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The Transition Zone: Impact of Riverbanks on Emergent Dragonfly Nymphs
Implications for Riverbank Restoration and Management.

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THE TRANSITION ZONE: IMPACT OF RIVERBANKS ON EMERGENT DRAGONFLY NYMPHS

IMPLICATIONS FOR RIVERBANK RESTORATION AND MANAGEMENT.

By

Kirsten H. Martin

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I would like to take this opportunity to acknowledge all of the individuals who have helped me traverse this long and often challenging path. From the very beginning, my family has supported me on this endeavor, providing words of encouragement, and ample cups of tea when most needed. My fellow faculty members have been ever ready with much needed humor, and stories of their own doctoral journeys. I also wish to thank my students, who have weathered this rocky path with me, and who in many cases have spent countless hours in the rain, mud, and searing sun, to monitor my often elusive research subjects. To my advisor, Dr. James Jordan, I express my gratitude for all of his support. I also wish to thank Dr. Charles Curtin, Dr. Mike Sutherland, and Dr. Betsy Colburn for their time and guidance. Finally I would like to thank my research subjects, the small, often overlooked dragonflies, for allowing me to witness a moment of their amazing lives.

The Transition Zone: Impact of Riverbanks on Emergent Dragonfly Nymphs Implications for Riverbank Restoration and Management.

Kirsten H. Martin

Abstract: The use of riprap in the restoration and stabilization of riverine landscapes is an issue of concern for many ecologists. While current methods of bank stabilization, especially those involving the placement of rocks (riprap) along the waterline, are effective in controlling erosion their presence changes habitat components (slope, substrate composition, near-shore river velocity) at the river-land interface. The additional impacts of river current, water temperature, soil composition, slope, and water level fluctuation, may further imperil emerging nymphs. The purpose of this research is to document the effects of riprap, location (upriver or downriver of hydroelectric intake/outtake facilities), water level fluctuation, river velocity, air temperature, water temperature, substrate temperature, and soil composition on the distance traveled to eclosure site by *G. vastus* and *S. spiniceps*, and the density of *S. spiniceps, G. vastus, N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. priniceps*. Knowledge of the conservation status of these species is fairly limited, although *S. spiniceps* (threatened), *G. vastus* (species of special concern), and *N. yamaskanensis* (species of special concern) are all currently listed on the Massachusetts Endangered Species list. Species density was determined through exuviae collection, and emergence distance was recorded from the edge of the waterline to the site of attached exuviae. Results of the study indicate that nymphal response to the observed abiotic features varies both with location and species. The presence of riprap had no significant effect on densities of *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. priniceps*, but did significantly reduce the distance traveled from the waterline by both *G. vastus* and *S. spiniceps*.

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Introduction

Riverine dragonflies experience two major habitat shifts in their life cycles. The first is during the emergence period when dragonfly nymphs must leave their aquatic habitat in order to venture out onto the adjacent terrestrial zone. The second shift occurs at the end of the emergence period, during eclosure, when the nymphs must shed their skin (exuviae) and emerge as fully formed adults capable of flight. In riverine dragonflies, these two major events often occur within riverbank habitats.

As with many other reservoirs that were created for the purpose of hydroelectric generation, water levels within the Turners Falls Reservoir (Gill, MA) can fluctuate. In the Turners Falls Reservoir, these fluctuations are further compounded by the inclusion of a hydroelectric intake/outflow facility. The intake/outflow facility draws water from the reservoir during periods of low energy demand and stores the water in a nearby reservoir at the top of Northfield Mountain. During periods of high energy demand, the water is released back into the Turners Falls Reservor. Therefore, water fluctuation is less a result of changes in precipitation, but instead is a function of energy demand. Water level fluctuations in the Turners Falls Reservoir average as much as 1.07m per day (U.S. Army Corps 1991).

In my three years of observing *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. priniceps* within the Turners Falls Reservoir in Gill, Massachusetts, 89 mortalities were recorded. The majority of the mortalities were the result of either predation (from snakes, frogs, birds, or chipmunks) or boat wake impact (Martin 2006, Martin 2006a., Martin 2008). The combined impact of boat wakes and water level fluctuation is of particular importance due to the heavy recreational boat activity in the reservoir. Of all of the species observed, *S. spiniceps* had the most observed mortality events (39) due to boat wake impact. On riverbanks without riprap, *S. spiniceps* traveled 0.37m from the waterline, while on riverbanks that had been stabilized with riprap *S. spiniceps* traveled only 0.28m from the waterline. This short travel distance may place the emerging nymphs within the boat wake zone.

During the course of my three year observational study, several interesting and noteworthy nymphal behaviors were observed. On several occasions, nymphs of *S. spiniceps* were observed among the riprap at the graded back bank site. The nymphs crawled under the rocks and eclosed. The newly emerged teneral then climbed up onto the side or top of the rock to complete the exoskeleton hardening process. A second note of interest involved the unusual "return to the river" behavior of several nymphs at a location that had an extensive sandy riverbank. Several nymphs of *G. vastus* and *S. spiniceps* were observed crawling back into the river, and were not seen to reemerge onto the same riverbank. Corbet (2004) suggests that this reentrance to the nymphal habitat may be the result of either (1) an issue of divided emergence (which is caused by the air temperature being below a certain emergence permitting temperature, or (2) the lack of suitable emergence support (rocks, emergent vegetation, etc.). The qualitative observations I recorded from my initial survey of the reservoir suggested to me that the various abiotic features at the waterline edge of the riverbanks might be related to both nymphal density and the distance nymphs travel from the waterline.

The natural history attributes of riverine dragonflies (Odonata) make them ideal candidates for indicating the success or failure of restoration ecology methodologies. Each stage in their life cycle is dependent upon the integrity and overall condition and function of both the biotic and abiotic components of their landscape. As juveniles, they are sensitive to changes in river substrate, water chemistry, and sedimentation. Adults may be sensitive to changes in both riverine vegetation and adjacent land cover (Corbet 2004). The emergence and eventual eclosure stage is the shortest and least researched period of an odonate's life, yet it may be the most important stage for ecologists to access riverbank restoration success. Successful completion of this stage may be affected by environmental changes both within the water column and on land; the actual physiological, chemical, or environmental factors triggering onset of eclosure are relatively unknown.

