THE EFFECT OF FLOW REGIME AND TURBIDITY ON PERIPHYTON IN A LABORATORY SYSTEM

by

CLAYTON LAMAR BIRKETT

(Under the direction of Ernest W. Tollner)

ABSTRACT

This study was to design and develop a laboratory system for studying flow regime and turbidity interactions on periphyton development under three turbidity regimes. The flume was configured to create areas of distinct flow regime by using multiple roughness conditions and an in-channel weir. The one dimensional hydraulic model HEC-RAS was useful for modeling channel velocities and related properties, although the abrupt channel geometry and narrow channels appeared to cause some discrepancies in depth predictions. The results suggest that turbidity should be kept between 100 mg/l and 200 mg/l for periphyton growth, tending to confirm the 250 mg/l limit in the literature. Mean velocity was a fairly good predictor of live biomass (chlorophyll a) and filament length. The turbulent RMS velocity was the better predictor of dead biomass (pheophytin). Chlorophyll a concentration at the 100 mg/l sediment concentration was greatest, most likely due to periphyton compensation to turbidity-induced shading.

INDEX WORDS: Periphyton, Chlorophyll, Turbidity, Flow Regime, HEC-RAS

THE EFFECT OF FLOW REGIME AND TURBIDITY ON PERIPHYTON IN A LABORATORY SYSTEM

by

CLAYTON LAMAR BIRKETT

B.S. Louisiana State University, 2002

A Thesis Submitted to the Graduate Faculty of the University of Georgia in Partial Fulfillment of the Requirements for the Degree

MASTER OF SCIENCE

ATHENS, GEORGIA

2004

THE EFFECT OF FLOW REGIME AND TURBIDITY ON PERIPHYTON IN A LABORATORY SYSTEM

by

CLAYTON LAMAR BIRKETT

Major Professor: Ernest W. Tollner

Committee: David Gattie
Amy Rosemond

Electronic Version Approved:

Maureen Grasso
Dean of the Graduate School
University of Georgia
December 2004

TABLE OF CONTENTS

LIST OF TABLES

Page

LIST OF FIGURES

Page

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Today's society is constantly changing and adapting the environment around us. Half of the streams and rivers in the United States are becoming impaired by habitat degradation (Henley et al., 2000). With the increasing levels of urbanization, streams are being changed. Conversion of forested land to urban uses can result in increased nutrient concentrations, increased sedimentation, and altered hydrology (Rosemond et al, 2001). Sedimentation and turbidity are increasing as a result of the increased sediment input. Sediment is an integral component of stream ecosystems, but human activities have altered stream sediment loading and transport, dramatically increasing the amount of inorganic material delivered to waterways (Waters, 1995). Sources of sedimentation and turbidity include agriculture, forestry, mining, construction, and urbanization. Most alluvial riverbeds move periodically, meaning sediment transport has an influence on aquatic communities (ASCE, 1992). Excessive sedimentation and turbidity have been considered the most important factors in effecting aquatic organisms (Henley et al., 2000). Elevated sediment levels can negatively affect fish, aquatic insects, other invertebrates, and algae (Schofield et al., 2004). The increased turbidity can also result in an adverse effect on the organisms. Benthic organisms, which dwell on the bottom of waterways, are particularly affected by the increased turbidity of streams. Potential effects of sediment on benthic primary producers include: abrasion of periphyton by medium-sized particles such as sand, uprooting of macrophytes by larger particles, smothering of periphyton and rooted plants by heavy fine-sediment deposition, and reduction by high turbidity of the light necessary for photosynthesis

(Waters, 1995). Velocity is also a large component in determining the level of benthic organism activity. Velocities can reach levels that cause the habitats of benthic organisms to be washed away and thus making it more difficult for the organisms to thrive. Our project comes from the fact that few studies have addressed the role of hydraulic characteristics other than water velocity on stream periphyton (Mulholland et al., 1994). Factors such as tractive force, scouring effects, and turbulence can also influence periphyton biomass and growth rates.

Factors Affecting Lotic Systems

Algal biomass in river and streams are controlled by a large number of factors (Ghosh and Gaur, 1998; Rosemond 1993; Rosemond et al., 2001; Sabater et al., 2000; Vis et al., 1998). These factors can be organized into two main groups, biotic or biological factors and abiotic or non-biological factors. Biotic factors include herbivore and predator activity, and types of animal and plant species and abundance. Abiotic factors include turbidity and sedimentation, nutrient concentrations (water quality), irradiance, and water velocity. Turbidity, water velocity and nutrient concentrations are among the most important factors influencing periphyton biomass in streams. These factors can fluctuate stochastically, induced by rainfall, human activity, etc. (Saravia et al., 1998). Human activity is becoming a much larger problem to natural environments. Human activity includes recreational activity, construction, and destruction of natural habitat. Storms can cause high discharges that act as major disturbances to the structure of periphyton communities (Pringle and Hamazaki, 1997). Seasonality also affects which types of controls are dominant. Rosemond et al. (2000) found that in the Walker Branch grazing by snails was the dominant control in the spring, but nutrient limitation was more important in the summer.

There have been many studies involving the interaction between periphyton and nutrients. Studies have involved the periphyton dynamics involving nitrogen, N, (Havens et al., 1999; Vis et al., 1998; Eriksson and Weisner, 1996), phosphorus, P, (Rosemond et al., 1993; Dodds, 2003), and metals (Gold et al., 2002; Paulsson et al., 2002). Adey et al. (1993) states periphyton assemblages can play several roles that lead to increased retention of nutrients. First, they can remove nutrients from the water column and cause a net flux of nutrients toward the sediments. Second, they can slow water exchange across the sediment/water column boundary thus decreasing advective transport of P away from sediments. Third, they can intercept nutrients diffusing from the benthic sediments or senescent macrophytes. Fourth, they can cause biochemical conditions that favor P deposition. Finally, they can trap particulate material from the water column. McKnight et al. (2003) studied the impact of algal mats on the control of nutrients in glacial meltwater streams. The results showed that streams without algal mats contribute more nutrients to the lakes than streams with algal mats. Nitrate and soluble reactive P concentrations were also found to be lower in streams with abundant algal mats. Periphyton plays several roles in removing P from the water column, including P uptake and deposition, filtering particulate P from the water, and attenuating flow, which decreases advective transport of particulate and dissolved P from sediments (Dodds, 2003). Periphyton photosynthesis locally increases pH by up to 1 unit, which can lead to increased precipitation of calcium phosphate, concurrent deposition of carbonate-phosphate complexes, and long-term burial of P. Dodds et al. (1997) examined controlling periphyton biomass through the control of nutrients. This study found that a total N concentration below 350 μg/l and a total P concentration below 30 μg/l would result in a mean benthic algae chlorophyll *a* density below nuisance levels of 100 mg/m^2 in most streams.

Environmental Indicators

For algal indicators to be effective indicators of stream quality they must meet a set of environmental assessment criteria (McCormick and Cairns, 1994). Ideally, an indicator must be:

1. *biologically relevant* – easily related to the maintenance of ecological community
2. *socially relevant* – obvious value to those involved in the decision making process
3. *broadly applicable* to many stressors and sites
4. *sensitive* to stressors, preferably without an all or none response or excessive natural variability
5. *measurable* – can be operationally defined and quantified using acceptable procedures with known precision and accuracy
6. *interpretable* – capable of distinguishing acceptable from unacceptable conditions in a manner that is scientifically and legally defensible
7. capable of *continuity* of measurement *through time and space*
8. of an *appropriate spatial and temporal scale* for the assessment under study
9. *not redundant* with other measures included in the monitoring program
10. *integrative* by summarizing information from other sources
11. *timely* – capable of providing information rapidly enough that management actions can be implemented before unacceptable damage occurs
12. *diagnostic* of the particular stressor causing the problem
13. *cost-effective* by providing the maximum amount of information per unit effort
14. one for which an historical database exists in order to define a normal operating range and detect trends in ecosystem condition
15. *nondestructive* to the ecosystem.

Although no organism will meet all these criteria, to be feasible most must be obtained. Benthic organisms manage to be applicable to over half of these criteria within the scope of this experiment. The benthic organism that we will be focusing on is periphyton.

Periphyton

Ecologists define periphyton as communities of microorganisms that are associated with various aquatic substrates, such as rock. The microorganisms that make up these communities include: algae, bacteria, fungi and associated macrophytes, invertebrate grazers and detritus. Periphyton is usually dominated by cyanobacteria or blue-green algae (SERC). These are the main organisms, which we will be studying. Periphyton was chosen for a variety of reasons. First, periphyton have a naturally high number of species and are an important source of food for many of the higher-level organisms in the stream (Sabater et al., 2000). They are an important food resource. Secondly, periphyton is easily affected by changes in water quality and does not have as many natural defenses for these changes (McCormick and Cairns, 1994). Also, periphyton have a rapid response time to both exposure and recovery. In addition, periphyton can be easily sampled and identified. Periphyton also plays a major role in the metabolic conversion and partial removal of biodegradable material in rivers and streams (Saravia et al., 1998). Finally, the levels of tolerance or sensitivity to specific changes in environmental condition are known for many species (US EPA, 2002). Since periphyton are some of the most easily effected organisms in a stream system, they make good indicators of overall stream life quality. If the rates of velocity and turbidity affect periphyton growth rate, this effect will be felt all the way up the food chain.

Turbidity and Sediment Effects on Periphyton

Turbidity is caused by particles, dissolved substances, and organic and inorganic particulate and suspended matter in a body of water. The greater the turbidity the less clear the water appears. Turbidity also contributes to changes in water color. Turbidity can be caused by a number of natural and man-made sources. Natural sources include phytoplankton, natural erosion, and resuspended bottom sediments. Man-made sources include sediment inputs due to removal of natural habitat, wastewater discharges, and other discarded materials (Waters, 1995). Turbidity causes light to be scattered while passing through the water. The additional scattering of light can lead to less light reaching the bottom of waterways and can impact the photosynthetic process of algae. The reduced photosynthetic levels can lead to lower levels of daytime oxygen release (Water on the Web). Also, high levels of turbidity can lead to increased sedimentation, which can be harmful to benthic organisms in two ways. First, the increased sedimentation could cause the periphyton to become smothered under the increased soil organic matter. Also, the increased turbidity along with increased velocity could act as a scouring force on the riverbed. This force could scrape away the habitat of the periphyton and the periphyton themselves. As the stream order or size increases, biotic contributions to turbidity increase (Henley et al., 2000). Turbidity also affects the amount of toxic chemicals and ionic activity in the water. The majority of toxic chemicals have environmental pathways that are primarily or exclusively associated with sediment and biological substrates and large amounts of ions are carried into streams on sediment particles; these processes have adverse effects on wildlife and plants (Droppo and Mitchell, 2001; Milan et al., 2001; Waters, 1995). Suspended sediment may be measured in terms of concentration of total particulate solids in parts per million (ppm=mg/L) (Waters, 1995).

The size of sediment particles is very important to the effect produced upon stream communities. An original grade scale for determining sediment size was developed by the American Geophysical Union in 1947 and comprised 24 categories of sizes and class names (Waters, 1995). However the commonly used classification of sediment categories was developed by Cummins (1962) and it included 11 particle sizes and names. In most sediment research, grain-size is given in phi intervals. Phi diameter is computed by taking the negative log of the diameter in millimeters. Phi diameters are used to simplify graphical presentations and statistical computations.

Table 1. -Sizes of sediment Particles (modified from Cummins 1962).

Category	Size Range	Phi scale
Boulder	>256mm	-8
Cobble	64-256mm	-6,-7
Pebble	16-64mm	-4,-5
Gravel	2-16mm	-1,-2,-3
Very coarse sand	1-2mm	0
Coarse sand	0.5-1mm	1
Medium sand	0.25-0.5mm	2
Fine sand	0.125-0.25mm	3
Very fine sand	0.0625-0.125mm	4
Silt	4-62um	5,6,7,8
Clay	<4um	9

Sediment size may vary with time and geophysical processes. Inorganic particles down through the size of silt are derived largely from the physical breakdown and weathering of silicate rocks (Waters, 1995). Particle size characteristics of suspended sediment exert fundamental control on its settling velocity (Walling et al., 2000). The larger a particle, the higher its settling velocity. The shape of sediment also plays a role in interactions with the environment. Holomuzki and Biggs (2003) found that grain shape significantly affects sediment transport thresholds. Water velocity effects the movement of sediment in streams, resulting in scouring and tractive forces that are injurious to periphyton biomass.

One Dimensional Hydraulic Modeling

Computer models are being used to determine and predict water flow conditions. The program WSP, water surface profile, computes normal and critical depth, classifies the water surface profile, and computes the water surface profile using the direct step method. These programs perform a one-dimensional integration of the continuity equation and energy (or momentum) equations, using the implicit method described in references such as French (1985). Visual Basic programs such as YOYC and YCOMP have also been used to predict water profiles such as critical depth. The U.S. Army Corps of Engineers has developed two methods for modeling stream dynamics. HEC-RAS is the most current version of their software and is described by Bruner (2002). It uses the standard step method (French, 1985) to compute a variety of stream flow characteristics. HEC-RAS is a modeling tool that enables one to compute a wide variety of fluid parameters in any stream where flow in one dimension is sufficient to describe the system.

Flow Regimes

There are several major flow regimes. Flows may be laminar-subcritical, laminar critical, and laminar supercritical (the latter two are unlikely). The subcritical, critical and supercritical flow may, and usually do, exist as turbulent flows. Particles in laminar flow move in definite and observable streamlines, which is rare in open channels. Laminar flow is characteristic of a viscous fluid or at least a flow in which viscosity plays a significant part (Jobson and Froehlich, 1988). Laminar flow is expected in open channels only when the depths are very small and velocities are very low, a condition which is rare in open channels. Turbulent flow is characterized by the appearance of the fluid moving with a mean velocity underlying an additional velocity component known as the root mean square (RMS) turbulent velocity and occurs when the Reynolds number is greater than 2000, where Reynolds number is defined as

$$R_e = v\rho l/\mu = vl/\nu \quad (1)$$

where
v = velocity of flow
ρ = density of fluid
l = characteristic length dimension
depth, μ = dynamic viscosity
ν = kinematic viscosity.

In open channels flow is driven by gravity. The Reynolds number is the ratio of inertial to viscous forces in the flow. Turbulent flow commonly occurs in streams that appear to be very smoothly flowing and in which there is no detectable source of disturbance (Jobson and Froehlich, 1988). Sediment transport along the flow and oxygen transport into the flow is enhanced by turbulence.

The Froude number is used to classify criticality of flow. Froude number is defined as

$$Fr = v/\sqrt{gD_h} = q/A\sqrt{gD_h} \quad (2)$$

where the hydraulic depth D_h is the cross-sectional area divided by the top width and g represents the gravitational constant. Flows are subcritical, with a Froude number of less than one, if a wave will propagate upstream from a disturbance. Flows behaving as a shooting jet in supercritical flows have a Froude number greater than one. Flows over weirs are characterized as critical flows and have a Froude number equal to one.

