (5 fish at 0, 10 and 15 NTU’s and 10 fish at higher levels) at 0, 5, 10, 15, 20, 25, 30, 35, 45 and 55 NTU’s. We chose these turbidity levels because they may occur in streams within the distribution of this species (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data). For spring/autumn trials, we measured reactive distance and capture success for 80 fish (10 at each turbidity level) at 0, 8, 16, 24, 32, 40, 48 and 56 NTU’s. Spring/autumn treatment levels were sufficient to depict trends present in summer data with fewer treatments. Each fish was allowed to acclimate for a period of at least 2 hours to the experimental chamber after test turbidities had been attained and was tested at only one turbidity level.

We began a trial by releasing prey at approximately two-minute intervals from a constant submerged position at the front of the experimental chamber. Forceps were used to release prey, which traveled the length of the flume as submerged drift. We began measuring reactive distance and prey capture success after a test fish captured its first prey, because fish did not always react to the first few prey even at low turbidities. At low turbidities (0 – 16 NTU’s), if a fish did not feed on the first five prey presented, it was not used in a trial. At high turbidities (20 – 56 NTU’s) if fish did not react, we began to record observations after five prey items had been placed in the tank. We used this criterion for higher turbidities because we could not distinguish “non-feeders” from fish that would feed but could not see prey. This could have potentially biased trials, however, only one fish was removed at low turbidities in spring/autumn trials.
whereas four were removed summer trials (2 at 0 NTU and 2 at 5 NTU). Consequently it is unlikely that the different criteria used for “non-feeders” significantly biased our results.

We measured reactive distance and prey capture using videography. Reactive distance was measured as the distance from the prey to the tip of the fish’s snout upon first orientation. We accomplished this during video playback by marking the position of the both the prey (released from a known position), and the fish’s snout after it oriented to the prey. With these positions marked, we used the grid or scale to determine both the vertical (above the fish) and horizontal (in front of the fish) distance of the prey from the fish’s snout. We then constructed a right triangle with these data, which allowed us to calculate reactive distance as the hypotenuse of a right triangle (Fig 1). Our estimates may have been slightly biased because we reduced a three dimensional process to two dimensions, however, our observations indicated that lateral movement of prey was not extensive during a trial. We calculated prey capture success as the percentage of prey captured in a given trial. Our null hypotheses were that turbidity had no effect on either reactive distance or prey capture success of rosside dace.

We quantified the relationships between turbidity, reactive distance and prey capture success data using a non-linear, four parameter, dose-response regression with a variable slope parameter: \( y = \min + \frac{(\max - \min)}{(1 + 10^{(\log EC50 - x)Hillslope})} \) (Sigmaplot v. 8.0, 2002). The parameters were: 1) minimum (bottom of the curve), 2) maximum (top of the curve), 3) median effective concentration (EC50) (i.e., concentration required to elicit a response equal to 50% of the maximum response observed), and 4) hillslope (i.e., slope of the curve at its midpoint). We also analyzed the data using two-way ANOVA with season and turbidity as main effects (SAS 1999). We considered seasonal turbidities that were within 3 NTU’s of each other to be identical.
treatments (see previous section). Therefore, only comparable turbidities from the two seasonal trials were used in the ANOVA with the remaining 4 treatments (summer – 5, 20, and 35 NTU’s and spring/autumn – 40 NTU’s) being excluded from analyses. We also used an analysis of covariance (ANCOVA) to determine if size had an effect on reactive distance and capture success with turbidity (SAS 1999).

RESULTS

Total Suspended Solids

Turbidity and TSS were strongly correlated during both spring/autumn (p<0.0001, $r^2 = 0.99$) and summer (p<0.0001, $r^2 = 0.98$) trials, and for both temperatures combined (p<0.0001, $r^2 = 0.95$, Figure 2). TSS values were slightly higher at a given NTU for spring/autumn temperature than for the same summer temperature. Unfortunately, due to a lack of replication it was impossible to test for temperature effects in the TSS relationship.