Dragonflies provide a unique lens through which ecologists can view the success or failure of riverine restoration efforts. Their complex life histories, which involve connections with multiple habitat zones (nymphs: rivers, shorelines, adults: riparian vegetation) provide ecologists with an opportunity for developing unique multi-habitat views of the impacts of bank restoration.

Most species of dragonflies complete their life cycles in freshwater habitats (rivers, marshes, streams, lakes) (Borror et al. 1989). Adults lay their eggs on submerged aquatic vegetation, soil surfaces near the water, or directly into the water. The larvae, or nymphs, remain in the water from a few months to several years depending on the species (McGavin 2001, Corbet 2004). Once larvae have reached their final instar, or growth stage, they must find species-specific eclosure sites. Eclosure is the process through which juvenile odonate nymphs split open their skins and emerge as young adults, which are called tenerals. In riverine systems, dragonfly nymphs eclose upon either riverbank substrate or emergent vegetation.

Studies of odonate biology and ecology have identified impacts of restoration strategies on odonate life cycle patterns (McLain et al. 2004, 2006). For example, the size of stones used in riprap bank stabilization may affect the distance traveled by *G. vastus* nymphs during emergence (McLain et al. 2006). Occurrence of perturbation, the scale at which it occurs, and changes in both chemical and physical (turbidity, flow regime) attributes may adversely affect emerging odonate species (Carchini and Rota, 1985, Death and Winterbourn, 1995, Rota 1987, Wagner et al. 1995, Buss et al. 2002).

Habitat use by dragonflies

Substrate composition and characteristics modify the availability of food resources, oxygen, and physical habitat structure, all vital components to the success of nymphal odonate populations (McClelland and Brusven 1980, Chovanec et al. 2002). Fluvial processes resulting in increased turbidity have also been shown to influence negative changes in exuvial densities and adult assemblages (D'Amico et al. 2004). Temporal changes in hydroperiod have also been shown to affect both density and diversity of predator-prey interactions among larval dragonflies (McCauley 2007). All of these factors influence the formation of the functional habitats within which juvenile odonates reside (Clifford et al. 2006).

Odonates have been well documented as being linked to specific habitat features. As nymphs, species such as *Coenagrion mercuriale* have been associated with areas that experience high groundwater inputs, while others like *Cercion lindenii* are found in areas with low groundwater flow (Castella 1987).

For many adult dragonflies, the presence of low-growing shoreline vegetation (i.e. reeds, emergent shrubs), aquatic macrophytes, and sun-exposed shorelines appear to be vital habitat features for oviposition and mating behavior (Chwala and Waringer 1996). Ethological studies conducted by Wildermuth and Spinner (1991), and Wildermuth (1993) indicate that choice of oviposition site depends not only on the presence of suitable vegetation, but also on the physical traits of the area. This research suggests that the presence of species-specific visual and tactile cues is vital to successful oviposition.

Dragonflies and disturbance

Because of their dependence on specific habitat variables and features, dragonflies present the ideal lens through which to view riverbank disturbance. Changes in habitat heterogeneity, defined by the quality and quantity of shoreline structures (banks), hydrological features, abiotic features (sunlight), and plant communities, may alter the density of dragonfly species present at a site (Schindler et al. 2003).

Instream changes, such as sediment size and deposition rate, may adversely impact nymphal species composition (Naiman and Decamps, 1997). Riverine sediment supply depends upon adjacent land uses, climatic changes, seasonal variability of discharge, flood regime, channel alteration and erosion. The composition of riverine substrate is highly complex, and is directly influenced by the geology and vegetation of an area (McClelland and Brusven 1980). Substrate composition and characteristics influence the availability of food resources, oxygen, and physical habitat, all vital components to the success of nymphal odonate populations (McClelland and Brusven 1980, Chovanec et al. 2002).

Fluvial processes resulting in increased turbidity have also been shown to cause changes in exuvial densities and adult assemblages (D'Amico et al. 2004). Diversity of adult dragonflies has also been correlated with macrophyte richness (Chwala and Waringer 1996, Rith-Najarian 1997) and vegetational succession (Suh and Samways 2005). All of these factors influence the formation of functional habitats within which juvenile odonates reside (Clifford et al. 2006). Temporal changes in hydroperiod have also been shown to affect both density and diversity of predator-prey interactions among nymphal dragonflies (McCauley 2007), and changes in both chemical and physical attributes affect distribution of dragonfly species (Carchini and Rota, 1985, Rota 1987).

While much has been learned about the ecology of adult and in-stream nymphal dragonfly stages, relatively little is known about species-specific habitat needs during the emergence and eclosure period. The definitive work on odonate ecology (Corbet 2004) specifically highlights this gap in knowledge. The lack of knowledge in this area is a barrier to understanding the ecological implications of managing and restoring riverbanks.

Bioindication of riverbank restoration

My research focuses on the spatial link that exists between the dragonfly nymphs and the riverbank habitat that they must cross in order to become adults. Specifically I aim to describe the impact of (1) riprap, (2) air, water, and substrate temperature, (3) riverbank slope, (4) width of riprap zone, (5) circumference of rocks within the riprap zone, (6) water velocity, (7) water level fluctuation, (8) sediment composition (%sand, %silt, %clay), and (9) site location relative to the hydroelectric intake/outflow facility on the (1) density of *Gomphus vastus* (Cobra clubtail), *Stylurus spiniceps* (Arrow clubtail), *Neurocordulia yamaskanensis* (Stygian shadowdragon), *Micromia illinoiensis* (Illinois river cruiser), *Epitheca princeps* (Prince baskettail), *Dromogomphus spinosus* (Black-shouldered spineyleg), and *Ophiogomphus rupinsulensis* (Rusty snaketail), and (2) the emergence distance traveled by *Stylurus spiniceps* and *Gomphus vastus*.

The study was located within the Turners Falls Reservoir (Gill, MA: Franklin Co.). The reservoir is bounded on the north by the Vernon, VT dam, and on the south by the Turners Falls dam. The Northfield Mountain Pump Storage intake/outflow facility designated the center of my study area. Fifty transects (25 control sites (eroded), and 25 riprap sites) were located upriver and fifty transects (25 control sites (eroded), and 25 riprap sites) were located downriver of this location. Each transect covered 1.5m (5ft) of shoreline and extended 12.19m (40ft) vertically up the slope of the bank. Each transact was visited at least twice a week beginning on June 5th, 2008, and ending on August 14th, 2008. Sites were visited between the hours of 10:00AM-3:00PM.