Laws of similitude must be satisfied to set the similarity of flows between models and prototypes. Geometric, kinematic and dynamic similitude are all needed to make accurate scale models of natural systems in the laboratory. Dynamic similitude is the condition wherein forces, flowrates and fluid properties are scalable in accordance with acting of the corresponding fluid masses related by dimensionless ratios. The dimensionless ratios are determined using well developed approaches (Jobson and Froelich, 1988). Many turbulent flow and open channel flow scenarios can be modeled based on Froude number similitude. Froude number similarity can be easily achieved so the model approach is ideal for rapidly varied flow problems where the gravity force dominates the flow (Jobson and Froelich, 1988). The Froude force ratio is given by the formula

$$v_m/\sqrt{l_m g} = v_p/\sqrt{l_p g} \quad (3)$$

where
v_m = characteristic velocity of model
l_m = characteristic length of model
v_p = characteristic velocity of prototype
l_p = characteristic length of prototype
g = gravity

Weirs, culverts and dams have all been defined by model studies and are assumed to apply to full scale situations with an equal Froude number and geometric similarity. It is envisioned that one can study many natural channel and engineered structures using Froude similitude to scale down to the laboratory scale.

Developing a Hydraulic Microcosm for Resolving Turbidity, Flow and Nutrient Interactions

The objective of this experiment is to develop a physical and computer model of a natural or engineered channel structure to determine the interactions between turbidity and flow factors on periphyton biomass. A flume was used to develop a controllable system to isolate and control these relationships. An arbitrarily selected structure was selected for the flume configuration tested in this study. Using multiple areas of flow regime we can examine how sediment concentration impacts periphyton. HEC-RAS will be used to model the stream characteristics of the flume. HEC-RAS is a computer program used in one-dimensional steady and unsteady flow analysis. If the HEC-RAS flow data is not significantly different from the data collected from the flume then it can be assumed that the HEC-RAS generated computer model flow characteristics are accurate and can then be extrapolated to a larger scale. The turbulence, tractive force, velocity and Froude number data obtained from the HEC-RAS model and the flume can be used to determine environmental control criteria in the natural rivers. By knowing the water flow criteria needed to maintain periphyton biomass we can develop strategies to prevent periphyton biomass loss and therefore help to limit human impacts on the environment. In the future, components can be added to this experiment to further develop the interactions between turbidity, flow factors, and potentially nutrients.

REFERENCES

Adey, W., Luckett, C. and K. Jensen. 1993. Phosphorus removal from natural waters using controlled algal production. *Restor. Ecol*. 1:29-39

Anon. Armfield Engineering Teaching and Research Equipment Instruction Manual S8 MkII Sediment Transport Channel. Issue 6 Dec 1995.

Anon. 2003. B-16 Hydraulic Demonstration Channel. Engineering Laboratory Design, Inc. Lake City, MN.

ASCE task Force. 1992. Sediment and aquatic habitat in river systems. *Journal of Hydraulic Engineering* 118(5):669-687.

Bishop, Robert H., Learning with LabVIEW 6i. Prentice Hall Inc. Upper Saddle River, NJ. 2001.

Biggs, B.J.F., R.A. Smith and M.J. Duncan 1999. Velocity and sediment disturbance of periphyton in headwater streams: biomass and metabolism. *Journal of North American Benthological Society* 18(2):222-241.

Bruner, G. W. 2002. HEC-RAS River analyses system hydraulic reference manual, version 3.1. US Army Corps of Engineers, Hydrologic Engineering Center, Davis, CA.

Cummins, K.W. 1692. An evaluation of some techniques for the collection and analysis of benthic samples with special emphasis on lotic waters. *American Midland Naturalist* 67: 477-504.

Dodds, W.K., V.H. Smith, and B. Zander. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: a case study of the Clark Fork River. Water Research 31(7): 1738-1750.

Dodds, W.K. 2003. The role of periphyton in phosphorus retention in shallow freshwater aquatic systems. *J. Phycol.* 39:840-849.

Droppo, I.G., and Y.L. Lau, and C. Mitchell. 2001. The effect of depositional history on contaminated bed sediment stability. *The Science of the Total Environment* 266:7-13

Environmental Protection Division Georgia State Government http://www.state.ga.us/dnr/environ/plans_files/plans/oconee-pdf/chapt-2.pdf 1998

Eriksson, P.G. and S.E.B. Weisner. 1996. Functional differences in epiphytic microbial communities in nutrient-rich freshwater ecosystems: an assay of denitrifying capacity. *Freshwater Biology* 36:555-562.

French, Richard H. 1985. Open-Channel Hydraulics. McGraw-Hill, Inc. New York.

Ghosh, M. and J.P. Gaur. 1998. Current velocity and the establishment of stream algal periphyton communities. *Aquatic Botany* 60:1-10.

Gold, C., A. Feurtet-Mazel, M. Coste, and A. Boudou. 2002. Field transfer of periphytic diatom communities to assess short-term structural effects of metals (Cd, Zn) in rivers. *Water Research* 36(14): 3654-3664.

Havens, K.E., T.L. East, A.J. Rodusky and B. Sharfstein. 1999. Littoral perihyton responses to nitrogen and phosphorus: an experimental study in a subtropical lake. *Aquatic Botany* 63(3-4): 267-290.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Reviews in Fisheries Science* 8(2):125-139.

Holomuzki, J.R. and B.J.F. Biggs. 2003. Sediment texture mediates high-flow effects on lotic macroinvertebrates. *Journal of the North American Benthological Society* 22(4):542-553.

Jobson, H.E. and D. C. Froehlich. Basic Hydraulic Principles of Open-Channel Flow. U.S. Geological Survey Open-File Report 88.707, 1988.

Lau, Y.L., and D. Liu. 1993. Effect of flow rate on biofilm accumulation in open channels. *Water Research*. 27(3): 355-360

McCormick, V. and J. Cairns. 1994. Algae as indicators of environmental change. *Journal of Applied Phycology* 6:509-526.

McKnight, D.M., R.L. Runkel, C.M. Tate, J.H. Duff and D.L. Moorhead. 2003. Inorganic N and P dynamics of Antarctic glacial meltwater streams as controlled by hyporheic exchange and benthic autotrophic communities. *Journal of the North American Benthological Society* 23(2):171-188.

Milan, D.J., G.L. Heritage, A.R.G. Large, and M.E. Charlton. 2001. Stage dependent variability in tractive force distribution through a riffle-pool sequence. *Catena* 44:85-109.

Mulholland P.J., A.D. Steinman, E.R. Marzolf, D.R. Hart, and D.L. DeAngelis. 1994. Effect of periphyton biomass on hydraulic characteristics and nutrient cycling in streams. *Oecologia* 98:40-47

Paulsson, M., Mansson, V., and H. Blanck. 2002. Effects of zinc on the phosphorus availability to periphyton communities from the river Göta Älv**.** *Aquatic Toxicology* 56(2): 103-113.

Pringle, C.M. and G.A. Blake. 1994. Quantitative effect of Atyid Shrimp (Decapoda: Atyidae) on the depositional environment in a tropical stream: use of electricity for experimental exclusion. *Can. J. Fish. Aquat. Sci.* 51: 1443-1450.

Pringle, C.M. and T. Hamazaki. 1997. Effects of fishes on algal response to storms in a tropical stream. *Ecology* 78(8):2432-2442.

Rosemond, A.D. 1993. Interactions among irradiance, nutrients, and herbivores constrain a stream algal community. *Oecologia* 94: 585-594.

Rosemond, A.D., P.J. Mulholland, and J.W. Elwood. 1993. Top-down and bottom-up control of stream periphyton: effects of nutrients and herbivores. *Ecology* 74(4) 1264-1280.

Rosemond, A.D. 1994. Multiple factors limit seasonal variation in periphyton in a forest stream. *J.N. Am. Benthol. Soc.* 13(3): 333-344.

Rosemond, A.D., P.J. Mulholland, and S.H. Brawley. 2000. Seasonally shifting limitation of stream periphyton: response of algal populations and assemblage biomass and productivity to variation in light, nutrients and herbivores. *Can. J. Fish. Aquat. Sci*. 57:66-75.

Rosemond, A.D., H.S. Weyers, M.J. Paul, J.L. Greenwood, and D.S. Leigh. 2001. Benthic algal biomass in the Etowah basin and implications to establishing nutrient criteria in streams. *Proceedings of the 2001 Georgia Water Resources Conference.* K.J. Hatcher, editor, Institute of Ecology, the University of Georgia, Athens, Ga.

Sabater, S., J. Armengol, E. Comas, F. Sabater, I. Urrizalqui, I. Urrutia. 2000. Algal Biomass in a disturbed Atlantic River: water quality relationships and environmental implications. *The Science of the Total Environment* 263:185-195.

Schofield, K.A., C.M. Pringle, and J.L. Meyer. 2004. Effects of increased bedload on algal and detrital-based stream food webs: experimental manipulation of sediment and macroconsumers. *Limnology and Oceanography* 49(4): 900-909

Saravia, L.A., F. Momo, and L.D. Boffi Lissin. 1998. Modelling periphyton dynamics in running water. *Ecological Modelling* 114:35-47

Southeast Environmental Research Center (SERC) at Florida International http://serc.fiu.edu/periphyton/periphyton.htm

Sturm, Terry W. 2001, Open-Channel Hydraulics. McGraw-Hill, Inc. Boston, MA.

United States Environmental Protection Agency http://www.epa.gov/bioindicators/html/periphyton.html 2 August 2002

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canada Journal of Fish and Aquatic Science* 37:130-137.

Vis C., C. Hudon, A. Cattaneo, B. Pinel-Alloul. 1998. Periphyton as an indicator of water quality in the St Lawrence River (Quebec, Canada). *Environmental Pollution* 101:13-24.

Walling, D.E., P.N. Owens, B.D. Waterfall, G.J.L. Leeks, and P.D. Wass. 2000. The particle size characteristics of fluvial suspended sediment in the Humber and Tweed catchments, UK. *The Science of the Total Environment* 251/252: 205-222.

Water on the Web http://wow.nrri.umn.edu/wow/under/parameters/turbidity.html

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects, and Control. American Fisheries Society, Monograph 7 Bethesda, Md.

CHAPTER 2

THE EFFECT OF FLOW REGIME AND TURBIDITY ON PERIPHYTON IN A MODEL CHANNEL[1]

[1] Birkett, C.L. and E.W. Tollner. 2004. To be submitted to Ecological Engineering

ABSTRACT Urbanization and environmental concerns frequently clash. Urbanization has lead to a decrease in the natural environment that controlled runoff volume and erosion. Periphyton was used in this experiment to determine its function as an ecological indicator. This study was to design and develop a laboratory system for studying flow regime and turbidity interactions on periphyton development. The flume was configured to create areas of distinct flow regime by using multiple roughness conditions and an in-channel weir. The one dimensional hydraulic model HEC-RAS was useful for modeling channel velocities and related properties, although the abrupt channel geometry and narrow channels appeared to cause some discrepancies in depth predictions. Periphyton was analyzed under sediment concentrations of 0, 100, and 200 mg/l. The results suggest that turbidity should be kept between 100 mg/l and 200 mg/l for periphyton growth. Increasing shear forces may limit filament length but do not necessarily reduce biomass unless turbidity is high. Mean velocity was a fairly good predictor of live biomass (chlorophyll a) and filament length. The turbulent RMS velocity was the best predictor of dead biomass (pheophytin). Chlorophyll a concentration at the 100 mg/l sediment concentration was greatest most likely due to periphyton compensation to turbidity-induced shading. Periphyton filament length showed the greatest change in the low velocity testing regions as sediment concentration increased. In the presence of added turbidity, periphyton biomass was significantly impacted by the 200 mg/l sediment concentration, tending to confirm the limit of 250 mg/l for periphyton growth in the literature. Industrial enterprises, such as waste treatment systems and stormwater management

systems, may tailor discharge rates to accomplish healthy periphyton biomass, which would be beneficial in nutrient retention and depletion. Aquacultural enterprises can use this information to develop better farming and feeding practices to improve water quality as well as provide a base food source.

KEYWORDS. Periphyton, Chlorophyll, Turbidity, HEC-RAS, Flow Regime

INTRODUCTION

Over the course of time human impact on the environment has become an intensifying problem, especially in streams and lakes. Half of the streams and rivers in the United States are becoming impaired by habitat degradation (Henley et al., 2000). With a growing awareness of the value of natural ecosystems, a move has been started from manipulation and control of rivers to restoration and conservation (Bockelmann et al., 2004). Over the past two decades, biological monitoring has risen to the forefront of environmental impact assessments and stream monitoring programs. Stream biological integrity reveals itself in the condition, abundance, and diversity of its biota (Hill et al, 2003). There is also a desire to evaluate storm water management and aquacultural production systems to improve ecological sustainability. Periphyton are a fundamental link in many aquaculture production systems, waste treatment systems, and stormwater management systems.

One of the many benthic organisms affected by human activities is periphyton. Periphyton is used as a biological component to examine the quality and pollution of surface waters (Sansone et al., 1998). Periphyton have a naturally high number of species and are an important source of food for many of the higher-level organisms in the stream (Sabater et al., 2000). Periphyton also plays a major role in the metabolic conversion and partial removal of biodegradable material in rivers and streams (Saravia et al., 1998) and aquacultural systems. Periphyton biomass is affected by many biotic and abiotic factors. Harding et al. (1999) indicated that agricultural intensity and physical conditions associated with agriculture activity (e.g. impacted waters, high turbidity and temperature) were strongly associated with the composition of benthic assemblages.

Studies have shown that turbulence and shear velocity have important effects on periphyton biomass (Bergstedt et al., 2004; Quinn et al., 1996; Nikora et al., 1997). Shear velocity and turbulence are factors in the effect of tractive force on the bottom of a stream system. Tractive force, the force exerted by flowing water, can be a factor in the attachment processes of periphyton by making the habitat unsuitable and by also physically removing the periphyton from the attached substrate (Asaeda and Hong Son, 2000; Biggs and Thomsen, 1995; Lau and Liu, 1993).

The detachment of periphyton must be considered when determining the affect of flow regime on periphyton. Increased exposure to turbulence increases the sloughing of epiphytic organisms from surfaces (Eriksson and Weisner, 1996). In addition to affecting growth, fluid velocity also affects the height to which the periphyton can grow (Nikora et al., 1997). Saravia et al. (1998) examined the degree of sloughing in periphyton. It was found that the degree of sloughing was greater in low velocity sites, meaning that changes in current velocity had a greater influence at these sites than in the rapid current ones. Also, current velocity was found to be negatively related to biomass as one would expect due to increased flow resistance.

The theoretical detachment curve in Hondzo and Wang (2002) shows periphyton persistence as a function of the nondimensional critical shear stress and decreasing L/d ratios. It was determined that algal removal increased as shear rate increases. These studies suggest that one should account for the effects of periphyton at the boundary of systems involving open or closed conduit hydraulic processes. The Hondzo and Wang (2002) detachment curves do not account for turbidity and are oriented toward micromorphology rather than macroscopic filament length.