Reactive Distance

The regression produced a strong negative curvilinear relationship between increasing turbidity and reactive distance for rosyside dace at both spring/autumn and summer seasonal temperatures (both, p’s<0.0001; spring/autumn $r^2 = 0.96$; summer $r^2 = 0.90$; Figure 3). Both curves displayed an exponentially declining phase, followed by a phase with a reduced negative slope at turbidities above 20 NTU’s (Fig. 3). Median effective concentration (EC50’s) values were approximately 9 NTU’s for both seasons (spring/autumn – 9.892, summer – 9.2245) indicating that a 50% loss in the maximum observed reactive distance occurred at relatively low
turbidities. Two-way ANOVA detected highly significant turbidity (2-way ANOVA: $F = 357.46$, $df = 6$, $p < 0.0001$) and seasonal temperature effects ($F = 137.79$, $df = 1$, $6$, $p < 0.0001$, Table 1) with a non-significant interaction term ($F = 0.00$, $df = 6$, $p = 1.0$) Rosyside dace had significantly greater reactive distances at spring/autumn temperatures than they did at summer temperatures, regardless of turbidity level (Fig. 3).

Analyses of reactive distance data within seasons showed that differences in mean reactive distances were most pronounced at low to intermediate turbidities for both seasons (Tukey tests, Table 1). *Aposteriori* t-tests between seasonal treatments also detected significant differences in reactive distance between seasons at intermediate to high turbidities (8 – 32 NTU’s) (Table 1). Despite the negative effects of turbidity on reactive distance, some rosyside dace reacted to prey at even the highest turbidities. Finally, fish size did not significantly affect reactive distances for either seasonal temperature (spring/autumn temperature ANCOVA length: $F = 2.20$, $df = 1$, $p = 0.1432$; mass: $F = 1.89$, $df = 1$, $p = 0.1741$; summer temperature ANCOVA length: $F = 0.02$, $df = 1$, $p = 0.9$; mass: $F = 0.1$, $df = 1$, $p = 0.7516$).

**Prey Capture Success**

We also obtained a strong and highly significant (both $p$’s $< 0.0001$) curvilinear negative relationship between turbidity and capture success for both spring/autumn ($r^2 = 0.88$) and summer ($r^2 = 0.70$) temperature trials (Figure 4). In both seasons, the relationship had an asymptote at low turbidities followed by an exponential decline and then a lower plateau at high turbidities. Median effective concentrations (EC50’s) for capture success at both temperatures indicated that capture success was reduced by 50% of the maximum at approximately 30 NTU’s.
Two-way ANOVA also showed that turbidity had a significant negative effect on capture success \((F = 106.48, df = 1, 6, p < 0.0001)\), as did temperature \((F = 11.04, df = 1, 6, p = 0.0012, \text{Table 2})\) whereas the interaction term was non-significant \((F = 0.00, df = 6, p = 1.0)\). Rosyside dace caught more prey at spring/autumn temperatures than they did at summer temperatures, regardless of turbidity level (Fig. 4).

Tukey *aposteriori* tests of within-season capture success data indicated that significant differences between turbidities were most common at intermediate to high turbidities (>15 – 16 NTU’s) for both seasonal temperatures (Table 2). *Aposteriori* t-tests between seasonal treatments also indicated that differences in capture success between spring/autumn and summer temperature trials were strongest at intermediate (16 – 15 and 24 – 25 NTU’s) turbidities (Table 2). Finally, ANCOVA indicated that neither specimen length nor mass significantly affected capture success of rosyside dace in either temperature treatment (spring/autumn temperature treatments: ANCOVA length: \(F = 0.74, df = 1, p = 0.3927\); mass: \(F = 1.53, df = 1, p = 0.2213\), summer temperature treatments: ANCOVA length: \(F = 0.01, df = 1, p = 0.9340\); mass: \(F = 0.20, df = 1, p = 0.6603\)).

**DISCUSSION**

Our results demonstrate that increased turbidity significantly reduced reactive distance and capture success at both spring/autumn and summer temperatures in rosyside dace. We observed significant reductions in foraging success at turbidities as low as 8 NTU’s; a turbidity that is frequently observed in streams in the Coweeta Creek region (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data). It is curious that rosyside dace had greater
capture success and longer reactive distances in spring/autumn trials even though TSS values were higher for all turbidities at this temperature. However, we did not replicate samples for TSS analyses; hence, it is unclear if the seasonal temperature differences in the TSS-NTU relationship are real.