Species density per 60.98m^2 (200ft^2) for *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, *E. princeps*, and the distance traveled from the edge of the water to the eclosure site for *G. vastus* and *S. spiniceps* was determined through either (1) direct observation of eclosing nymphs, or (2) presence of attached exuviae (Chovanec and Waringer 2001, Corbet 2004, Foster and Soluk 2004).

Specimens were identified to the species level using a northeastern Anisoptera key (Soltesz 2002). Analysis was divided into three parts, (1) comparison of abiotic features and location, (2) species density, and (3) emergence distance. The effect of location on abiotic factors was analyzed using a Mann-Whitney U test. The influence of location on species density and emergence distance was also compared using Mann-Whitney U. Further analysis of the correlation between species density, emergence distance, and abiotic factors was conducted using Pearson correlation. In order to more clearly encompass the complexities of the Turners Falls Reservoir, backwards elimination multiple regression (BEMR) was conducted on the connections between (1) location, abiotic factors, and *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus*, *O. rupinsulensis*, *M. illinoiensis,* and *E. princeps* densities, and (2) location, abiotic factors, and emergence distance of *G. vastus* and *S. spiniceps*. BEMR analysis resulted in the generation of several significant models. One "best-fit" model was chosen based on the greatest number of significant contributions by individual abiotic factors and the highest possible r^2 value. For the density analysis, r^2 >0.50 were considered to be biologically significant, due to the small scale effects of habitat change on emergence behavior all r^2 were considered to be biologically significant.

The BEMR models generated by this study are illustrative of the location and species-specific responses to habitat variables. While no one dragonfly species emerged as the ultimate bioindicator for this location, the intriguing connections between location and species response suggest that a multi-species, and multi-scale approach to riverbank bioindcation may be extremely useful when considering riverbank restoration or management.

The following dissertation is organized into three major sections. The first section provides an overview of how the scale framework has been incorporated into the field of restoration ecology and presents an overview of several current multi-scale riverine bioassesment programs. This section also includes a discussion on how dragonfly bioindicators can be included within these frameworks. The second section presents the core findings of the research study.

Redefining restoration to include bioindicators

The aim of restoration ecology is to restore damaged systems to a prior natural state. Ultimately, the determination of what constitutes "natural" depends upon the ability of ecologists to define the physical and ecological processes present in the system. Ecosystems are highly complex and variable, and the ecologist's ability to accurately describe the intricate workings of these systems is often limited by time and budgetary constraints. The application of bioindicators can aid in clarifying connections, and can help ecologists create more holistic models of natural systems. In my case study, I focused on the potential impacts of one component of riverbank restoration, the placement of riprap within the boat wake zone.

Since the ultimate goal of restoration ecology is to restore systems to a prior natural state, entire ecosystem functioning should be considered when implementing restoration programs. Bradshaw (1983) provides a theoretical description of ecosystem development following restoration. In his description, ecosystem function is aligned with biomass and nutrient content, and ecosystem structure is tied with species richness and overall ecosystem complexity. Depending on the species or system being studied, ecologists choose the scale and methodology with which they will work. In the case of odonates, considerations of scale (temporal and spatial) need to flexible based on which species is being studied, and the stage of life that is the focus. Since these choices are made prior to the beginning of research, ecologists and restoration managers must base their judgments of what constitutes natural states on multiple layers of different perceptions.

Consideration of the perceived original state is also essential to Jordan's (2000) concept of restoration ecology. If a model landscape or ecosystem is classified as natural, then the implementation of a restoration program is a way to prevent that area's ecological structure and function from being impacted by management.

The decision on which ecological model to use directly influences which methodology is implemented. This in turn determines the success and/or failure of the restoration effort. Ultimately the question of restoration is one of description. The desired goals, and particularly the desired state(s) to which an area or system will be restored to, need to be delineated prior to the start of restoration. Bioindicators can be used at all stages of the restoration process. In the beginning, bioindicators can be used to assay current ecological conditions, and can highlight areas of greatest concern. During the restoration process, bioindicators can provide ecologists with vital feedback on how restoration measures are impacting the system. Ultimately, because bioindicators are integral members of the system being modified, and not an outside unit of measurement, their density, behavior, and life history traits provide ecologists with a clearer analysis of restoration impact.

The characteristics of the restored ecosystem proposed by Ewel (1987) are remarkably similar to the definition of a living organism. Like living organisms, ecosystems are composed of both living and non-living parts. If ecologists were to view ecosystems as living organisms, composed of a myriad of interconnected components, then restoration ecology should reflect this framework. In order to understand the workings of a human body, the intricacies of cellular systems and chemical interchanges are first studied, then the study moves on to tissues, organ and skeletal systems, and finally to the organism as a whole. Study of the human form does not end at that point. We could move outwards to studying interpersonal interactions, sensory awareness of place, community organization, ever expanding into greater and greater scales of research. Restoration ecology would benefit from examination through a similar matrix. Prior to performing major surgery to “restore” a “damaged” system, care must be taken to adequately understand the myriad parts that make up the whole.

The same introspective examination of potential effects is vital when applying restoration to riverine systems. Regardless of what method of restoration is implemented, it is imperative that ecologists focus not only on rehabilitation of the entire ecosystem (in terms of structure and function), but also on individual components (both abiotic and biotic) of the riparian system. It is the connections between these myriad components that form the basis of a healthy, functional, river ecosystem.

Usefulness of scale frameworks in restoration

As with many other issues in restoration ecology, defining the scale at which an ecosystem is being viewed is a vital first step towards determining restoration goals. From an ecological perspective, the implementation of a restoration program and ultimately the definition of success are tied to the initial scale at which the ecological system of interest is viewed.

In order to begin to understand the inherent processes that power ecological systems, ecologists must first strive to define their underlying patterns and processes. Riverine systems are highly complex landscapes. The interplay between the physical characteristics of natural systems and their biological inhabitants is variable, and can change depending upon external factors such as seasonality and topography. Fluvial action, erosion, transport, and deposition of sediments also helps create a dynamic and diverse riverine landscape (Vannote et al. 1980, Ward 2002).