Urbanization has reduced capacity of the environment to control runoff volume and erosion through the removal of natural barriers such as trees. Ecological ramifications of sediment loads have been observed, including destruction of habitat, decreased fecundity, and inhibited feeding (Parkhill and Gulliver, 2002). Elevated sediment levels can negatively affect fish, aquatic insects, other invertebrates, and algae (Schofield et al., 2004). Light attenuation is also effected by increases in sediment levels. Eroded silts and clays seriously reduce light penetration in some streams (Hill, 1996). Our study examines how sediment levels in conjunction with flow regime affect periphyton biomass and filament length

Nikora et al. (2002) states that knowledge of mass transfer processes and physical interactions between periphyton and the turbulent stream flow, which control periphyton growth and losses, may become critical to understanding the functioning of natural benthic communities in streams. Studies have examined the interactions of turbidity on periphyton in natural streams, but this project goes one step further and examines the interactions between turbidity and flow regime and their affect on periphyton biomass. Filamentous algal assemblages can serve as filters or create areas of reduced flow, which has the net effect of removing suspended particulate materials from the water column (Stevenson, 1996), thus the significance of filament length along with biomass. By further exploring the relationship between periphyton biomass, turbidity, and flow regime guidelines can be established that can be used to design channels to be both engineering and ecologically satisfying. The results of this research can be applicable to the design of aquacultural systems, stormwater management systems, and wastewater discharge channels in two primary ways. First, control of flow regimes and turbidity in aquacultural facilities, wastewater treatment plants, and stormwater management channels could allow for control of periphyton biomass to help in nutrient retention (Vis et al., 1998; Dodds, 2003; Adey

et al., 1993; Graneli and Solander, 1988), removal of biodegradable material and metals (Saravia et al., 1998; Lau and Liu, 1993; Paulsson et al., 2002), and ecosystem productivity. Secondly, wastewater discharge carries large amounts of nutrients that can lead to large algal assemblages which can lead to eutrophication (Bokn et al., 2002; Mattila and Raisanen, 1998; Riegman et al., 1992; Boynton et al., 1995). By finding the effect of flow regime on periphyton biomass, changes in flow regime could be used to manage periphyton populations. This study uses a laboratory system to enable control of turbidity and flow regime (in addition to other nutrient parameters if desired) in an effort to isolate respective effects on periphyton biomass and filament length.

OBJECTIVES

Using the laboratory flume we tried to achieve three specific objectives: 1) to determine the feasibility of using HEC-RAS to model small laboratory flumes, 2) to determine the effect of selected flow parameters and turbidity on periphyton biomass indicators, and 3) to determine the effect of selected flow parameters and turbidity on macrofilament length.

MATERIALS AND METHODS

The experiments were performed using an Engineering Laboratory Design, Inc. B-16 Hydraulic Demonstration Channel. The channel is 3.048m (10ft.) by 0.3048m (1ft.) by 0.3048m (1ft). It has two pumps, a collecting tank, and two orifice meters for reading pressure measurements.

The flume was divided into two identical waterways. A 0.3048m length of 25.4mm gravel was placed 0.3048m from the entrance of the flume to dissipate an entrance hydraulic, and to provide an additional testing region. There were 42 test tiles, 1 placed on the rocks, 8 placed

in downslope from the weir, 4 on top of the weir, and 8 placed upslope from a weir. The setup is shown in figure 2.1.

Figure 2.1 Laboratory flume setup showing the rock bottom and tiled zone up-stream from the weir and the tiled zone below the weir.

An IFA-300 Constant Temperature Anemometer equipped with a one dimensional probe was used to measure the mean velocity and turbulent (RMS) velocity at each testing site. Isotropic turbulence was assumed. Anemometers have been used to measure flow rate and turbulence parameters in various studies (Hayashi et al., 1988; Nezu et al., 1997; Stefes and Fernholz, 2003). Once the anemometer was calibrated, the flume was started and mean velocity and turbulent (RMS) velocity measurements were made at a designated place on the rocks, upslope tile, weir, and downslope tile. Mean and RMS velocities were measured at six tenths the depth of the water to enable comparison with HEC-RAS average velocity predictions.

The flume system was modeled using HEC-RAS 3.1.2. HEC-RAS is an acronym representing Hydrologic Engineering Centers - River Analysis System and was designed by contractors for the US Army Corps of Engineers. HEC-RAS is a computer program used in one-

dimensional steady and unsteady flow analysis with mixed subcritical and supercritical flow. It uses the implicit methods in solving the gradually varied flow equation. Known flow depth at the inlet and outlet boundaries was used to model the flow through a waterway in the flume. The bottom height and roughness condition of the channel was changed to mimic the change in height from the rocks to the tiles to the bottom of the flume. A flowrate of 0.0047855 m^3/s was used for study. The Manning's n values were initially assigned as 0.009 for the acrylic flume itself, 0.013 for the divided plexiglas, 0.018 for the tiles, and 0.028 for the rocks. Known water surfaces were used as (0.0496824 m) and outlet (0.0161544 m) boundary values as measured with a hook gauge. The velocity comparison between the anemometer and the HEC-RAS model was made to determine the viability of the computer model. The anemometer was used to measure local velocity at approximately 0.6 times the depth in the center of the channel, which is generally accepted to be the channel average velocity. The HEC-RAS velocity was computed from the average velocity over the entire testing region.

The periphyton used in the study was *Stigeoclonium tenue*, which occurs naturally in Georgia streams. The *Stigeoclonium tenue* samples were obtained from UTEX, The Culture Collection of Algae at the University of Texas at Austin[1]. Three 10ml samples of *Stigeoclonium* were obtained to start the experiment.

The periphyton was grown on prepared 75 mm by 75 mm white ceramic tiles, discussed previously facilitate periphyton recovery for photosynthesis measurements. The tiles were placed in an incubator for initial biomass. The *Stigeoclonium* was grown in four 305 mm by 152 mm by 51 mm glass baking dishes with 12 test tiles per dish. A standard freshwater Wright-Chu (WC) medium (Guillard and Lorenzen, 1972) was used to culture the periphyton. The

[1] The University of Texas at Austin, The Culture Collection of Algae (UTEX) 1 University Station A6700 Austin, TX 78712-0183

testing tiles were scraped every 2-3 days to simulate grazing. The baking dishes and tiles were placed inside an incubator set at 20^{o}C and with a light/dark cycle of 13/11 hrs. The periphyton was allowed to grow for three weeks before being moved into the flume.

The testing cycle was designed to run 20 days. WC medium was used as the water and nutrient source and was changed every five days. Deionized water was pumped from a 3,785 l holding tank into the flume reservoir. The WC medium was mixed in the reservoir. At the end of the testing period the testing tiles were transferred out of the flume and analyzed to determine chlorophyll a and pheophytin using the procedure in Hauer and Lamberti (1996).

The concentration of chlorophyll a indicated periphyton live cells, while the pheophytin concentration indicated periphyton dead/scensenent cells. After each testing cycle, the flume was drained and wiped with a cloth to remove the periphyton that had grown on the side of the flume. During testing, the length of the periphyton filaments was measured to determine if there was a difference in each testing region. Algal tiles were allowed to sit in standing water so that the filaments were fully extended, thereby enabling filament length measurement. Then the filament lengths were measured randomly and averaged.

Reagent grade kaolin[2] was added as the sediment source for the turbidity runs. Colloidal kaolin was chosen to minimize damage to the recirculation pumps. Sediment concentrations were set at 0mg/l, 100 mg/l, and 200 mg/l. The sediment concentrations were set below the 284 mg/l limit for periphyton growth (EPA, 2001).

A LI-COR quantum/radiometer/photometer model Li 189[3] was used to measure the photosynthetic active radiation (PAR) at the maximum water depth of 10.496 cm with varying sediment concentrations in a simulated flume environment. This simulated environment was

[2] ALDRICH Chemical Company, Inc., P.O. Box 355 Milwaukee, WI 53201
[3] Electronic System Ecosystem 2229 Fifth St. Berkeley, CA 94710

created by mixing the sediment concentration in a beaker with the previously mentioned depth and placing the beaker and light intensity detector under the growth light at the same distance from the light. Selected water quality tests [pH, conductivity, BOD, total N, total P, Ortho-PO_4, ammonia, and nitrate/nitrite ratio] were run every 20 days at every sediment concentration to assess any change in nutrients. The entire experiment was replicated twice at each sediment concentration. The results were analyzed using NCSS (2001).

RESULTS AND DISCUSSION

Small Flume Flow Measurements and Modeling

The first objective was to understand the performance of the flume system based on modeling and experimental measurements. A study was done to select the optimum Manning n values for the acrylic flume, tiles and rock materials. Figure 2.2 shows the flume diagram with water flow.

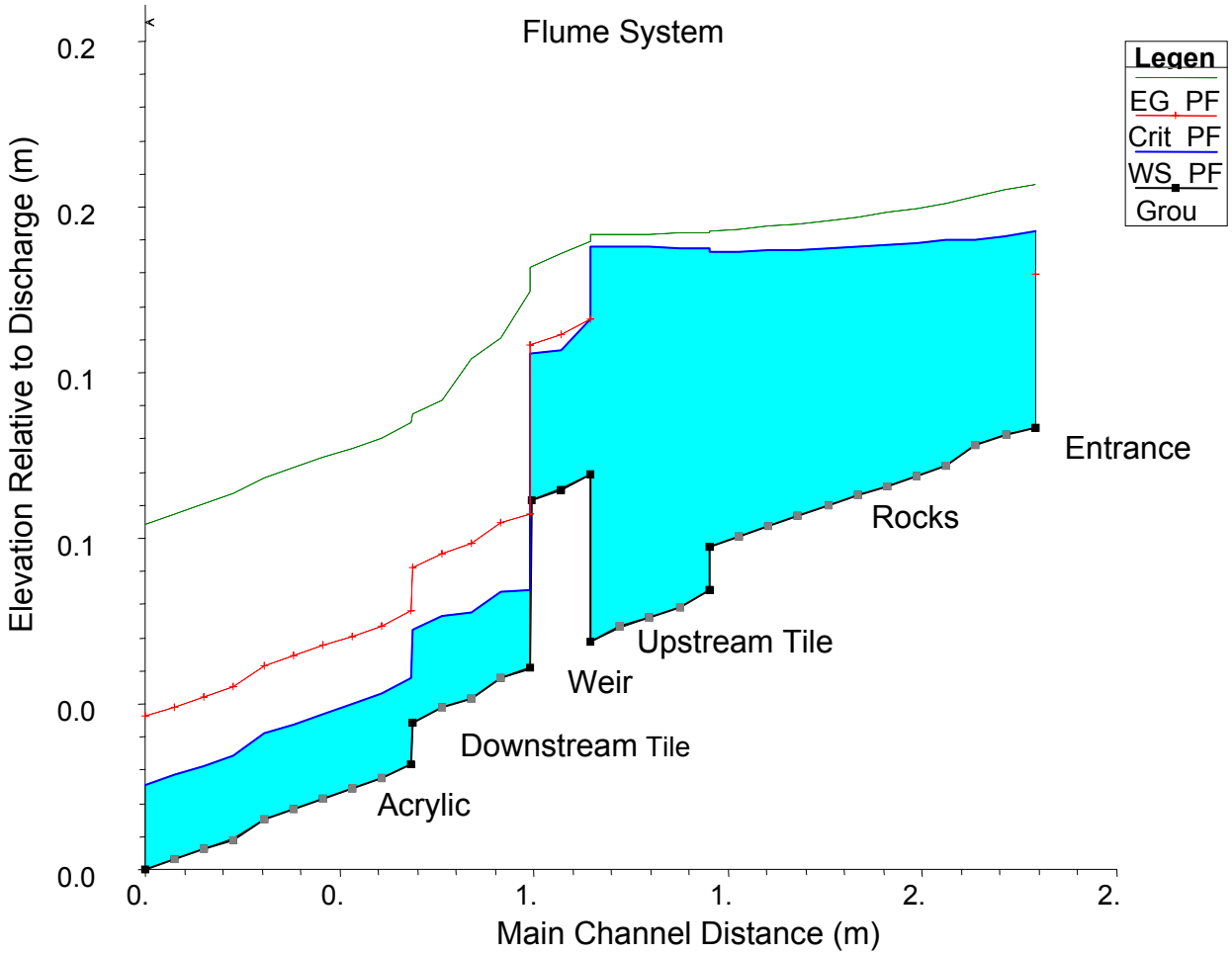


Figure 2.2 Channel flow profile showing HEC-RAS modeled water depth, critical flow depth and energy grade line and annotated to show the major channel features.

HEC-RAS depth was compared to the actual flume depth measured by a hook gauge. The depth in the HEC-RAS model was on average 0.019 m higher (e.g., about twice the thickness of the tiles and half the thickness of the rock layer) than actual mean depth measured from a hook gauge. The increased depth was attributed to the inability of HEC-RAS to model rapid transitions in flow profile. Flow through the rocks and to a lesser extent between the tiles may have contributed to the discrepancy in the depth measurements. The narrowness of the channel may also have contributed some two-dimensional effects that HEC-RAS was unable to adequately model.

Anemometer mean velocity measurements are shown in Figures 2.3.

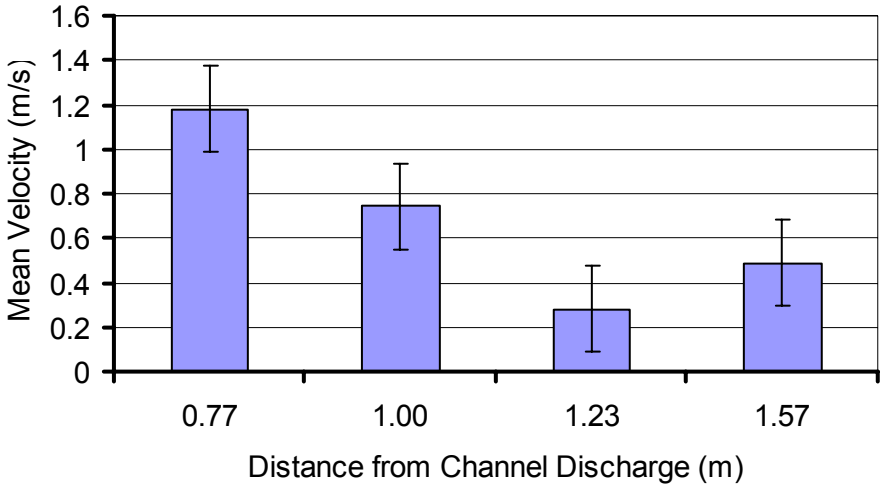


Figure 2.3 Mean velocity as a function of average distance from the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

The mean velocity readings were highest at the downslope testing sections following weir passage and lowest at the upslope testing sections prior to weir passage. The errors shown are consistent with measurements in turbulent flows.

RMS velocity measurements were made with the anemometer along the channel center, with results shown in Figure 2.4. The highest RMS velocity was in the downslope testing section followed closely by the upslope testing region. In the downslope region, the result of the waster fall after the weir and the shallow depth contribute to the high RMS velocity. In the upslope testing region, the weir causes a backup in water velocity, creating eddies that influence RMS velocity. The lowest RMS velocity was found immediately above the weir. The highest RMS velocity was in the downslope testing section, followed closely by the upslope testing region. These two readings were expected. In the downslope region, the result of the waster fall after the weir and the shallow depth contribute to the high RMS velocity. In the upslope testing region, the weir causes a backup in water velocity, creating eddies that influence RMS velocity. The lowest RMS velocity was found immediately above the weir.