The range of turbidities used in this study encompassed the range found in the streams from which these fish were collected (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data). Minimally disturbed streams, (forested with no disturbance), within the region have turbidities that are nearly always lower than 40 NTU’s (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data). Disturbed streams have turbidity levels greater than the EC50’s for reactive distance (>9 NTU’s) for both seasons over 50 percent of the time and have turbidity levels that exceed our highest levels slightly more than 10 percent of the time (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data). In addition, rosyside dace in disturbed streams would also experience turbidities above the EC50 for capture success (>30NTU’s) at least 10 percent of the time (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data). By contrast, rosyside dace in minimally disturbed streams would have foraging rates that were not significantly different from 0 NTU’s over 90% of the time, although their reactive distance and capture success would be affected at approximately one percent of the time (M. Riedel, Coweeta Hydrologic Laboratory, Otto, N.C. unpublished data).

Behavioral shifts may allow fish to forage effectively even with reduced reactive distances. For example, fish may increase searching behavior as turbidity increases (Sweka and Hartman 2001b). Observations of rosyside dace at high turbidities revealed that some fish would increase their search efforts by searching faster and more widely within the flume rather than
holding a static position (personal observation). This would increase their search volumes. Once a rosyside dace detected a prey item, even in the highest turbidities, it nearly always captured that prey item (personal observation). Sweka and Hartman (2001a) observed similar phenomena with brook trout, which also displayed reduced reactive distances with increased turbidity, but no change in either prey capture or consumption rates once prey was detected. Nonetheless, increased searching behavior resulted in higher net energy expenditure per prey and certainly would have a negative effect on individual fitness. Because we used optimal velocities for rosyside dace in our experiments, and because these velocities typically are occupied by rosyside dace in the field (Grossman et al. 2002), the reduced foraging efficiency observed at intermediate and high turbidities probably will occur in field populations subjected to similar turbidities.

The shape of reactive distance and capture success curves was curvilinear and followed a dosage dependent relationship. A similar curvilinear response has been seen for reactive distance in other species (Vinyard and O’Brien 1976, Gregory and Northcote 1993, Benfield and Minello 1996, Sweka and Hartman 2001a, Sweka and Hartman 2003) which suggests that this may be a general relationship. If true, the use of dosage-dependent curve may be advantageous statistically for future turbidity studies.

Although we expected prey capture success to be highest at 0 NTU’s, there was little difference between the prey capture success at the first two turbidities in both seasonal trials. At 0 NTU’s rosyside dace appeared more unsettled than at 5 or 8 NTU’s. Perhaps rosyside dace perceived a more secure feeding environment (i.e., greater visual isolation) in slightly turbid water. Even though the majority of predators in these streams are avian (Grossman et al. 1998), fish may have perceived an increased vulnerability to predation in clear water due to the lack of
physical structure. These suggestions are similar to those of Gregory and Northcote’s (1993) for chinook salmon, however their results are confounded by a prey-visibility effect (Robertis et al. 2003) and by their study being conducted in static tanks.

Studies of the effects of turbidity on foraging behavior of fishes have been conducted in both flowing water and static systems. We have compared our results primarily to those from other studies that used flowing water systems because static tanks produce many artifacts when used with drift-feeding stream fishes. It is likely that some of the differences noted among studies (e.g., reduced foraging success only occurring at extremely high turbidities - 810 NTU’s Gregory and Northcote, 320 NTU’s Rowe et al. 2003) can be attributed to use of static versus flowing water systems.