Riverine landscapes contain ecotones, which are transition zones between adjacent ecological habitats or systems that are described by a unique set of characteristics, both spatial and temporal (Naiman and Decamps 1997). In riverine systems these ecotones can exist within the water column as seasonal differences in water temperature and chemistry or differences in substrate composition (due to fluvial processes). Ecotones also exist in the boundary areas, the transition zones between the river and the accompanying upland landscape.

Rivers impart their own unique pressures on the restoration ecology debate. As with terrestrial systems, a discussion of riverine system restoration must begin with a designation of scale. Within this scale designation lies a description of the inventory tools and methods, which would be most effective. Noss (1990) describes three elements of ecological systems (functional, structural, and compositional) (Fig 1).

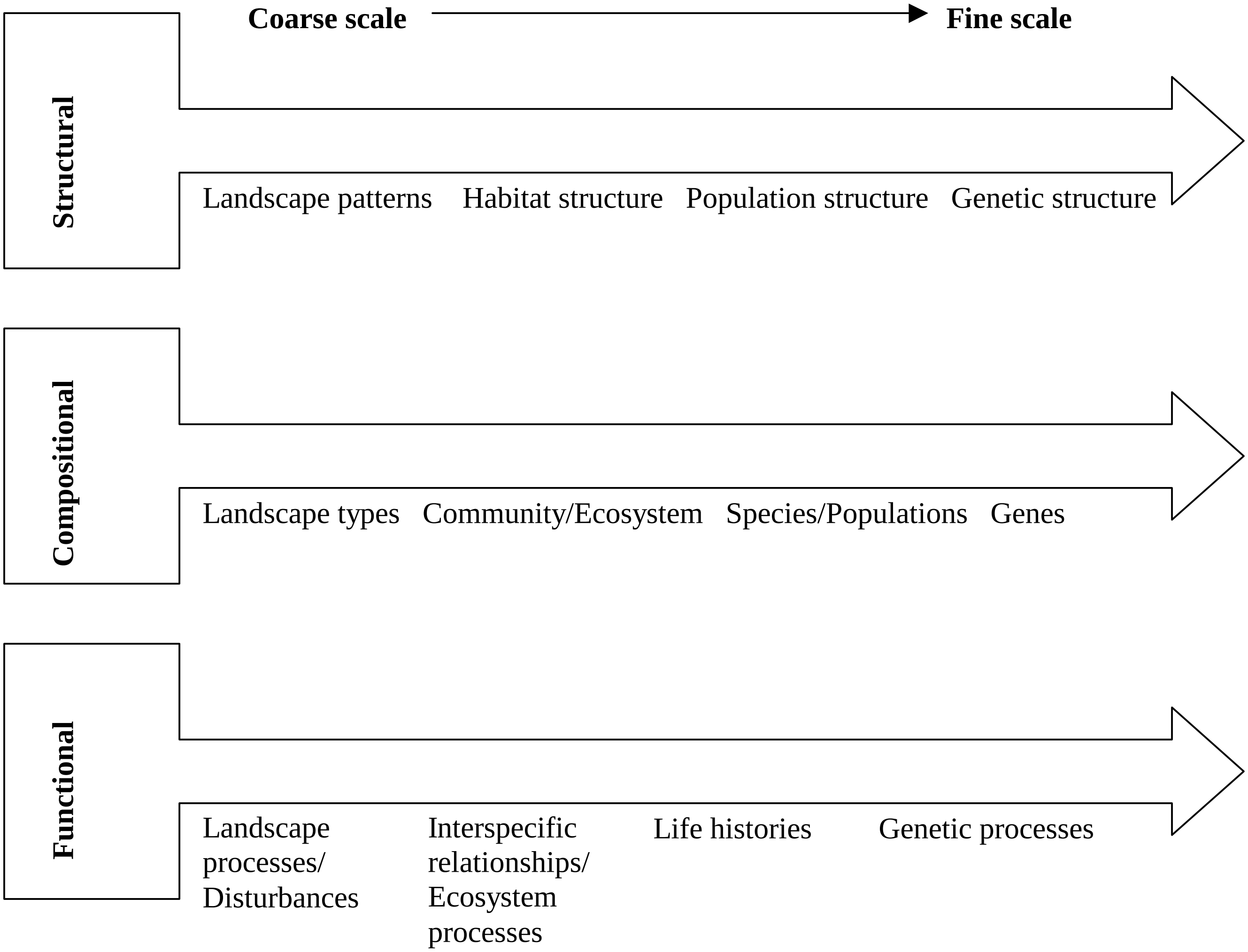


Figure 1. Structural, compositional, and functional aspects of biodiversity with related scales of organization. Adapted from Noss 1990.

Each of these elements can be viewed from coarse to progressively finer scales. For example, when a system is viewed through its functional aspect we would begin at the coarse scale of landscape processes and disturbance. By proceeding through finer and finer scales we would ultimately reach the finest scale (genetic processes). The use of this methodology encourages consideration of the interconnected multi-scalar aspects of ecosystems. By moving back and forth along the gradient, a more holistic view of the ecological system may be obtained.

While each aspect can be the focus of independent research, ultimately they must be considered as interconnected elements. In riverine systems the three elements of Noss's (1990) scheme can help direct the focus of ecological research. Each of the elements can be viewed at a variety of scales, from the fine scale (species-level models) to the coarse scale (ecosystem-level models). The shifting of focus, between fine scale and coarser scale frameworks is essential when viewing riverine systems.

Perception of scale is the fulcrum of restoration efforts. The level at which an ecosystem is initially perceived mandates which restoration efforts will be initiated. Allen and Hoekstra (1987) state "one of the most important sources of confusion in ecology is inexact reckoning of scale, since a change in scale changes the degree of inclusiveness of entities in the system." The temporal or spatial scale that an ecologist chooses to view an ecosystem through is entirely dependent upon the species being studied. Hull and Robertson (2000) argue that the scale chosen by the researcher directly determines the ecological descriptors that will be studied and which temporal, spatial, and organizational properties of those descriptors will be the focus of the research. Scale designation centers both the methodology and the ultimate outcome of the research.

Defining restoration success

The term success, when used in restoration ecology, is rarely easy to define. Depending upon the goals of the individual, success may be defined as (1) a return to a naturally occurring ecological state, (2) formation of an ecologically functional state, or (3) establishing an aesthetically pleasing state. According to Ewel (1987) successfully restored ecosystems can be typified as having (1) the ability to resist invasive species, (2) the ability to be sustainable, and (3) the ability to hold nutrients and be productive (i.e. restored areas should retain at least as many nutrients as unrestored reference areas).