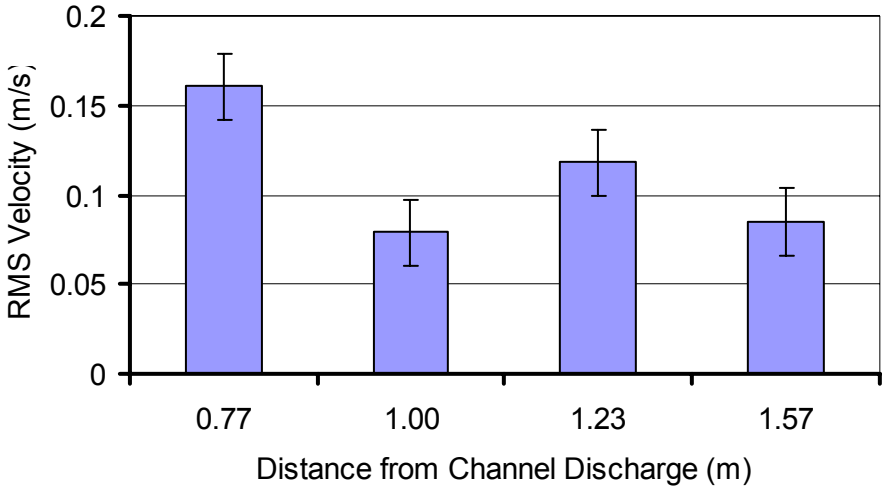


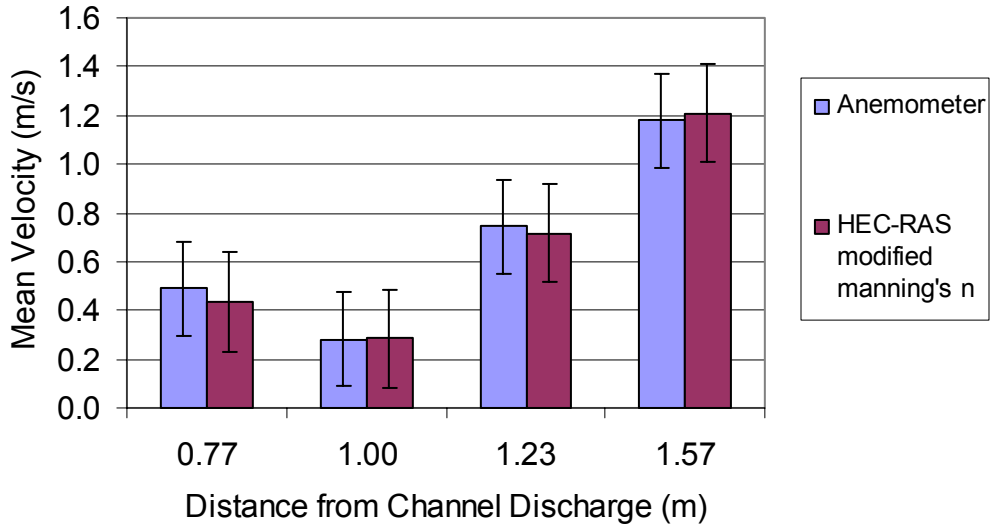
Figure 2.4 RMS velocity as a function of average distance from the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

Table 2.1 shows the results and comparison of the anemometer and HEC-RAS velocity values. The HEC-RAS mean velocity values were about -3 to 4% different from the anemometer mean velocity. In the rock section the difference was 11%. The differences between the HEC-RAS and measured mean velocities are most likely due to calibration issues and to the fact that the classical logarithmic profile and associated boundary layer development in open channel flow is not well developed in the flume due to the short reach. The shallow depths in the flume caused some problems with achieving the 0.6 depth placement of the probe (the approximate point of mean velocity in a logarithmic profile flow), further contributing to error possibilities. In the rock testing region, some flow occurred within the rock media and the shape of the rocks may have caused localized changes in the anemometer mean velocity.

Table 2.1 Mean velocity data from the anemometer and HEC-RAS

Position from channel discharge (m)	Channel Feature	Anemometer Mean Velocity (m/s)	Percent Difference[a]			
			normal n	high n	low n	selected n
0.77	rock	1.18	-11	-3	-13	-3
1.00	upslope	0.75	2	4	3	4
1.23	weir	0.28	-2	-2	-2	-2
1.57	downslope	0.49	11	14	11	11

[a] Percent difference = [(Anemometer mean velocity-HEC-RAS mean velocity)/Anemometer mean velocity]*100



I

Figure 2.5 Comparison of mean velocities as a function of average distance from the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

The velocities in channels 1 and 2 showed no significant difference in the upper and lower testing regions, but did show a significant difference in the rock section at 95%. The difference between the velocities in each channel was attributed to the difference in the rock structure in the section. At 60% of the depth, the rock could have affected the water flow in each channel. The lack of developed boundaries may have resulted in higher than expected anemometer readings. At each testing region, anemometer readings were collected five times and an average was attained.

Modeling the flume system in HEC-RAS using a weir instead of a raised section gives lower velocity readings than actually occur in the flume. Changing the weir coefficient changes the mean velocities only slightly. The best weir coefficient for mimicking the flume is 3.9. The HEC-RAS velocities with this weir coefficient give mean velocities with percent differences of 14% in the rocks, -2% in the upslope section, 37% on the weir, and 35% in the downslope section. Due to the small size of the channel, the weir dominates flow and does not allow velocity to change.

HEC-RAS modeled Froude number indicates supercritical flow below the weir and subcritical flow above the weir, with critical flow at the weir as one would expect. Figure 2.5 shows the Froude number vs. channel position. The Froude number data obtained from HEC-RAS supports this assumption. Through the comparison of mean velocities and Froude number results with HEC-RAS and the mean velocities from the anemometer, it can be suggested that HEC-RAS can be used to accurately model small flume systems especially when one compensates for depth deviations.

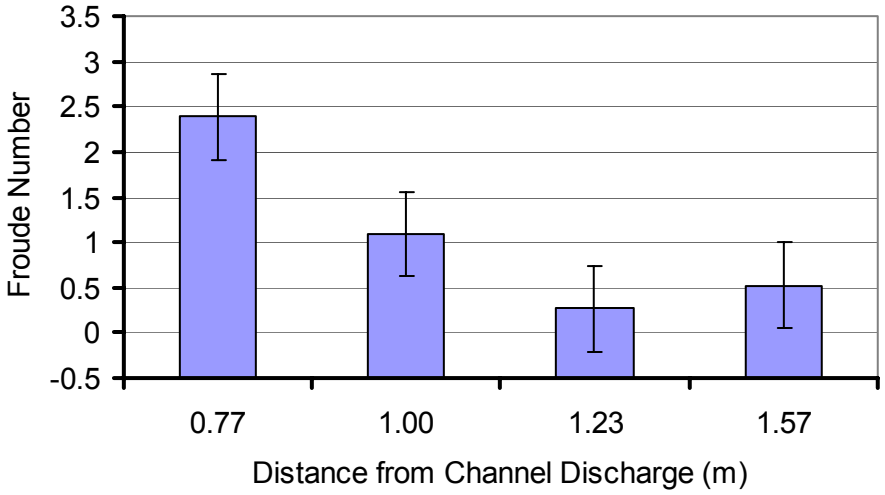


Figure 2.6 Froude number as a function of average distance from the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

HEC-RAS computed the tractive force acting on the bottom of the channel along the entire system. Using the tractive force from HEC-RAS, shear velocity was determined along the flume. As shown in figure 2.6, the shear velocity decreases to a minimum prior to passage over the weir, then increases as velocity increases.

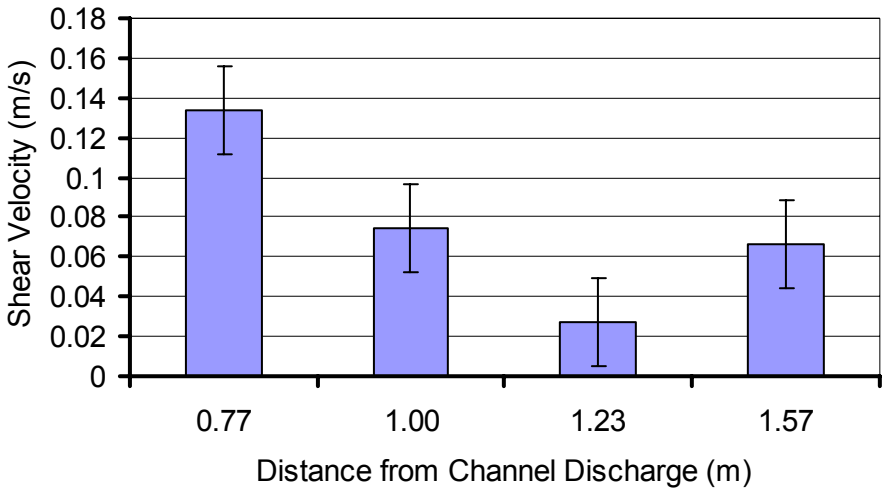


Figure 2.7 Shear velocity as a function of average distance from the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

Biomass Indicators

Figure 2.8 shows the chlorophyll a concentration at each sediment concentration, in each testing region. The first 0 mg/l turbidity run showed a considerably lower chlorophyll a concentration than the second 0 mg/l turbidity run. Also, in the first run, chlorophyll a concentration decreased along the flume length. In the second run, chlorophyll a concentration showed similar results in each testing region.

Three testing cycles were run with 100 mg/l of added turbidity. The first elevated turbidity run was disregarded because the sediment was becoming trapped in the rocks at the beginning of the flume and did not have a chance to affect the testing tiles. The testing runs with

turbidity levels of 100 mg/l were similar in the two runs that were kept. The first 100 mg/l cycle showed the highest chlorophyll a concentration in the upper region and the lowest concentrations in the middle testing region. The second 100 mg/l cycle showed similar concentrations in the upper and middle testing regions and a decrease in chlorophyll a concentration in the lower testing region. Also, the 100 mg/l runs and the second run at 0 mg/l showed similar results in each testing region. The two testing cycles with 200 mg/l were considerably lower than the previous three testing runs. Both 200 mg/l cycles yielded similar results in the middle and lower testing regions. In the upper testing region, the second cycle yielded higher chlorophyll a concentrations than the first cycle. The first run showed an increase in chlorophyll a concentration along the flume length, but the second run showed similar results in the upper and middle regions.

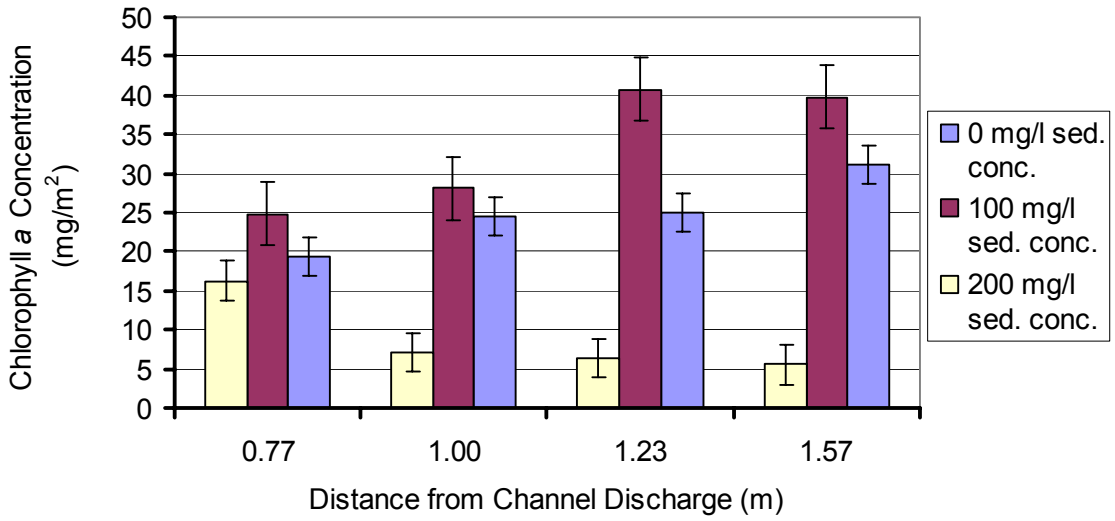


Figure 2.8 Chlorophyll a concentration at each testing region as a function of the distance to the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

ANOVA was used to determine the significance of the variables at the 95% confidence interval. When analyzing the chlorophyll a concentration as a function of reps, sediment

concentration, and flume position it was determined that sediment concentration and flume position was a significant factor. Newman-Keuls Multiple-Comparison tests were run to find if any sediment concentration showed a significant difference in the concentration of chlorophyll a. Chlorophyll a concentration showed a significant ($P \leq 0.05$) difference in each testing region.

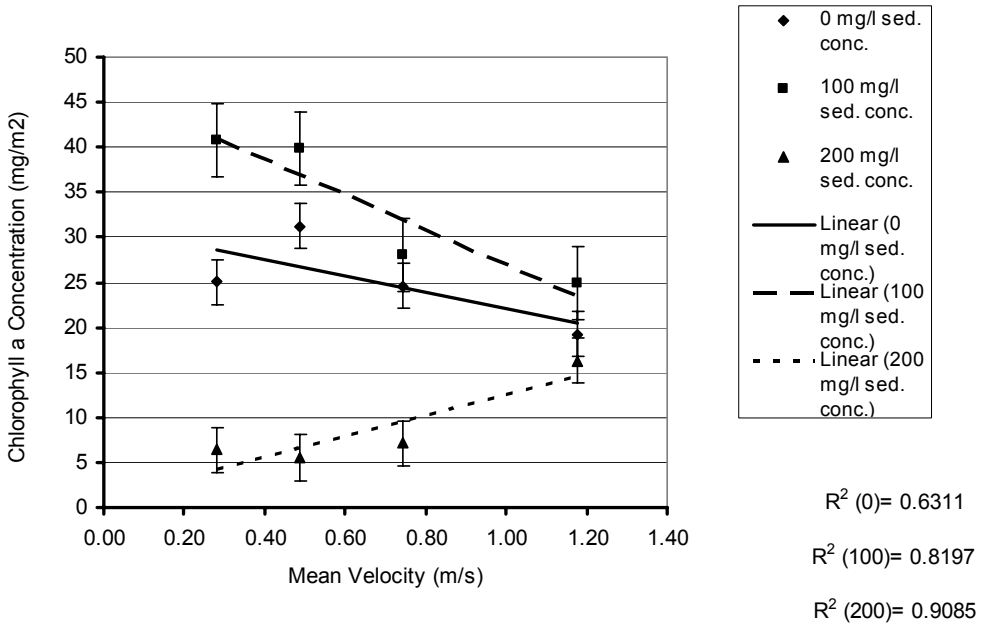


Figure 2.9 Chlorophyll a concentration as a function of mean velocity at each testing region (downslope-1.18 m/s, weir-0.75 m/s, upslope-0.28 m/s, rocks-0.49 m/s) with error bars showing standard error of the means.

Chlorophyll a concentration at each sediment concentration was plotted against mean velocity in figure 2.9. The 0 mg/l ($R^2 = 0.6311$) and 100 mg/l ($R^2 = 0.8197$) sediment concentrations showed a decrease in chlorophyll a concentration with increasing mean velocity. At the 200 mg/l ($R^2 = 0.9085$) sediment concentration there was an increase in chlorophyll a concentration with increasing mean velocity.

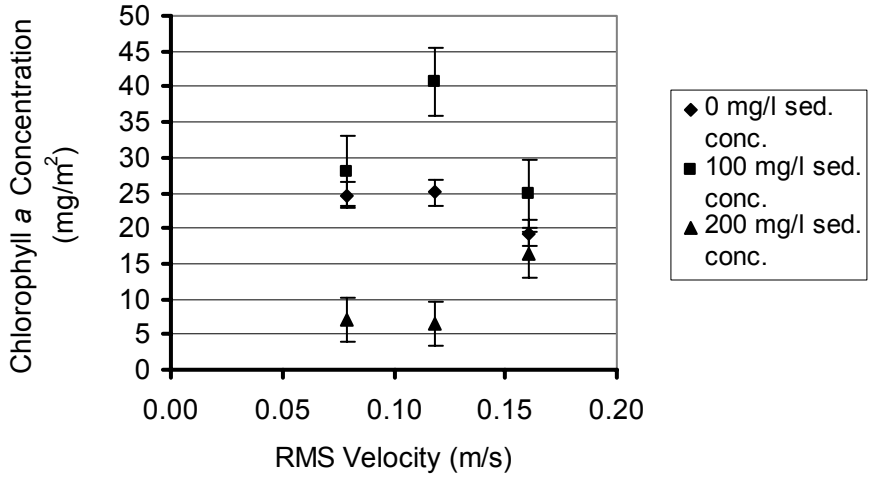


Figure 2.10 Chlorophyll a concentration as a function of RMS velocity at each testing region (downslope-0.16 m/s, weir-0.08 m/s, upslope-0.12 m/s) with error bars showing standard error of the means.