To our knowledge, there are no extant studies on the effects of temperature on reactive distance in fishes. However, the studies of the effects of temperature on capture success show that increased temperatures may: 1) increase prey consumption and/or attack rate (Persson 1986, Bergman 1987, Johnson and Mathias 1994, Taniguchi et al. 1998), 2) decrease handling times (Persson 1986, Bergman 1987, Taniguchi et al. 1998), 3) increase capture success (Persson 1986, Magoulick and Wilzbach 1998, Taniguchi et al. 1998), 4) decrease or have no effect on capture success (Schmidt and O’Brien 1982, Persson 1986, Bergman 1987, Johnston and Mathias 1994, Taniguchi et al. 1998) prey capture success. Increased temperatures produce increases in capture success in roach (12, 15, 18, and 21°C, Persson 1986), brook and rainbow trout (13 and 18°C, Magoulick and Wilzbach 1998) and creek chub (range of treatments from 3 – 26°C, Taniguchi et al. 1998). Nonetheless, increased temperature did not affect capture success of perch (12, 15, 18, and 21°C, Persson 1986; range of treatments from 4 – 20°C, Bergman 1987), ruffe (4 – 20°C,
Bergman 1987), or walleye (15, 18.5, and 22°C, Johnston and Mathias 1994) and was higher at colder temperatures for stream-dwelling arctic grayling (5, 10 and 15°C, Schmidt and O’Brien 1982) and brook and brown trout (3 – 26°C, Taniguchi et al. 1998). The differences in capture success for grayling likely are due to slower prey escape responses at low temperatures (Schmidt and O’Brien 1982), whereas, the lack of differences for brook and brown trout in Taniguchi et al. (1998) study probably are attributable to temperatures near the upper thermal limit for trout survival. By contrast, Hill and Grossman (1993) showed that capture success of rosidside dace and rainbow from Coweeta Creek was much more strongly affected by velocity and distance of the fish from the prey than by temperature. In our study, rosidside dace had consistently higher capture success at spring/autumn temperatures but the differences were only significant at intermediate turbidities. Consequently, the relationships between turbidity, temperature and prey capture success are complex and their interpretation problematical.

Our results may provide a mechanism for the observation that rosidside dace and other highland endemic species are found at reduced densities, or are absent, from streams with high turbidities and substrata dominated by fines in western North Carolina (Jones et al. 1999; Scott and Helfman 2001). Fine sediments may affect streams in many ways (e.g., smothering invertebrates, reducing habitat complexity, deoxygenating sediments), hence fishes occupying these habitats face habitat degradation in a variety of forms, besides reduced foraging efficiency (Berkman and Rabeni 1987). Nonetheless, our results suggest that even intermediate turbidities that persist in western North Carolina streams will result in reduced foraging success and ultimately, declines in abundance and the distribution of this species.
In conclusion, our results demonstrate that turbidity significantly reduced reactive distance and capture success of rosystide dace at both spring/autumn and summer temperatures. Even relatively small amounts of red clay produced turbidities that reached the EC50 values for dace at both seasonal temperatures, and probably would result in reduced individual fitness for these fish in the field. At present, similar turbidities are not uncommon in southeastern streams and suggest that increasing land disturbance within the range of rosyside dace could negatively affect its abundance and distribution, especially if they are persistent rather than ephemeral. Our data should be useful in the determination of total maximum daily loads (TMDL’s) in streams inhabited by this species. It is likely that experiments that examine the effects of turbidity on fitness-related parameters such as foraging success will be useful in establishing environmental standards and interpreting correlative relationships between suspended sediments and fish abundance/biodiversity.

**Literature Cited**


Sigmaplot. 2002. Version 8.0. Chicago, IL, USA


Taniguchi, Y., F. J. Rahel, D. C. Novinger, and K. G. Gerow. 1998. Temperature mediation of competitive interactions among three fish species that replace each other along


Waters, T. F. 1995. Sediment in streams: source, biological effects, and control, American Fisheries Society, Bethesda, MD, USA.
Tables and Figures

Table 1. Mean reactive distances for turbidity treatments in seasonal trials. Reactive distance data are also presented as the percent of the maximum observed value. Significant differences in reactive distance between turbidities within seasonal treatments indicated by *aposteriori* Tukey tests are shown with different superscripts. Results from *aposteriori* t-tests between seasonal treatments are also shown in the last column.