Bradshaw (1987) uses the analogy of a watch to describe the usefulness of restoration ecology. He writes, "it is easy to take a watch apart and to describe its parts…we can put the watch together… If we succeed, we have tested our understanding quite critically". It is only in the perfect system that all parts to the watch are in place before we attempt to take it apart. Bradshaw (1987) acknowledges this in regards to the role of restoration ecology in reassembling damaged ecosystems. He states, "the successful restoration of a disturbed ecosystem is an acid test of our understanding of that system" (Bradshaw 1987). But, are the parts of a river system truly interchangeable? Can localized effects of climate, precipitation, and temperature be so insignificant as to relegate these natural assets to the level of Bradshaw's (1987) interchangeable watch parts? Ultimately the success or failure of a riverine restoration program depends on how well the physical and biological components of the system are understood.

Bradshaw (1983) suggests that it is "not whether we can take ecosystems to bits on pieces of paper, however scientifically, but whether we can put them together in practice and make them work". According to Bradshaw (1983), a degraded ecosystem in which no restoration efforts are undertaken will have two possible outcomes. In the first outcome the system will degrade even further (as a result of neglect). In the second outcome, the system will progress toward a state of increased ecosystem structure. Species richness and complexity will increase through normal ecosystem development, but overall ecosystem function will remain limited.

If the goal of restoring the original ecosystem is achieved, then the system will theoretically experience complete restoration. Restoration in this context implies both an increase in ecosystem structure and an increase in ecosystem function. Ultimately, the definition of “success” is dependent upon two variables. The first (the goal of the restoration) is in itself dependent upon the second variable (who is defining the goals). Biologically based restoration goals can be aimed at single-species, community, landscape, or ecosystem recovery. The choice of which goals are predominant are designated by a variety of potential user groups (scientist, manager, politician, community). The goals of any one particular individual or group are highly variable and can directly influence the determination of restoration success.

Bioindicators and biological assessment

Bioindicators are species, groups of species, guilds, or communities which have been selected based on their specific sensitivities to chosen biotic or abiotic environmental features. Once sufficient information has been gathered, these bioindicators can then be used in place of direct measurement of abiotic features. The use of living organisms as indicators of ecological system status has been well documented. Generally, bioindicators display limited tolerance of a specific ecological unit (i.e. pH, temperature, sedimentation, etc.). The presence or absence, or in some cases behavioral changes, of these bioindicators can then be used to determine the relative “health” of the system (Gadzala-Kopciuch et al. 2004).

Insects have been used extensively as bioindicators in a variety of systems, from the use of ants as indicators of land use changes in terrestrial systems (Anderson et al. 2002), dung beetles as indicators of succession in savanna ecosystems (McGeoch et al. 2002), grasshoppers as indicators of disturbance (Andersen et al. 2001), and fruit flies, beetles, and wasps as indicators of logging induced forest disturbance (Akutsu et al. 2007).

In aquatic systems, macroinvertebrates are commonly used to monitor levels of chemical accumulation (Gadzala-Kopciuch et al. 2004), and general habitat quality. In flowing water (lotic) systems such as rivers and streams, macroinvertebrates have been used as indicators of organic pollution (Hilsenhoff 1987), stream acidification (Braukmann 2001), water flow (Clausen and Biggs 1997), and sedimentation (McClelland and Brusven 1980) . Aquatic macroinvertebrates have also been used as indicators in still water (lentic) systems (Foote and Hornung 2005).

Dragonflies as bioindicators

The use of dragonflies as bioindicators in lentic habitats, such as lakes, ponds, wetlands, is well documented (Schmidt 1985, Corbet 1993, Samways et al.1996, Culhane 2005, Niba and Samways 2006, Bried et al. 2007). Dragonfly bioindices can readily be applied to lotic (riverine) habitats as well (Chovanec and Waringer 2001). Each stage of odonate life history can potentially be used for bioindication.

Unfortunately, due to our limited knowledge of species-specific emergence and eclosure behavior the ability to create models based on entire dragonfly life cycles is limited at this time. One of the main purposes of my research is to provide insight into the emergence and eclosure behavior of several New England riverine dragonflies, and to propose a holistic life cycle model, which can then be used in restoration risk assessments.

One advantage of using dragonflies as bioindicators is the relatively small number of species in the infraorder. In most cases, identification of species, whether by exuviae or adult surveys, can be determined in the field (Chovanec and Raab 1997). Many species of odonates are dependent on heterogeneous riverine and terrestrial microhabitats, which make them good bioindicators of the ecological conditions of river- terrestrial ecotones (Chovanec and Waringer 2001, Clarke et al. 2003). The relatively long life cycle of most odonate species enables long-term monitoring of environmental stressors.

Odonates are also useful as bioindicators because the presence of breeding populations can be determined either through the direct observation of breeding behavior, or through records of nymph emergence and exuviae density (Chovanec and Raab 1997). Finally, reaction to changes in habitat can be observed (1) through the presence or absence of specific species, or (2) by changes in density of those species (Chovanec and Raab 1997).

The relative sensitivity of individual dragonfly species may also serve as a vital bioindicator for riverine health. Stenotopic species (species which are sensitive to changes in particular habitat (abiotic or biotic) conditions are better indicators of changes in biotope conditions than euryotopic species (those that can withstand changes in habitat conditions) (Samways and Steytler 1996). Specific species, or assemblages of species, can be selected as bioindicators based on their individual habitat sensitivities, providing a site-, region-, or landscape-specific analytical tool.

Multi-scale riverine assessment and dragonfly bioindicators

There are several existing multi-scale frameworks through which to view “natural” riverine ecosystem functioning. One example of a multi-scale framework is the European Union’s Water Framework and Habitats Directive (WFD). The WFD mandates that all rivers be considered in terms of their inherent ecological quality (Newson and Large 2006). The WFD framework (Table 1) does not exclude processes of change (erosion and sedimentation) but instead views the functionality of river systems in providing for biotic habitat.