Chlorophyll a concentration vs. RMS velocity graph is shown in figure 2.10. Examining chlorophyll a concentration with increasing RMS velocity shows that in the 0 mg/l and 100 mg/l sediment concentrations there is an increase from the upslope region to the weir and then a decrease to the downslope testing region. In the 200 mg/l sediment concentration there is a decrease in chlorophyll a concentration from the upslope testing region to the weir, and then an increase to the downslope region. Chlorophyll a at each sediment concentration was plotted against shear velocity. R^2 values were low in all sediment concentrations. The results indicate that there is a not a significant relationship with increasing shear velocity at each sediment concentration and therefore it was not shown. Froude number plots were almost identical to the mean velocity plots and therefore were not shown.

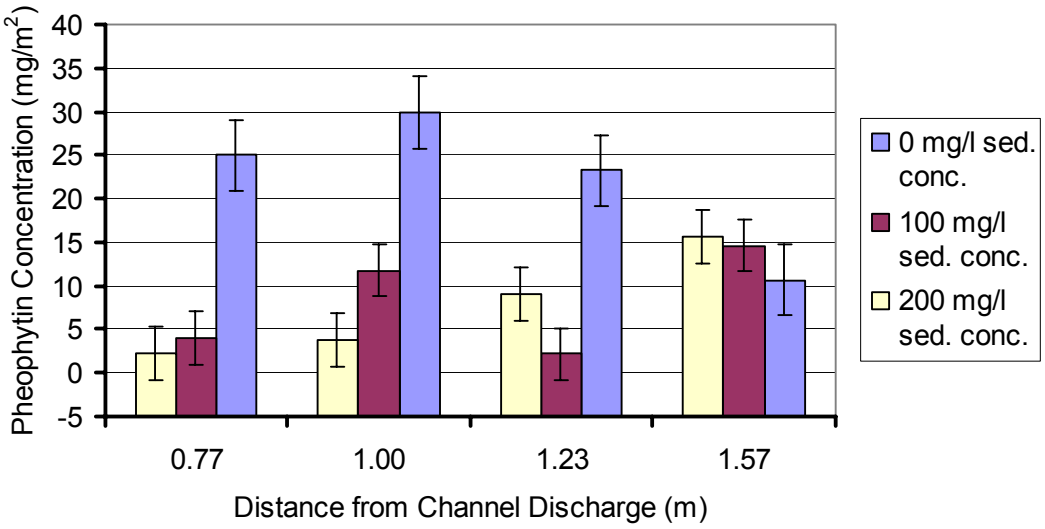


Figure 2.11 Pheophytin concentration at each testing region as a function of the distance to the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m, rocks-1.57 m) with error bars showing standard error of the means.

Figure 2.11 shows pheophytin concentrations at each sediment concentration in each testing region. Pheophytin concentrations decreased with increasing sediment concentration in the rock and upslope testing regions. In the weir region, pheophytin was highest in the 0 mg/l sediment concentration and lowest in the 100 mg/l concentration. In the downslope testing region, pheophytin concentration increased with increasing sediment concentration. In the 0 mg/l sediment concentration, pheophytin concentration increases to a maximum at the upslope testing region and decreases along the rest of the channel. The 100 mg/l sediment concentration shows an increase in pheophytin concentration from the rock section to the upslope region, then a decrease in the weir, followed by an increase to a maximum in the downslope region. In the 200 mg/l sediment concentration pheophytin increases along the flume length.

Pheophytin concentration was analyzed with reps, sediment concentration and flume position and was found to be significant ($P \leq 0.05$). Newman-Keuls Multiple-Comparison tests were run to find if any sediment concentration showed a significant difference in the

concentration of pheophytin. Pheophytin concentration at 0 mg/l was significantly different ($P \leq 0.05$) than the concentration at 100 mg/l and 200 mg/l.

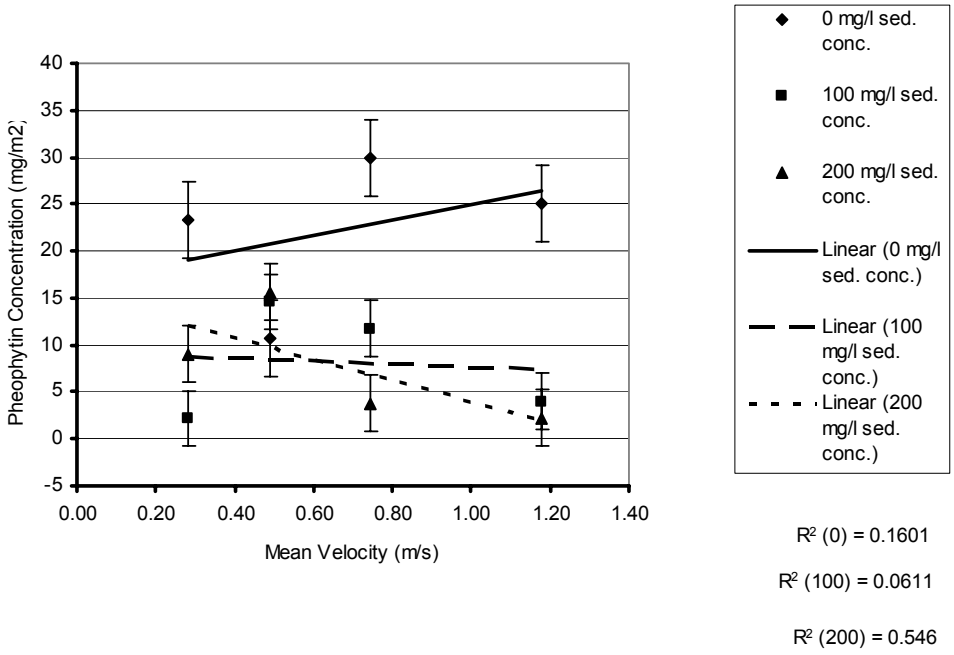


Figure 2.12 Pheophytin concentration as a function of mean velocity at each testing region (downslope-1.18 m/s, weir-0.75 m/s, upslope-0.28 m/s, rocks-0.49 m/s) with error bars showing standard error of the means.

Pheophytin concentration at each sediment concentration was plotted against mean velocity in figure 2.12. Pheophytin concentrations varied at each sediment concentration. At the 0 mg/l ($R^2 = 0.1601$) sediment concentration there was an increase in pheophytin concentration with increasing mean velocity. There was a decrease in pheophytin concentration with increasing mean velocity at 200 mg/l sediment concentration ($R^2 = 0.546$). There was a very low correlation at the 100 mg/l sediment concentration ($R^2 = 0.0611$). All the R^2 values were low in the pheophytin concentrations, so the relationships were not deemed significant.

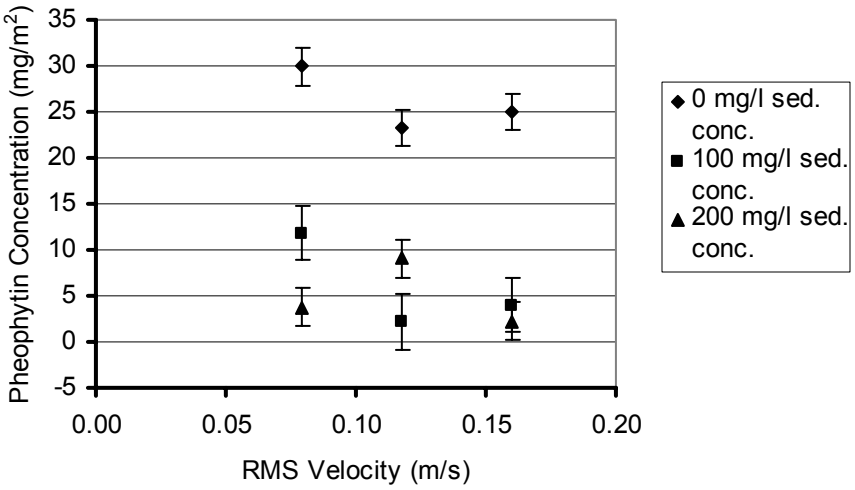


Figure 2.13 Pheophytin concentration as a function of RMS velocity at each testing region (downslope-0.16 m/s, weir-0.08 m/s, upslope-0.12 m/s) with error bars showing standard error of the means.

Pheophytin concentration as a function of RMS velocity is shown in figure 2.13. Examining pheophytin concentration shows a reverse pattern than chlorophyll a concentration in all sediment concentrations. In the 0 mg/l and 100 mg/l sediment concentrations there is a decrease from the upslope region to the weir and then an increase to the downslope testing region. In the 200 mg/l sediment concentration there is an increase in pheophytin concentration from the upslope testing region to the weir, then a decrease to the downslope region. Pheophytin concentration at each sediment concentration was plotted against shear velocity. R^2 values were low in all sediment concentrations. Pheophytin concentration did not have a high correlation with increasing shear velocity at each sediment concentration and therefore it is not shown.

Filament Length

Filament length was obtained by taking a random sample of the filament lengths at six points in each testing region and averaging them to obtain a mean. Figure 2.14 shows filament length at each sediment concentration at each testing region. As sediment concentration

increased, mean filament length increased. Filament length decreased from the upslope region to the weir to the downslope region, except in the upslope testing region at 200 mg/l sediment concentration. The upslope region at the 200 mg/l sediment concentration had the lowest length in the entire flume. This is due to the high turbidity in this section reducing the light availability, therefore reducing periphyton filament length.

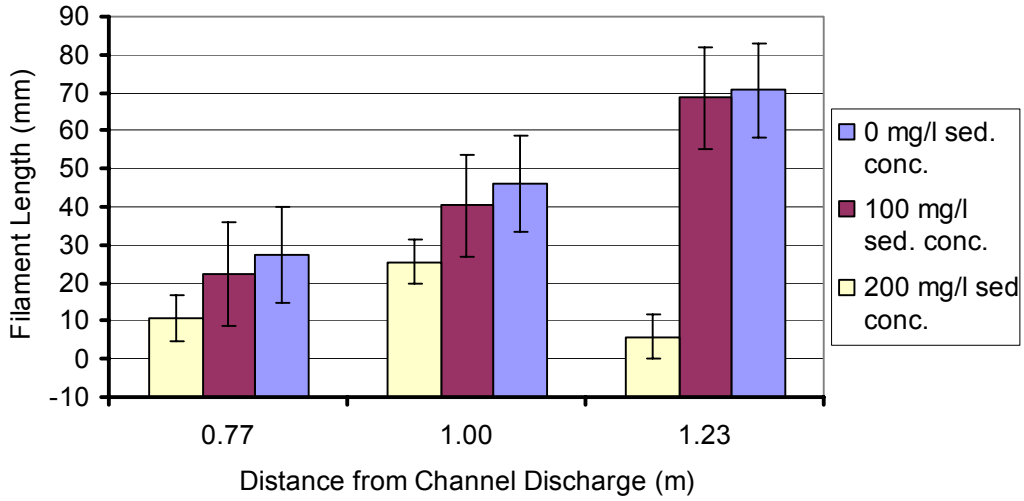


Figure 2.14 Filament length at each testing region as a function of the distance to the channel discharge (downslope-0.77 m, weir-1.00 m, upslope-1.23 m) with error bars showing standard error of the means.

The filament length testing showed that the influence of turbidity did have an effect on the periphyton. Filament length was analyzed using 1-way ANOVA to determine the interactions between sediment concentration and flume position. It was determined that both sediment concentration and flume position were significant at the 95% confidence interval. Using the Newman-Keuls Multiple-Comparison Test it is determined that the filament length at the sediment concentration of 200 mg/l was significantly ($P \leq 0.05$) shorter than the filament lengths at the sediment concentrations 100 mg/l and 0 mg/l. There was no significant difference ($P \leq 0.05$) between the filament lengths at sediment concentrations 0 mg/l and 100 mg/l. The Newman-Keuls Multiple-Comparison Test also showed a significant difference ($P \leq 0.05$)

between the filament lengths in the lower region and the filament lengths in the middle and upper regions. One-way ANOVA was then run while omitting the 200 mg/l sediment concentration, as inclusion of the 200 mg/l data resulted in a significant ($P \leq 0.05$) interaction between position and sediment concentration. Sediment concentration was not significant ($P \leq 0.05$) while flume position remained significant ($P \leq 0.05$). The Newman-Keuls Multiple-Comparison Test changed and showed that there was no significant difference ($P \leq 0.05$) between the three positions. This leads us to believe that only sediment concentration over 200 mg/l adversely affected the periphyton filament length. One-way ANOVA omitting the 200 mg/l sediment concentration was run with variables of sediment concentration and mean velocity to determine significance ($P \leq 0.05$). This process was also run with RMS velocity mean velocity as a variable. RMS velocity was found to be significant ($P \leq 0.05$) while mean velocity was not significant ($P \leq 0.05$). Newman-Keuls Multiple-Comparison Tests indicated that all testing regions were significantly different ($P \leq 0.05$) from each other with mean velocity and RMS velocity. The Newman-Keuls Multiple-Comparison Test was run to determine if there was a significant difference ($P \leq 0.05$) in biomass above, upslope and downslope of the weir at 200 mg/l. It showed that there was a significant difference ($P \leq 0.05$) between the middle testing region and the lower and upper testing regions.

Two-way ANOVA was then run with the added covariate of RMS velocity to see if significance ($P \leq 0.05$) was affected. Position, sediment concentration, and the interaction between them were all significant at 95%. Under these conditions, Newman-Keuls Multiple-Comparison tests showed the sediment concentration 200 mg/l was significantly different ($P \leq 0.05$) from the 0 and 100 mg/l concentrations. Two-way ANOVA was then run with mean velocity as a covariate. Under this condition flume position was not significant at 95%. There

was no difference (P≤0.05) in the Newman-Keuls Multiple-Comparison tests under this new condition.