<table>
<thead>
<tr>
<th>Turbidity Treatment (NTU)</th>
<th>Reactive distance (cm)</th>
<th>Reactive distance (% Max)</th>
<th>T-test</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>SPR/AUT</td>
<td>SUM</td>
<td>SPR/AUT SUM</td>
</tr>
<tr>
<td>0 – 0</td>
<td>49.4 ± 2.9(^a)</td>
<td>46.5 ± 2.9(^a)</td>
<td>100.00 100.00</td>
</tr>
<tr>
<td>8 – 10</td>
<td>30.1 ± 1.4(^b)</td>
<td>27.9 ± 2.5(^b)</td>
<td>60.1 60.0</td>
</tr>
<tr>
<td>16 – 15</td>
<td>23.5 ± 1.1(^c)</td>
<td>19.0 ± 3.4(^c)</td>
<td>47.6 40.9</td>
</tr>
<tr>
<td>24 – 25</td>
<td>21.3 ± 2.2(^c,d)</td>
<td>14.8 ± 3.1(^c,d)</td>
<td>43.1 31.8</td>
</tr>
<tr>
<td>32 – 30</td>
<td>20.7 ± 2.2(^d)</td>
<td>13.8 ± 2.5(^d)</td>
<td>41.9 29.7</td>
</tr>
<tr>
<td>48 – 45</td>
<td>13.9 ± 2.7(^e)</td>
<td>11.8 ± 1.6(^d)</td>
<td>28.1 25.4</td>
</tr>
<tr>
<td>56 – 55</td>
<td>12.6 ± 3.0(^e)</td>
<td>9.9 ± 2.9(^d)</td>
<td>25.5 21.3</td>
</tr>
</tbody>
</table>
Table 2. Mean capture success for turbidity treatments in seasonal trials. Capture success data are also presented as the percent of the maximum observed value. Significant differences in capture success between turbidities within seasonal treatments indicated by *aposteriori* Tukey tests are shown with different superscripts. Results from *aposteriori* t-tests between seasonal treatments are also shown in the last column.

<table>
<thead>
<tr>
<th>Turbidity Treatment (NTU)</th>
<th>Successful captures (# caught out of 10 possible)</th>
<th>Prey capture success (% Max)</th>
<th>T-test</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td><strong>SPR/AUT</strong></td>
<td><strong>SUM</strong></td>
<td><strong>SPR/AUT</strong></td>
</tr>
<tr>
<td>0 – 0</td>
<td>9.3 ± 0.8&lt;sup&gt;a&lt;/sup&gt;</td>
<td>9.4 ± 0.9&lt;sup&gt;a&lt;/sup&gt;</td>
<td>100.0</td>
</tr>
<tr>
<td>8 – 10</td>
<td>9.3 ± 0.8&lt;sup&gt;a&lt;/sup&gt;</td>
<td>8.8 ± 1.1&lt;sup&gt;a,b&lt;/sup&gt;</td>
<td>100.0</td>
</tr>
<tr>
<td>16 – 15</td>
<td>9.2 ± 0.9&lt;sup&gt;a,b&lt;/sup&gt;</td>
<td>8.1 ± 1.1&lt;sup&gt;a,b&lt;/sup&gt;</td>
<td>98.9</td>
</tr>
<tr>
<td>24 – 25</td>
<td>7.8 ± 1.0&lt;sup&gt;b&lt;/sup&gt;</td>
<td>6.1 ± 1.6&lt;sup&gt;b,c&lt;/sup&gt;</td>
<td>83.9</td>
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<td>32 – 30</td>
<td>5.5 ± 1.2&lt;sup&gt;c&lt;/sup&gt;</td>
<td>4.0 ± 2.9&lt;sup&gt;c&lt;/sup&gt;</td>
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<td>48 – 45</td>
<td>1.7 ± 1.2&lt;sup&gt;d&lt;/sup&gt;</td>
<td>1.5 ± 2.3&lt;sup&gt;c,d&lt;/sup&gt;</td>
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<td>56 – 55</td>
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<td>1.1 ± 1.4&lt;sup&gt;d&lt;/sup&gt;</td>
<td>18.3</td>
</tr>
</tbody>
</table>
Figure 1. Diagram of the artificial stream used in experiments. Prey release and measurement of reactive distance are also illustrated (Figure by Peter Hazelton).
Figure 2. Non-linear regression of Total Suspended Solids (TSS, mg/L) versus turbidity (NTU) within the flume used for turbidity experiments. Spring/Autumn temperature samples (●) are represented by a solid line ($y = -0.0052x^2 + 0.7832x - 0.9229, r^2 = 0.985$). Summer temperature samples (○) are represented by a dashed line ($y = -0.0031x^2 + 0.6158x - 1.4658, r^2 = 0.973$). A combination of spring/autumn and summer temperature trials are represented by a dotted line ($y = -0.0037x^2 + 0.6797x - 1.2819, r^2 = 0.947$).
Figure 3. Non-linear regression of reactive distance (cm) versus turbidity (NTU).