WFD reference sites are classified according to a hierarchical water body typology (rivers, lakes, transitional/coastal waters, artificial water bodies, or heavily modified water bodies) (Davy-Bowker et al. 2006). This is in direct contrast with the RIVPACS-type (River Invertebrate Prediction and Classification System) predictive models that first classify sites based entirely on their macroinvertebrate fauna (Davy-Bowker et al. 2006, Feio et al. 2006).

Table 1. Classification of ecological status in rivers based on the WFD framework (adapted from Newson, M., and A. Large (2006), and the potential incorporation of emergent dragonfly ecology into the framework.

Quality elements (EU WFD)	Description (EU WFD)	Incorporation of Emergent Dragonfly Ecology
Biological elements (habitat and species)	Composition and density of aquatic flora Composition and density of benthic macroinvertebrate fauna	Vegetation analysis of riverbank, species specific eclosure locations
Hyrdomorphological elements which support the biological elements (sediment flows, fluvial geomorphology)	Quantity and dynamics of the water flow Hydrological regime River continuity River depth and width variation Structure and substrate of the river bed Structure of the riparian zone Connection to groundwater	Impact of sedimentation and nearshore water velocity on ability of nymphs to emerge at riverbank
Chemical and physio-chemical elements supporting the biological elements	Thermal conditions Oxygenation conditions Salinity (where applicable) Acidification status Nutrient conditions Specific pollutants Pollution by all priority substances identified as being discharged into the body of water Pollution by other substances identified as being discharged into the body of water	Impact of air, water, and substrate temperatures on nymphal densities and emergence distance

The WFD provides a defined method of determining the overall health (and potentially attainment of restoration success) of riverine systems. It holistically includes the impact of scale (landscape dynamics) yet its successful implementation relies explicitly on fluvial geomorphology as a regulating tool (Newson and Large 2006). The WFD not only requires members of the EU to monitor and assess water systems, but more importantly it mandates improvement of the “ecological quality status” of surface waters (Woods 2004, Davy-Bowker et al. 2006). The “ecological quality status” relates to riverbank restoration as it addresses issues of channelization, increased flow rate, sedimentation, and the ability of the local system to sustain diverse ecological function. Ecological status is determined from the assessment of several variables at different scale considerations (Table 2).

Table 2. EU WFD normative descriptions, and use of benthic invertebrates, in defining high, good, and moderate ecological status in rivers (modified from European Parliament Directive 2000/60/EC).

Element used in status classification	High status	Good status	Moderate status
General	No (or minor) anthropogenic alterations to physico-chemical and hydromorphological elements Biological quality elements are aligned with undisturbed river condition	Low levels of anthropogenic alterations to system Biological quality elements deviate only slightly from undisturbed norms	Moderate levels of anthropogenic alterations to system Biological quality elements deviate moderately from those normally associated with undisturbed river conditions.
Benthic invertebrate fauna	Taxonomic composition and abundance correspond totally to undisturbed condition Ratio of disturbance sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed level Diversity levels show no signs of alteration from undisturbed level	Slight changes to composition and abundance of taxa compared to undisturbed condition Ratio of disturbance sensitive taxa to insensitive taxa shows slight alteration from undisturbed level Diversity levels show slight alteration from undisturbed level	composition and abundance of taxa differ moderately compared to undisturbed condition Major taxonomic groups are absent The ratio of disturbance sensitive to disturbance insensitive taxa are lower than good status Diversity levels show slight alteration from undisturbed level

The WFD framework for accessing surface water conditions serves as a good model for assessing riverbank conditions. The model proposed by Grantham et al. (2008) (Fig 2), provides an ideal framework for placing odonate ecological studies at the center of riverbank assessment.