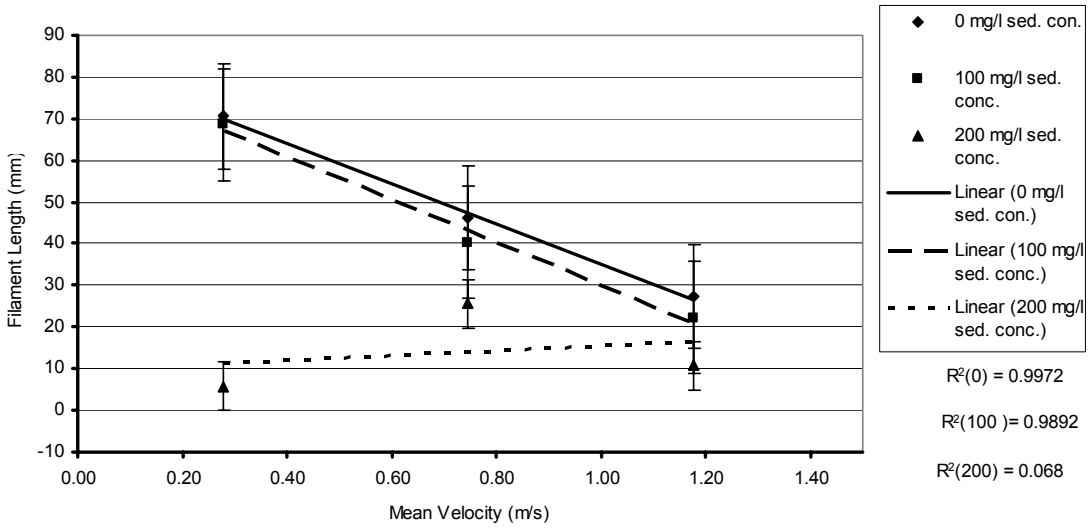


Figure 2.15 Filament length as a function of mean velocity at each testing region (upslope-0.28, weir-0.75, downslope-1.18) with error bars showing standard error of the means.

Figures 2.15 shows graphical results of filament length versus mean velocity. Filament length showed a decrease with increasing mean velocity at the 0mg/l and 100 mg/l sediment concentrations. Filament length showed high R^2 values at the 0 mg/l ($R^2 = 0.9972$) and 100 mg/l ($R^2 = 0.9892$) sediment concentration. Due to the high R^2 values a significant relationship between filament length and mean velocity can be assumed. At the 200 mg/l ($R^2 = 0.068$) sediment concentration showed no correlation between filament length and mean velocity.

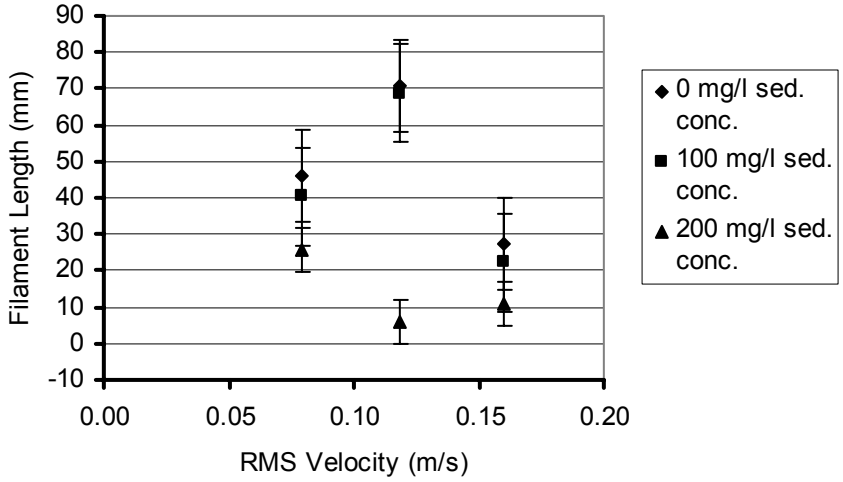


Figure 2.16 Filament length as a function of RMS velocity at each testing region (downslope-0.16 m/s, weir-0.08 m/s, upslope-0.12 m/s) with error bars showing standard error of the means.

Filament length was examined against RMS velocity in figure 2.16. Filament length shows the same patterns as chlorophyll a concentration in all sediment concentrations. Filament length showed a decrease with increasing shear velocity at the 0mg/l and 100 mg/l sediment concentrations. These results are similar to the plot of filament length vs. mean velocity and was excluded. From the results it can be determined that shear velocity plays a role in periphyton morphology but has no effect on the periphyton biomass. Examining Froude number to determine its relationship with filament length resulted in plots that were almost identical to the plots versus mean velocity and therefore were excluded.

The increase in water velocity also caused the filaments to take different shapes. In the upper testing area, the filaments stood up, while in the lower testing region the water velocity forced the periphyton to lay flat against the tiles. When the added turbidity was introduced, the way the periphyton grew changed. In the upper testing region the presence of sediment concentration 200 mg/l caused the periphyton to grow in thin mats that were hard to accurately quantify their length. In the middle testing region the periphyton filaments trapped a visible

amount of sediment in them, so the filaments could be seen with thick amounts of sediment trapped near the bottom of the group of filaments. The majority of the lower testing region did not show any visible differences between each cycle except for a longer initial growth period. In the higher turbidity runs, the tiles that were next to the elevated testing sites had a different growth pattern then the rest of the lower testing tiles. The tiles next to the elevated testing site showed a different filament texture then the rest of the lower testing site. The differences in periphyton filament lengths in each testing region resulted from a variety of factors. In the upper testing region, the filaments grew long due to the slow water velocity and high water depth. As sediment concentration increased, the filament length decreased the largest amount due to a combination of decreased light penetration, due to turbidity, and an increase in sedimentation. The middle testing region initially had a smaller filament length than in the upper testing region due to lower water depth and increased shear velocity due to increased water velocity. In the middle testing region the filament length decreased as sediment concentration increased due to the amount of sediment that became trapped in the growing filament and increased sediment scouring. The lower testing region had the lowest initial periphyton filament length due to having the highest velocity and lowest depth in the entire flume. The filament lengths in this region showed the smallest decrease being only affected by the increased scouring rate.

In Hondzo and Wang (2002), periphyton biomass was minimal in stagnant flow conditions and increased to a maximum at a shear velocity of 0.7m/s. Pheophytin, not studied by Hondzo and Wang (2002), showed no significant correlation in the 0 mg/l sediment concentration. The highest shear velocities occurred in the downslope testing region and the lowest velocities occurred in the upslope testing region. Periphyton biomass decreased (R^2 = 0.4971) with an increase in shear velocity at the 0 mg/l sediment concentration. Periphyton

biomass was lower than in Hondzo and Wang (2002), likely due to low light levels in this study. Filament length showed a decrease in length as shear velocity increased in the 0 and 100 mg/l sediment concentrations. In the 200 mg/l sediment concentration, the filament length increased then decreased as shear velocity increased. This shows that the 200 mg/l sediment concentration more dramatically effects filament length than the shear velocity. The added turbidity inhibited the filament growth in the upslope region which had the least amount of shear velocity.

With no additional turbidity the periphyton grew in filaments that required significant effort to remove filaments from the tiles. As the turbidity was introduced, the periphyton changed to grow in thin mats that flaked off easily. The periphyton in the upper testing region flaked off easier then the tiles from the other testing regions. The increasing pheophytin with turbidity would explain the lower degree of attachment to the tile. Unfortunately these changes lead to the losing of some periphyton during transport to the laboratory. Visually one could see a difference in the periphyton growth in each testing region and in each testing cycle. The periphyton in the upper testing region with low velocity grew in thicker quantities, while the lower testing region grew more sparsely at first. In the middle testing region it was a combination of the two.

Summary Discussion

The PAR study suggested that the amount of light reaching the periphyton in the upslope section was 31.97 μmol/m^2s and decreased to 21.375 μmol/m^2s at 100 mg/l sediment concentration and to 13.43 μmol/m^2s at 200 mg/l sediment concentration. These light intensities are approximately equal to the amount of light reaching streams due to shading effects from terrestrial vegetation (Hill, 1996).

Ammonia and pH stayed at a constant level over the entire experiment. All other nutrient readings increased along with an increase in sediment concentration. This was not entirely unexpected because sediment has a tendency to bind to nitrogen and phosphorus and ions while in streams. Two-sided T-tests showed no significant difference ($P \leq 0.05$) in the water quality data.

The results of this experiment indicate that there is a significant relationship between periphyton biomass, filament length and the flow parameters selected in this study. This study can indicate parameters to predict and control periphyton biomass can be used to maximize industrial usefulness and ecological feasibility. Chlorophyll a concentration and filament length both showed a decrease with increasing mean velocity. Pheophytin concentration showed a better correlation with RMS velocity. These results indicate that more testing needs to be done to better determine the relationship between periphyton biomass indicators, filament length and these selected flow parameters. Hydraulic parameters were not affected by periphyton biomass. The periphyton biomass was not great enough to move the Manning's coefficient of roughness beyond published ranges for the respective flume materials, based on some computations with roughness heights expected from periphyton aggregations following methods found in Chow (1959). This is consistent with the constant depth throughout the testing run observed in the course of each run.

Ideally, there would have been more flowrates and velocity conditions evaluated. Turbidity proved challenging and required significant time to handle such that turbidity was predictable through the course of each run. Future work should consider additional flowrates to further validate these results. Prediction of pheophytin with RMS velocity needs further study.

Industrial practices, such as waste treatment systems and stormwater management systems, may tailor discharge rates to accomplish healthy periphyton biomass, which would be beneficial in nutrient retention and depletion. Aquaculture practices can use this information to develop better farming and feeding practices to improve water quality as well as provide a base food source.

CONCLUSIONS

The flume system comprising a rock media and tiles arranged to form a broadcrested weir enabled a variety of flow regimes that can be easily controlled. The flow provided three distinct testing regions. Configuring a laboratory flume to hydraulically mimic a natural channel will enable additional measurements not generally possible in the field. HEC-RAS provides a reasonably good basis for computing flow regimes in natural channels based on the velocity simulations.

Mean velocity appears to be a good indicator of periphyton biomass and filament length at the turbidity levels evaluated. RMS velocity provided the best correlations with pheophytin at the various turbulence levels. Additional studies are needed to further evaluate these results. The addition of 100 mg/l sediment concentration caused an increase in chlorophyll *a* concentration due to a higher chlorophyll *a*/cell concentration in low light treatments (Richardson et al., 1983) caused by shading due to higher turbidity. The 200 mg/l sediment concentration showed a large decrease in chlorophyll *a* concentration from the two previous sediment concentrations, leading us to believe that at this concentration turbidity is high enough to overcome periphyton response to light deficiency. An increase in sediment concentration caused a decrease in filament length. These studies provided insights needed into the design of sustainable fish production as well as waste and water management systems.

ACKNOWLEDGEMENTS

Many people have helped with the progress in this project. I would like to thank Dr. David Gattie and Dr. Amy Rosemond for sharing their knowledge and experience. I would like to thank Dr. Alan Covich for the use of his incubator. I would like to thank Ryan Adolfson for help with setting up the flume and tank system. I would like to thank Antonio Travassos for the help with chlorophyll a concentration testing. Also, I would like to thank Javier Sayago for help with water quality testing and the use of his laboratory. Finally, I would like to thank Dr. William Kisaalita and Dr. Mark Eiteman and the members of their laboratories for all the help they have provided.

REFERENCES

Adey, W., Luckett, C. and K. Jensen. 1993. Phosphorus removal from natural waters using controlled algal production. *Restor. Ecol*. 1:29-39

Asaeda, T. and D. Hong Son. 2000. Spatial structure and populations of a periphyton community: a model and verification. *Ecological Modelling* 133(3): 195-207

Bergstedt, M.S., Hondzo, M.M., and J.B. Cotner. 2004. Effects of small scale fluid motion on bacterial growth and respiration. *Freshwater Biology* 49:28-40.

Bockelmann, B.N., E.K. Fenrich, B. Lin, and R.A. Falconer. 2004. Development of an ecohydraulics model for stream and river restoration. *Ecological Engineering* 22:227-235.

Bokn, T.L., F.E. Moy, H. Christie, R. Karez, K. Kersting, P. Kraufvelin, C. Lindblad, N. Marba, M.F. Pedersen and K. Sorensen. 2002. Are rocky shore ecosystems affected by nutrient enriched seawater? Some preliminary results from a mesocosm experiment. *Hydrobiologia* 484: 167-175.

Boynton, W.R., D. Baird, and R.E. Ulanowicz. 1995. Seasonal nitrogen dynamics in Chesapeake Bay: a network approach. *Estuarine, Coastal, and Shelf Science* 41(2): 137-162.

Chow, V.T. 1959, Open-Channel Hydraulics. McGraw-Hill, Inc. New York, NY.

Dodds, W.K. 2003. The role of periphyton in phosphorus retention in shallow freshwater aquatic systems. *J. Phycol.* 39:840-849.

Eriksson, P.G. and S.E.B. Weisner. 1996. Functional differences in epiphytic microbial communities in nutrient-rich freshwater ecosystems: an assay of denitrifying capacity. *Freshwater Biology* 36:555-562.

Graneli, W., and D. Solander. 1988. Influence of aquatic macrophytes on phosphorus cycling in lakes. *Hydrobiologia* 170:245-266.

Guillard, R.R. and C.J. Lorenzen. 1972. Yellow-green algae with Chlorophyllide $C^{1,2}$. *J. Phycol.* 8:10-14.

Harding, J.S., R.G. Young, J.W. Hayes, K.A. Shearer, and J.D. Stark. 1999. Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology* 42:345-357.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Reviews in Fisheries Science* 8(2):125-139.

Hill, B.H., A.T. Herlihy, P.R. Kaufmann, S.J. DeCelles and M.A. Vander Borgh. 2003. Assessment of streams of the eastern United States using a periphyton index of biotic integrity. *Ecological Indicators* 2:325-338.

Hill, W.R. 1996. Factors affecting benthic algae *In* Stevenson, R.J., M. L. Bothwell, and R.L. Lowe [Eds.] Algal Ecology: Freshwater Benthic Ecosystems. Academic Press, San Diego, CA, pp. 341-374.

Hintze, J.L. 2001. Number Cruncher Statistical System 2004 version. Kaysville, Utah.

Hondzo, M. and H. Wang. 2002. Effects of turbulence on growth and metabolism of periphyton in a laboratory flume. *Water Resources Research* 38(12):1277.

Lau, Y.L., and D. Liu. 1993. Effect of flow rate on biofilm accumulation in open channels. *Water Research*. 27(3): 355-360

Mattila, J. and R. Raisanen. 1998. Periphyton growth as an indicator of eutrophication; an experimental approach. *Hydrobiol*. 377:15-24.

Nezu, I., Kadota, A., and H. Nakagawa. 1997. Turbulent structure in unsteady depth-varying open-channel flows. *Journal of Hydraulic Engineering* 123(9):752-763.

Nikora, V.I., Goring, D.G., and B.J.F. Biggs. 1997. On stream periphyton-turbulence interactions. *New Zealand Journal of Marine and Freshwater Research* 31:435-448.

Nikora, V.I., Goring, D.G. and B.J.F. Biggs. 2002. Some observations of the effects of micro-organisms growing on the bed of an open channel on the turbulence properties. *J. Fluid Mech.* 450:317-341.

Parkhill, K.L., and J.S. Gulliver. 2002. Effect of inorganic sediment on whole-stream productivity. *Hydrobiologia* 472:5-17.

Paulsson, M., Mansson, V., and H. Blanck. 2002. Effects of zinc on the phosphorus availability to periphyton communities from the river Göta Älv**.** *Aquatic Toxicology* 56(2): 103-113.

Quinn, J.M., Hickey, C.W., and W. Linklater. 1996. Hydraulic influences on periphyton and benthic macroinvertebrates: simulating the effects of upstream bed roughness. *Freshwater Biology* 35:301-309.

Riegman, R., J.D.L. Van Bleuswuk, and C.P.D. Brussard. 2002. The use of dissolved esterase activity as a tracer of phytoplankton lysis. *Limnol. Oceanogr*. 47: 916-920.

Richardson, K., J. Beardall, and J.A. Raven. 1983. Adaptation of unicellular algae to irradiance: an analysis of strategies. *New Phytol.* 93: 157-191.