Spring/autumn temperature trials (●) are represented by a solid line

\[ y = \frac{57.7527}{1 + \left(\frac{x}{9.892}\right)^{0.6772}} \]

EC50 = 9.892, \( r^2 = 0.96 \) and summer temperature trials (○) are represented by a dashed line

\[ y = 5.5717 + \frac{42.2936}{1 + \left(\frac{x}{9.2245}\right)^{1.1938}} \]

EC50 = 9.2245, \( r^2 = 0.90 \).
Figure 4. Non-linear regression of capture success (#prey captured/10possible) versus turbidity (NTU). Spring/autumn temperature trials (●) are represented by a solid line 
\[
y = 0.9419 + \frac{8.3802}{1 + \left(\frac{x}{33.3398}\right)^{5.051}} \]
EC50 = 33.3398, \( r^2 = 0.88 \) and summer temperature trials (○) are represented by a dashed line 
\[
y = \frac{9.2034}{1 + \left(\frac{x}{29.011}\right)^{3.2812}} \]
EC50 = 29.011, \( r^2 = 0.70 \).
SUMMARY AND CONCLUSIONS

Understanding how turbidity affects non-game drift-feeding fishes is imperative to preserve fish diversity in North America under continuing human development of the landscape. Numerous studies have been conducted on the effects of turbidity on fish foraging, however very few of these have focused on drift-feeding fishes and those that do are concentrated on salmonids. It has also been shown that temperature has varying effects on capture success, however there are few, if any, studies that examine the combined effects of turbidity and temperature on fish foraging. Therefore, I examined the effects of turbidity at seasonal temperatures on reactive distance and capture success of rosyside dace (*Clinostomus funduloides*), a species of special concern in North Carolina.

Experiments were conducted in an artificial flume at both spring/autumn temperatures (12°C) and summer (18°C). For summer temperatures, I measured reactive distances and prey capture success for 85 individual fish (5 fish at 0, 10 and 15 NTU’s and 10 fish at each higher turbidity level) at 0, 5, 10, 15, 20, 25, 30, 35, 45 and 55 NTU’s. For the spring/autumn trials, I measured reactive distance and capture success for 80 individual fish (10 at each turbidity level) at 0, 8, 16, 24, 32, 40, 48 and 56 NTU’s. Spring/autumn treatment levels were sufficient to, which were sufficient to depict trends present in summer data with fewer treatments. These results were analyzed using a four parameter dose-response curve and with two-way ANOVA. This allowed for the determination of what turbidities would produce responses equal to 50% of the maximum response observed, as well as determining subtle effects between temperatures.
I showed that turbidity reduces reactive distance and capture success of rosyside dace at both spring/autumn and summer temperatures. Specifically, that turbidity reduces reactive distances by half of the maximum observed at intermediate turbidities (~9 NTU) and reduces capture success for by half of the maximum observed at high turbidities (~30 NTU) at both spring/autumn and summer temperatures. Reactive distance and capture success was significantly consistently better at spring/autumn temperatures than at summer temperatures. Although interesting, the ramifications of these temperature effects with turbidity is unknown, but should be taken into account when setting sediment regulations depending on the season. The shape of the dosage dependent reductions to reactive distance and capture success was curvilinear. This curvilinear response has been seen for reactive distance in other species (Vinyard and O’Brien 1976, Gregory and Northcote 1993, Benfield and Minello 1996, Sweka and Hartman 2001a, Sweka and Hartman 2003). This suggests that although different species have different reactive distances, the shape of the relationship between reactive distance and turbidity is similar. This lends credence to the further use of dosage dependent models in future studies. Of particular interest is the EC50 value, the concentration required to elicit a response equal to 50% of the maximum response observed, as it could prove useful in setting regulations on turbidity, TSS, and total maximum daily loads (TMDL’s) when managing the effects of turbidity on a given fishery.

In conclusion, relatively small amounts of red clay were required to produce turbidities that would reach the EC50 values that produced a 50% reduction in reactive distance and capture success in rosyside dace. If turbidity levels experienced by fishes are ephemeral then these values could carry limited ecological importance to continued foraging success. However, since
anthropogenic disturbances in North America usually cause prolonged changes to the watershed it is likely that these changes will have a long-term effect on sediment inputs to a given stream. These long-term increases in turbidity will, if not properly regulated, limit feeding in native fishes and may lead to reduced reproductive success in affected individuals allowing for a possible loss of fish diversity. Therefore, increased understanding of the mechanisms of how turbidity affects foraging in drift-feeding fish both seasonally and in a dosage dependent way should lead to the development of better management practices for impacted fisheries.

**Literature Cited**