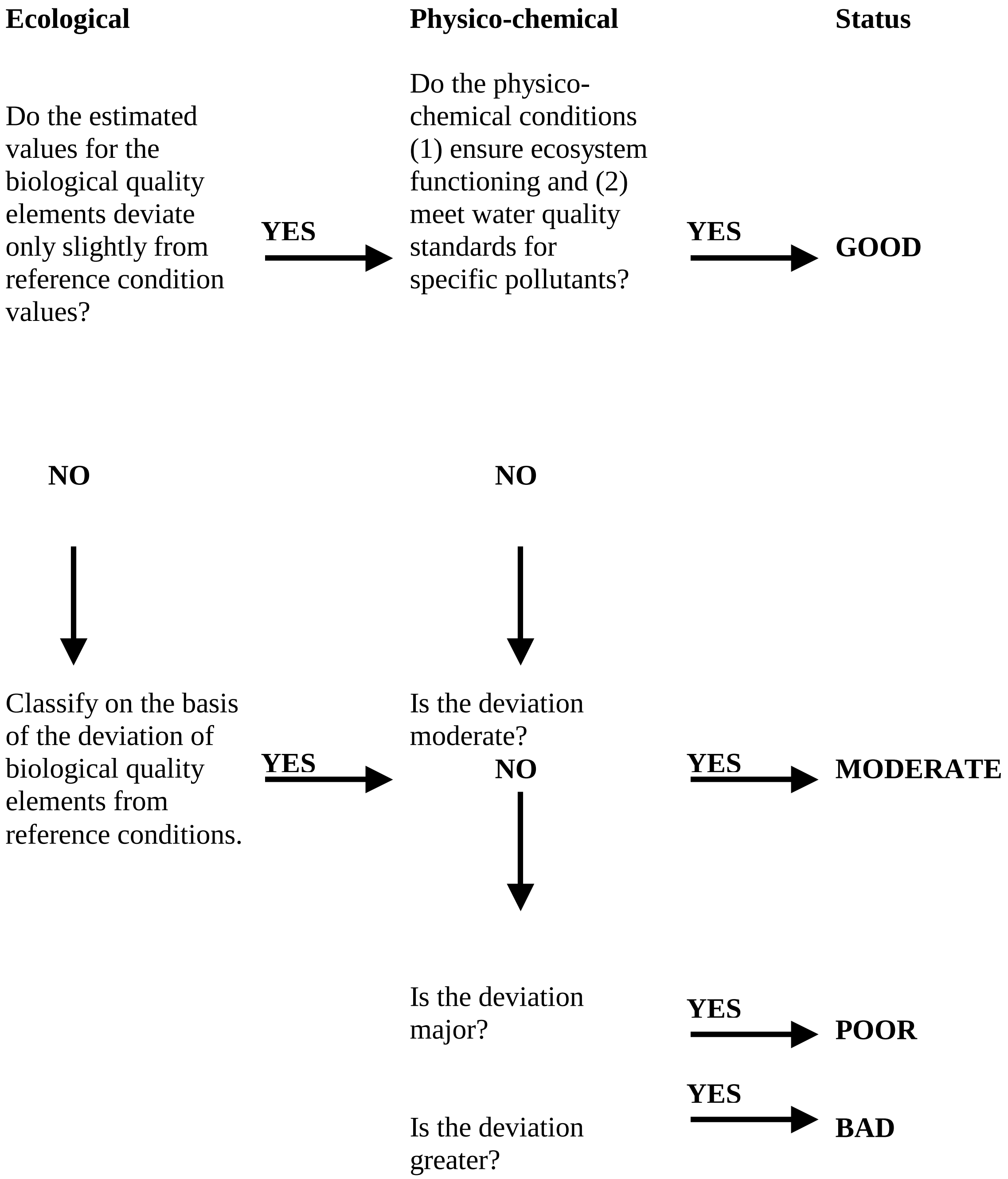


Figure 2. Classification of surface waters under the Water Frameworks Directive (Grantham et al. 2008).

The WFD is focused on large-scale (i.e. watershed) assessment of management, but with some acknowledgment of the importance of individual site-based features, such as reservoirs, and riverbanks, its overall framework may be a valuable guide for riverbank restoration (Table 13). The WFD can provide a foundation for viewing odonates as bioindicators of restoration success. At the biological elements level, analysis of odonate communities provides an indicator of habitat structure and function. Odonate nymphs may also provide insight into the hydromorphological elements level, although this level traditionally focuses on abiotic indicators (water flow, river depth, etc.). The presence or absence of specific odonate species is directly correlated with these indicators.

One of the greatest strengths of the WFD framework is its acknowledgement of the differences between natural water bodies and those that have been modified or created (slow-moving water systems like reservoirs, lakes, and wetlands). Based on the unique fluvial processes and restoration objectives in controlled rivers, analysis of dragonfly assemblages may be an ideal tool to evaluate the health of the system.

The aim of RIVPACS models is to define a set of appropriate environmental predictor variables which can be measured at any type of river location (Clarke et al. 2003). Test sites are compared to reference sites through predictive analysis (observed versus expected). The current debate between the WFD framework and the RIVPACS model centers around their applicability to larger scales (i.e. landscape level). RIVPACS models generally are better descriptors of community variation, but may not be the best models when looking at larger scales (Davy-Bowker et al. 2006). A second controversy involving RIVPACS models is their reliance on reference conditions. Reference conditions are inherently spatially and temporally based; even if the actual site no longer exists, historical conditions gleaned from the site can still be used (White and Walker 1997).

The counterpart to the WFD in the United States is the Environmental Protection Agency's (EPA) Environmental Monitoring and Assessment Program (EMAP). The EMAP is similar to the WFD in that both incorporate multi-scale research strategies. The designation of indicators is based on multiple layers of biological organization (U.E. EPA 2002). The EMAP framework is similar to the scale framework discussed by Noss (1990), and contains three main categories (1) structural, (2) level of organization (Noss (1990) terms this compositional), and (3) processes (Noss (1990) terms this functional). Within the structural and processes categories, the EMAP framework allows for consideration of multiple scales (Table 3).

In addition to incorporating a multi-scale framework into their protocol for determination of indicator species, the EMAP research protocol relies heavily on cross-scale monitoring. The monitoring strategy includes course scale analysis of landscape changes as well as fine scale analysis of site-specific features such as substrate size and canopy coverage (U.E. EPA 2002).

Table 3. EMAP framework for development of indicators with consideration of different biological organization levels and the structural and functional (process) aspects of each biological level (modified from U.S. EPA 2002).

Level of Organization	Structure	Processes
Gene	Heterozygosity	Recombination rate Mutation rate Polyploidy rate
Individual	Condition Anomalies/deformities Maximum size Tissue contamination	Metabolic rate Growth rate Fecundity
Population	Abundance Age class distribution Size class distribution	Reproduction rate Growth rate of population Death rate Evolution/speciation
Community	Relative abundance Richness Trophic composition Reproductive composition Habitat guilds	Competition/predation Disease/parasitism Mutation Recovery rate
Watershed or Landscape	Regional diversity Homogeneity Hot spots Patches/patterns Fragmentation/recovery	Water delivery Chemical delivery Material delivery (i.e. sediment) Energy flow Population sources and sinks Nutrient cycles

Another tool currently being used seeks to incorporate the abiotic components of hydrobiology and perturbation ecology with aspects of organismal ecology. The Odonate Habitat Index (OHI) is an assessment tool that has great applicability for riverine restoration ecology. The OHI is calculated from determinants of species-specific values (habitat preferences, species densities, and indication weight) (Chovanec and Waringer 2001). These aspects distinguish eurytopic (habitat generalists) from stenotopic (needing specific habitat or habitat attributes) species. The OHI can then be used in comparison with known reference conditions for the species, or species assemblages being used as bioindicators, and serves as an evaluation of restoration success (Chovanec and Waringer 2001).

The main difficulty with the OHI is the lack of species-specific and area-specific reference data. Odonate density is linked with habitat suitability, which varies considerably between locations. Under the OHI, the entire Turners Falls Reservoir would be classified as an H1 habitat type (main river channel, and associated inshore structures), and odonate density would be hypothesized to be similar along the entire stretch of the system. Although the Turners Falls Reservoir is a managed system, its habitat is certainly not uniform. An OHI value, created for the Reservoir, would therefore be unduly generalistic and highly unsuitable for microscale questions about the impact of bank restoration.

A second potential problem with using the OHI to access the impacts of bank restoration on odonates is the difficulty of determining eurytopic from stenotopic species. Eurytopic species by definition would be found throughout the system because they do not normally display specific habitat associations. Stenotopic species would be considered a better model for assessing restoration impact. In my study, all seven species were found throughout the Reservoir. This would suggest that they were all eurytopic and not overly suitable predictors of habitat change. But each (with the exception of *E. princeps*) displayed very individual responses to both location (above or below the intake/outtake facility) and the selected abiotic features, this suggests that while the species may be present throughout the system, each responds differently to specific habitat attributes.