Sabater, S., J. Armengol, E. Comas, F. Sabater, I. Urrizalqui, I. Urrutia. 2000. Algal biomass in a disturbed Atlantic River: water quality relationships and environmental implications. *The Science of the Total Environment* 263:185-195.

Sansone, U., M. Belli, M. Riccardi, A. Alonzi, Z. Jeran, J. Radojko, B. Smodis, M. Montanari, and F. Cavolo. 1998. Adhesion of water-borne particulates on freshwater biota. *The Science of the Total Environment* 219:21-28.

Saravia, L.A., F. Momo, and L.D. Boffi Lissin. 1998. Modelling periphyton dynamics in running water. *Ecological Modelling* 114:35-47

Schofield, K.A., C.M. Pringle, and J.L. Meyer. Effects of increased bedload on algal and detrital-based stream food webs: experimental manipulation of sediment and macroconsumers. *Limnology and Oceanography* 49(4): 900-909

Article in Press Stefes, B., and H.H. Fernholz. 2003. Skin friction and turbulence measurements in a boundary layer with zero-pressure-gradient under the influence of high intensity free-stream turbulence. *European Journal of Mechanics B/Fluids*

Steinman, A.D. and Lamberti, G.A. 1996. Biomass and pigments of benthic algae. *In* Hauer, F. R. & Lamberti, G.A. [Eds.] Methods in Stream Ecology. Academic Press, San Diego, CA pp. 295-309

Stevenson, R.J. 1996. The stimulation and drag of current. *In* Stevenson, R.J., M. L. Bothwell, and R.L. Lowe [Eds.] Algal Ecology: Freshwater Benthic Ecosystems. Academic Press, San Diego, CA, pp. 341-374.

U.S. Environmental Protection Agency. 2001 Chattooga River Watershed Ecological/Sedimentation Project. Federal Interagency Sedimentation Conference.

Vis C., C. Hudon, A. Cattaneo, B. Pinel-Alloul. 1998. Periphyton as an indicator of water quality in the St Lawrence River (Quebec, Canada). *Environmental Pollution* 101:13-24.

CONCLUSIONS AND FUTURE NEEDS

Periphyton biomass is influenced by nutrient abundance, water velocity, irradiance, water turbulence, turbidity and sedimentation. Though the nutrient factors have been extensively researched, the physical factors have not been as thoroughly studied. Selected physical factor interactions are examined in this paper. By designing a laboratory system to test flow regime and turbidity we enabled isolation of these factors in a way that is difficult to achieve in the natural environment. Designing the flume to simultaneously achieve different areas of specific flow regime, then by adding sediment concentrations we are able to look at the effects of the interactions between these factors. Flows requiring rigorous similitude in velocity profiles may not be possible in the system due to the difficulty of establishing true boundary layer development in a laboratory flume. HEC-RAS was used to determine the interactions between turbidity and flow factors and if the factors can be scaled to natural stream scale systems. It was also determined that HEC-RAS could be used to scale the model channel to mimic a natural channel. We hypothesized that we could develop a controlled system to isolate flow regime/sediment interactions. Our objectives were accomplished showing distinct areas of flow regime. Ghosh and Gaur (1998) stated that periphyton accumulation and current velocity have an inverse relationship. The results of this paper support these findings. In the areas of high velocity the periphyton biomass was lower than in the area of low water velocity. Periphyton filament length was also influenced by the shear velocity. An increase in sediment concentration has an effect of periphyton biomass and filament length. Chlorophyll a concentration was highest in the 100 mg/l sediment concentration, due to the adaptation of higher per cell

concentrations at lower light intensity (Richardson et al., 1983). At 200 mg/l the biomass and filament length drastically changed from earlier sediment concentrations. This leads us to believe that to promote good periphyton assemblages sediment concentrations must be kept under 200 mg/l. In comparing our results to the Hondzo and Wang (2002) periphyton biomass vs. shear velocity curve, we observe a similar decreasing trendline. Our results show the same decreasing pattern of periphyton biomass with increasing shear velocity. In summary, this work was confirmatory of Hondzo and Wang (2002) for the 0 turbidity case and extended their work by adding a turbidity dimension and documenting pheophytin response. In future research, Hondzo and Wang (2002) could be combined with our study to create a study using their experimental setup and determining the influence of increased sediment in that system.

In comparing our results to the Hondzo and Wang (2002) periphyton biomass vs. shear velocity curve, only the 100 mg/l sediment concentration results showed a significant decreasing pattern of chlorophyll a concentration with increasing shear velocity. Using a larger flume would help to generate more realistic boundary conditions and provide for more space for sampling. Flow parameters designed to maximize periphyton biomass can be used in wastewater treatment systems, stormwater management systems, and aquacultural systems, to improve nutrient retention, removal of biodegradable material, and improve ecosystem sustainability.

Laboratory flume yielded results that indicated that turbidity, and velocity were all significant in their effects on periphyton biomass and filament length. By accurately modeling the flume using HEC-RAS we can scale the results to those of a natural river system or a design treatment system. Using these results we can hope to achieve data that will help to prevent the degradation of river ecosystems and in the design of ecologically sound treatment systems. Ideally, there would have been more flow rates and velocity conditions evaluated. Turbidity

proved challenging and required significant time to handle such that turbidity was predictable through the course of each run. Future work should consider additional flowrates to further validate these results. In the future, components can be added to this experiment to further develop the interactions between turbidity, flow factors, and potentially nutrients.

This experiment can also be modified to study more interactions involving specific nutrients, sediment sizes, and light penetration levels. Pringle and Hamazaki (1997) examined the effects of fishes on algal response to storms in a tropical stream. The results of our project could add a dimension to their study by giving a way to model the change in stream flow regime caused by storms. The knowledge of flow characteristics can provide added knowledge about tractive force and turbulence caused by the added flow rate. Vis et al. (1998) tried to determine if periphyton was an indicator of water quality in a Canadian river. Though they examined the nutrient additions due to wastewater additions, they did not take into account that there might be an impact of added sediment carrying phosphorus, metals, and ions into the stream. Though they examined the seasonal changes in the stream it can be further explored to include tractive force as a variable in future studies.

REFERENCES

Ghosh, M. and J.P. Gaur. 1998. Current velocity and the establishment of stream algal periphyton communities. *Aquatic Botany* 60:1-10.

Hondzo, M. and H. Wang. 2002. Effects of turbulence on growth and metabolism of periphyton in a laboratory flume. *Water Resources Research* 38(12):1277.

Pringle, C.M. and T. Hamazaki. 1997. Effects of fishes on algal response to storms in a tropical stream. *Ecology* 78(8):2432-2442.

Richardson, K., J. Beardall, and J.A. Raven. 1983. Adaptation of unicellular algae to irradiance: an analysis of strategies. *New Phytol.* 93: 157-191.

Vis C., C. Hudon, A. Cattaneo, B. Pinel-Alloul. 1998. Periphyton as an indicator of water quality in the St Lawrence River (Quebec, Canada). *Environmental Pollution* 101:13-24.

APPENDIX A:

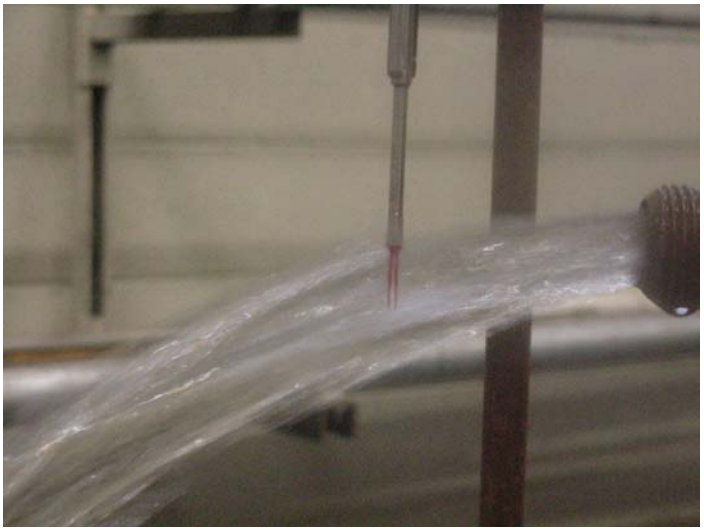
ANEMOMETER CALIBRATION

APPENDIX A-1: CALIBRATION CHARTS AND PICTURES

The anemometer was calibrated using a hose with a known velocity and a common water faucet. The faucet was set at four discharge levels: off, full flow and two intermediate levels. The water was turned on and the anemometer discharge rate was measured. The picture of the calibration setup is given below.

Anemometer Calibration Chart

	A, m^2	Q, m^3/s	V, m/s
1	0.003785	0	0
2	0.003785	0.000199	0.052632
3	0.003785	0.000379	0.1
4	0.003785	0.000473	0.125

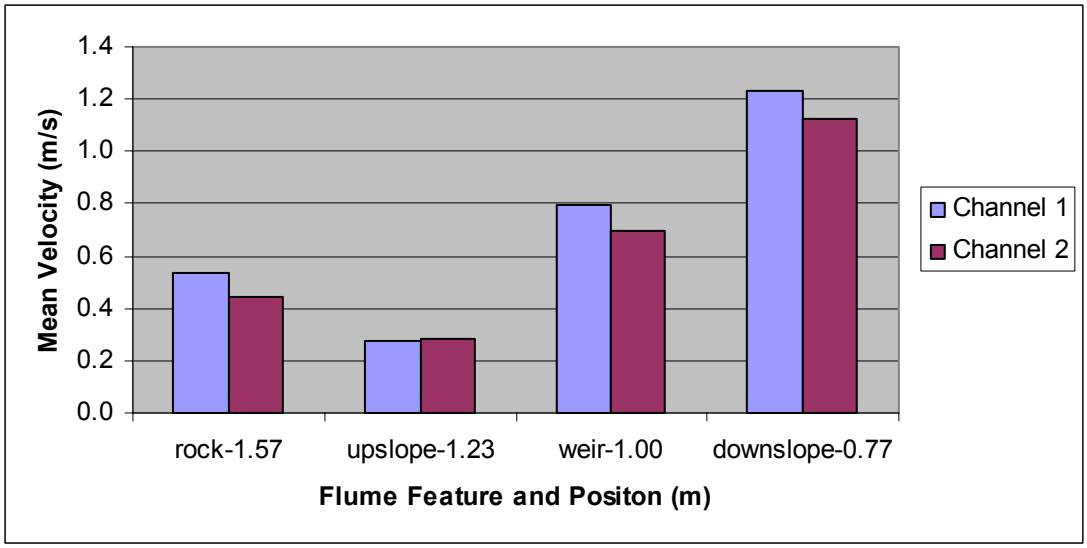


APPENDIX B:

ANEMOMETER CHANNEL COMPARISON

APPENDIX B-1: ANEMOMETER FLOW REGIME

Mean Velocity (m/s)



APPENDIX C:

DEPTH COMPARISON

APPENDIX C-1: DEPTH COMPARISON

Position from channel discharge (m)	Hook Gauge Mean Depth (m)	Percent Difference[a]			
		normal n	high n	low n	selected n
0.77	0.034	18	18	41	18
1.00	0.068	36	36	36	36
1.23	0.102	-8	-8	-8	-8
1.57	0.093	20	18	20	20

[a] Percent difference = [(Anemometer mean depth-HEC-RAS mean depth)/Anemometer mean depth]*100

Depth comparisons between the hook gauge mean depth and the HEC-RAS mean depth at different Manning's n values.

APPENDIX D:

LIGHT INTENSITY

APPENDIX D-1: LIGHT INTENSITY TESTS

A LI-COR quantum/radiometer/photometer model Li 189[4] was used to measure the light intensity at the maximum water depth of 10.496 cm. The LI-COR meter was not submergible, thus the experiment was replicated using a clear glass beaker. The beaker was filled with the 101.14 mm of water and the LI-COR meter was placed under the beaker. The growth lights were turned on and the light intensity was measured at the three sediment concentrations. The light intensity was significantly reduced as the sediment concentration increased. The table below states the results in μmol and gives the light intensity of a clear sunny day.

Light Intensity at the upper testing region depth of 4.1 in at each sediment concentration

Sed. Conc. (mg/l)	Light Int. (μmol)
Daylight	1610.3
0	31.97
100	21.375
200	13.43

[4] Electronic System Ecosystem 2229 Fifth St. Berkeley, CA 94710

APPENDIX D-2: LIGHT INTENSITY TESTING PICTURE

The light intensity testing setup

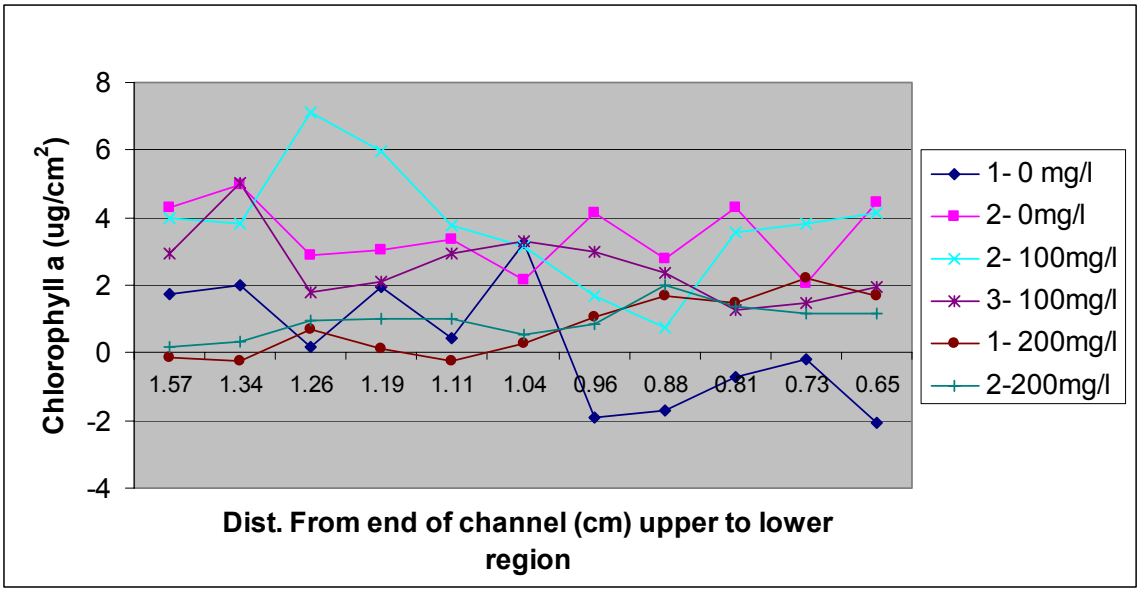
APPENDIX E:

PERIPHYTON BIOMASS DATA

APPENDIX E-1: CHLOROPHYLL a DATA

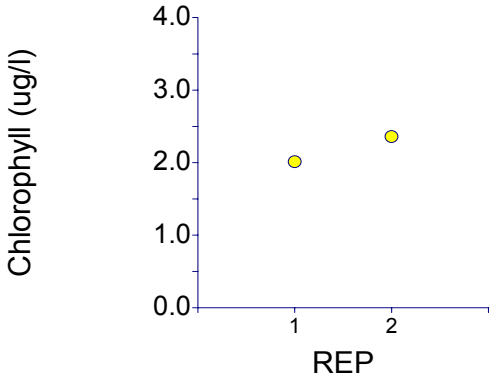
Chlorophyll a (µg/cm^2) shown as an average value of both channels in each run

RUN	1	2	3	4	5	6	7
R	1.7568	4.2703	2.4829	3.9801	2.9193	-0.1292	0.1598
U1	1.9822	4.9574	0.4843	3.8449	5.0363	-0.2565	0.3472
U2	0.1496	2.9085	1.0565	7.0887	1.7973	0.6952	0.9496
U3	1.9706	3.0401	0.2510	5.9905	2.1072	0.1434	1.0227
U4	0.4544	3.3437	-2.5413	3.7543	2.9524	-0.2313	1.0096
M1	3.2692	2.1646	3.8621	3.1263	3.2834	0.2900	0.5470
M2	-1.8891	4.1608	1.7485	1.6835	2.9806	1.0739	0.8577
L1	-1.6824	2.8072	2.6727	0.7653	2.3669	1.7097	1.9853
L2	-0.7262	4.2958	-0.5236	3.5446	1.2868	1.5000	1.3585
L3	-0.1935	2.0626	5.0873	3.8443	1.4693	2.2278	1.1523
L4	-2.0674	4.4513	1.9318	4.1343	1.9588	1.6899	1.1802

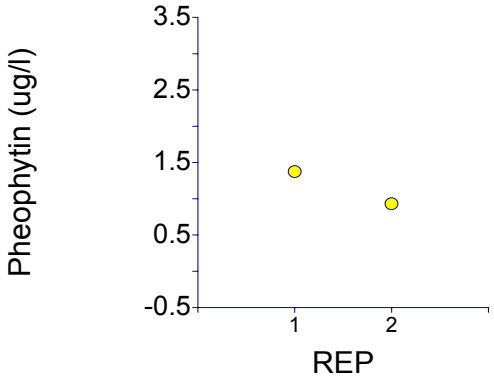


Chlorophyll a concentration along the flume reach for each testing cycle, µg/cm^2

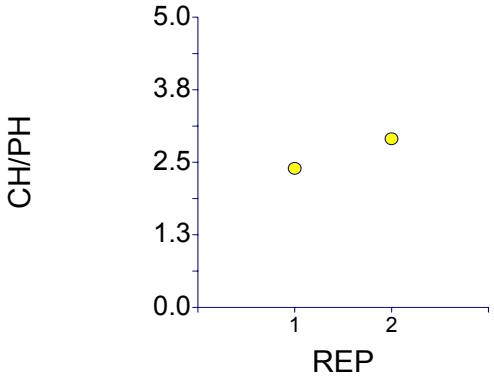
APPENDIX E-2: CHANNEL PERIPHYTON BIOMASS COMPARISONS



Mean chlorophyll a concentration for total flume system in channels 1 and 2



Mean pheophytin concentration for total flume system in channels 1 and 2



Ratio of the total flume chlorophyll a concentration to total flume pheophytin concentration in each channel

APPENDIX E-4: WATER QUALITY DATA

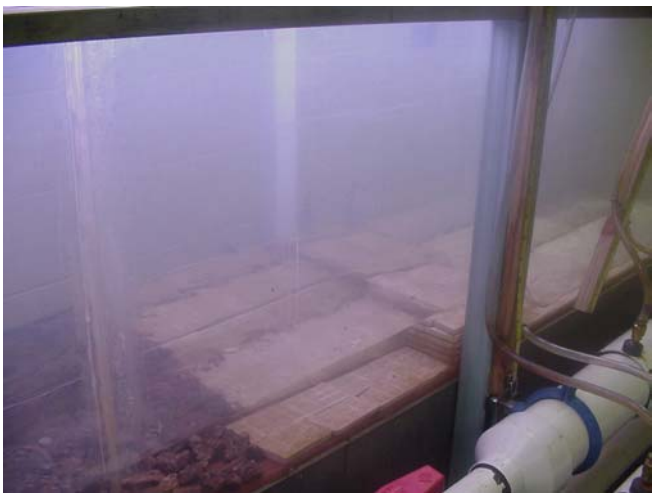
Nutrient analysis for flume water at each sediment concentration (put this in the appendix of the thesis

Nutrients	0 mg/l	100 mg/l	200 mg/l
pH	7.54	7.73	7.62
Conductivity µseimen/cm	445	566	647
BOD	5.21	3.8	6.17
Total N, ppm	29.344	35.874	42.834
Total P, ppm	2.379	2.54	2.97
NH_3 ,ppm	<0.09	<0.09	<0.09
NO_2/NO_3 ,ppm	25.342	33.625	40.777
OrthoPO_4 ,ppm	1.786	2.138	2.563

APPENDIX F:

PICTURES OF LABORATORY FLUME AND BIOMASS TESTING

APPENDIX F-1: SEDIMENT CONCENTRATION INITIAL SETUP

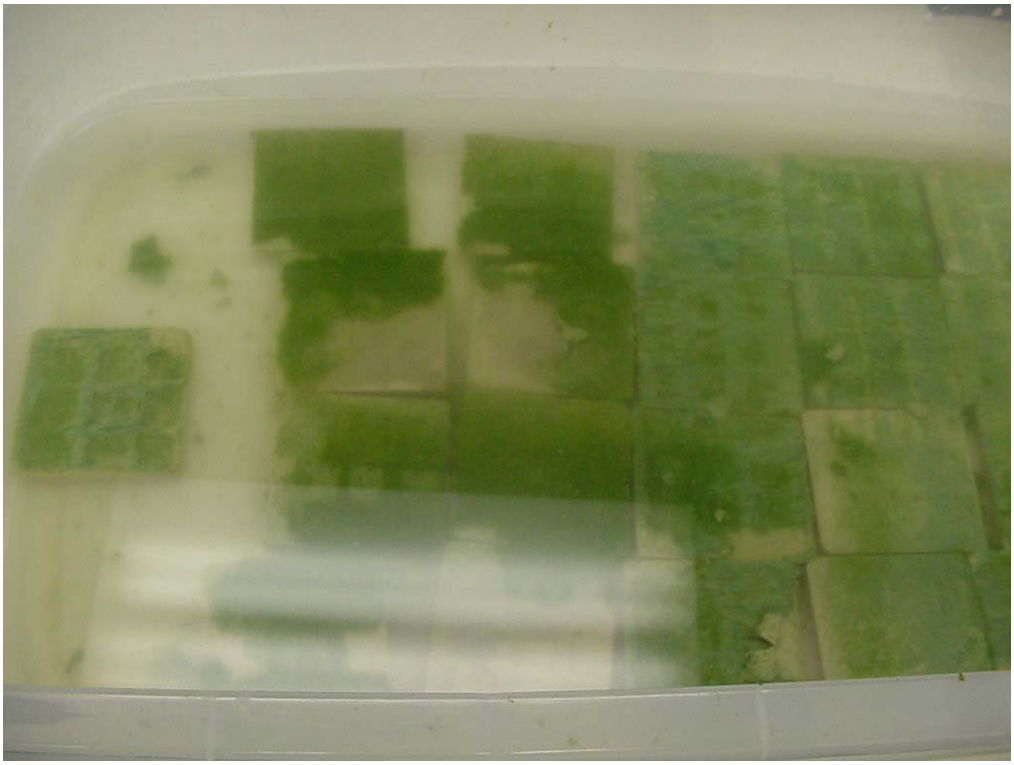


0 mg/l day 1

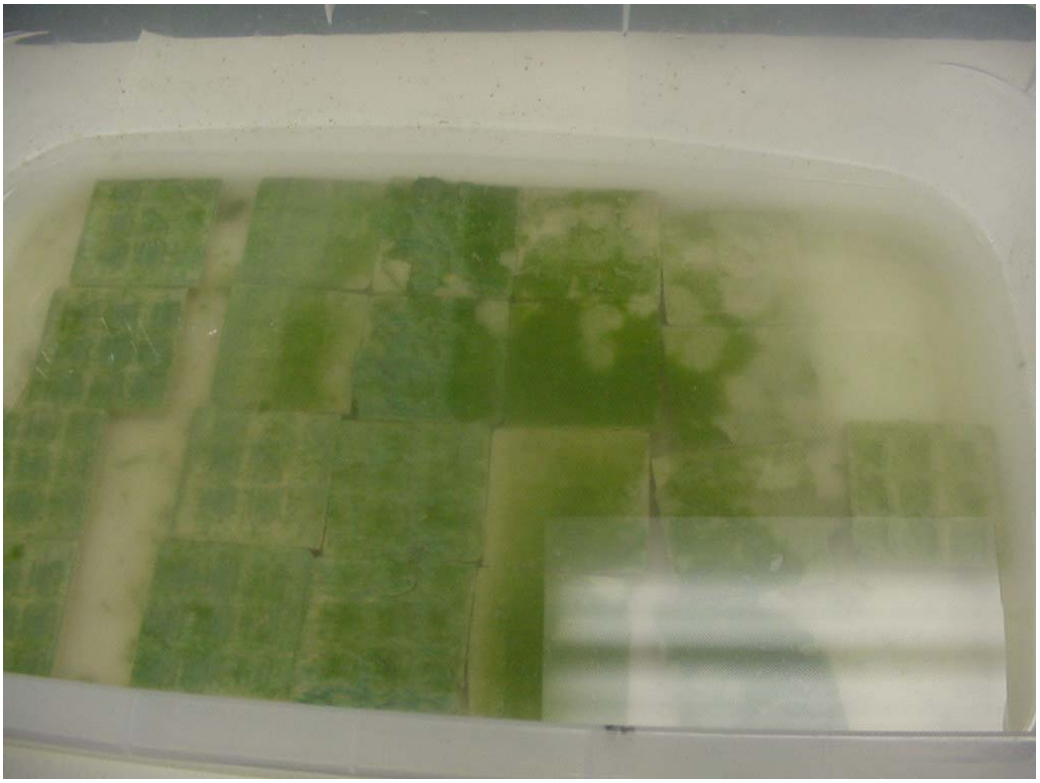
100mg/l day 1

200 mg/l day 1

APPENDIX F-2: BIOMASS TESTING 100 mg/l CONCENTRATION

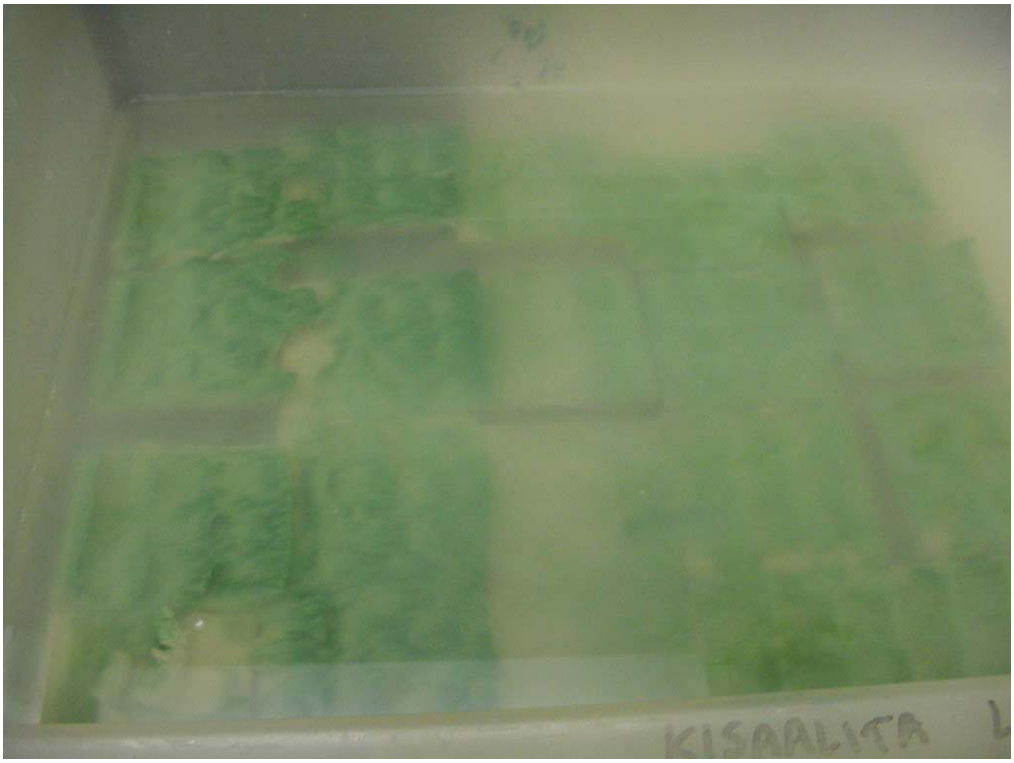


100mg/l testing rock to upslope region

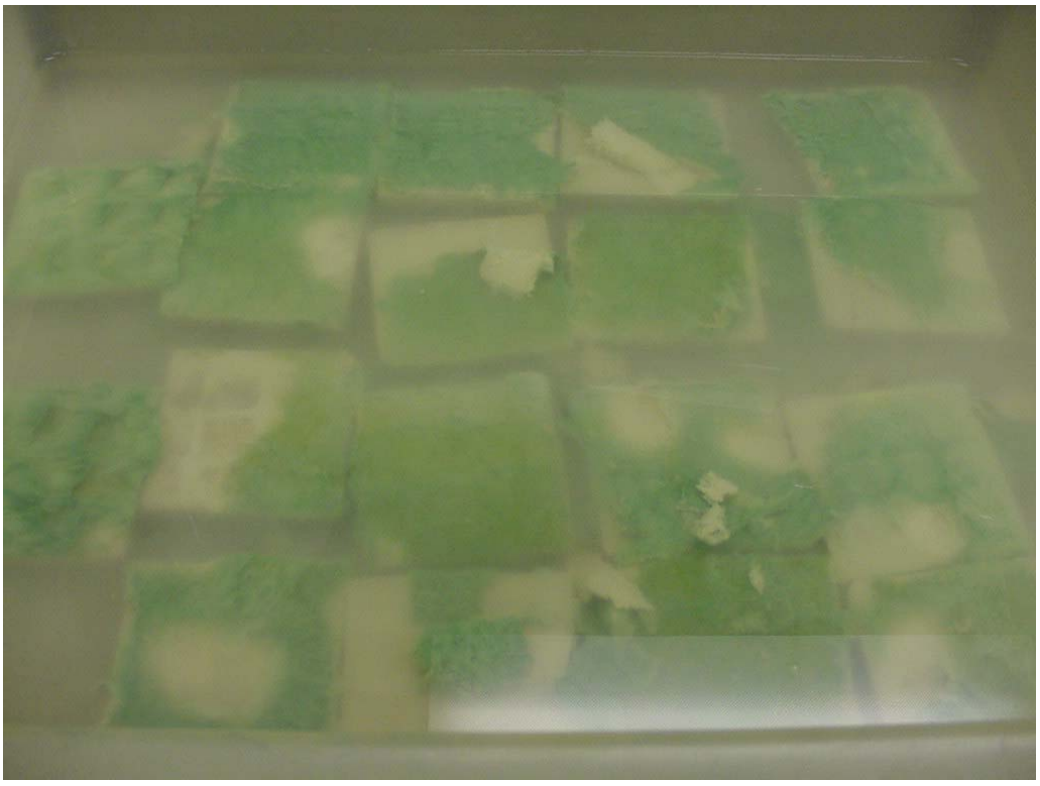


100mg/l testing weir to downslope testing region

APPENDIX F-3: BIOMASS TESTING 200 mg/l CONCENTRATION



200mg/l testing weir to downslope testing region



200mg/l testing rock to upslope testing region