Riverine odonates are prime indicator species of river status, and by extension restoration success. As one of the few species that has ties to both the aquatic and terrestrial habitat, and one which uses a specific localized area for its transformation into an adult, odonate nymphs can serve not only as indicators of aquatic health, but more importantly indicators of restoration health.

Application of odonate bioindicators

With the advent of the frameworks such as the WFD, the OHI, and the EMAP, the role of odonates as bioindicators of riverine ecosystem functioning has become increasingly more important. The study of dragonflies provides one vital clue for the overall state of restored systems. Nymphal forms of odonates and other benthic insects are important components of many aquatic food chains (Lemly 1982), and the presence or absence of these organisms is correlated with the amount of inorganic sedimentation (McClelland and Brusven 1980, Lemly 1982).

Chovanec (2000) proposes a way to incorporate dragonfly bioindicators into the WFD surface water classification scheme. In the WFD framework, the ecological status of surface water is classified on a range of five values from the highest quality to the lowest quality. For controlled water systems, the highest level of ecological quality would be typified by (1) a high density of the dominant dragonfly species, (2) a direct correspondence to a type-specific reference community, and (3) a high proportion of autochthonous stenoecious species (species that originate in the place where they are found, and require a specific set of habitat characteristics). A system classified as having moderate ecological quality would show (1) significant changes from a type-specific reference condition (as much as a 75% loss of species), and (2) lack of autochthonous stenoecious species. The lowest WFD surface water classification scheme would be typified by the absence of both autochthonous and stenoecious dragonfly species (Chovanec 2000).

As with most landscape-level frameworks, care should be taken when considering their application to site-specific questions. As suggested by the results of my case study, single abiotic features such as the location of a hydroelectric intake/outtake facility can have very significant effects on species-specific density and emergence behavior.

Odonate nymphs are directly dependent upon the aquatic environment, and their long ontogenetic process allows for long-term monitoring (Chovanec and Waringer 2001, Clarke et al. 2003, D'Amico et al. 2004). Dragonflies are susceptible to habitat changes whether they are anthropogenically or naturally induced. In a study by Carchini and Rota (1985), a correlation was found between certain chemico-physical parameters and select species of odonate nymphs. The study suggests that specific odonate species, or potentially assemblages of species, may serve as indicators of pollution.

A high correlation exists between the structural components of aquatic habitats (specifically the quality and quantity of aquatic vegetation, hydrological features, sun exposure, and shoreline structures) and the presence of certain odonate species (Chwala and Waringer 1996, Schindler et al. 2003). In one study conducted by Steytler and Samways (1995), populations of adult dragonflies were monitored to assess habitat preference. Steytler and Samways (1995) suggest that dragonflies, specifically adult males, have identifiable habitat preferences. Several taxa were negatively correlated with shaded sites, while two species (Zygoptera) *Pseudagrion hageni* and *Chlorolestes tessallatus* appeared to be heliophobic. Several species were correlated with slow-moving water, while others were associated with flowing water. The physiognomy of shoreline vegetation also appeared to play a role in odonate presence. Several species of Anisoptera were positively correlated with the presence of short grass, while others required the presence of conspicuous projections (sticks, rocks, etc.) for perching. Since presence of adult dragonflies correlates with an areas potential for mating success, the absence of adults could serve as an indicator of habitat homogeneity.

Species-specific habitat preferences during the eclosure period are relatively undocumented. While substrate composition in the near-shore zone appears to influence nymphal emergence and there have been documented instances of emerging nymphs reentering the water following emergence onto “unsuitable” shores (Corbet 2004), these data are largely incidental, and have not been supported by either direct manipulation of habitat or extensive observations.

Unfortunately, one of the greatest weaknesses of the WFD, or many other standardized assessment tools, is their reliance on both (1) a known reference condition and site, and (2) in-depth analysis and monitoring of the area in question. Describing a reference site, or a range of reference conditions for reservoir systems is difficult. The influence of countless site-specific variables makes it virtually impossible to delineate ideal reference conditions. The species-specific life history needs of dragonflies, and other bioindicators, add an additional difficulty to defining one “best-fit” reference condition. The other difficulty when using bioindicators, and particularly dragonflies, with many standardized assessment tools, is the relative lack of in-depth life history data. I have continually encountered this “data-void” during this study. Dragonflies have enormous potential to serve as biological windows into the relative success or failure of riverine restoration methods; unfortunately the biggest hurdle to their successful integration into assessment schemes is the absence of both species-specific and area-specific data.

As dragonflies continue to be monitored as vital bioindicators of overall river system health, careful consideration should be taken when choosing both the species that will be observed, and the locations where observations will take place. Based on the results of my case study of the Turners Falls Reservoir, these two considerations are of paramount importance. The two species I chose to document had been found in large numbers during previous surveys, and they were chosen because they did not appear to display any site preferences. These assumptions did not remain clearly defined following analysis of the data. If dragonfly behavior (particularly emergence behavior) is being used to inform bank restoration practices, then there needs to be a clearer understanding of species-specific responses to site-based abiotic features. Although restoration practices are often tailored to fit the specific needs of the location, seemingly short stretches of restored bank may represent a wide spectrum of different environmental features to emerging dragonflies.

The limited boundaries of my own study highlight the need for extensive “pre-assessment” of existing conditions and dragonfly populations. While the species I studied appeared to display species-specific responses to habitat conditions, location seemed to have a distinct influence on those responses. The issue of a unique feature, in the case of my study the presence of a hydroelectric intake/outflow facility, may potentially offset the ability of ecologists to predict any one species response to restoration changes. The finite spectrum of abiotic factors I chose to study does not represent the complexity of features that might possibly affect dragonfly nymphal density and emergence distance, and the impact of any one factor may vary considerably between spatial and temporal scales.

Regardless of which method of restoration is implemented, it is imperative that ecologists focus not only on the whole system outcomes of restoration, but also on individual components (both abiotic and biotic) of the river system. It is the connections between these myriad components that form the basis of a healthy (in terms of structure and function) riverbank ecosystem. In their quest to create restored systems quickly, ecologists risk devaluing, or even missing, integral components. Inclusion of these components and their "health" may ultimately determine whether or not a restored system continues to function in a sustainable manner.

Ultimately, the strength of frameworks such as the WFD, the OHI, or the EMAP relies on their focus on cross-scale analysis. Dragonfly bioindication can provide the ideal tool in assessing restoration impacts, as long as sufficient data are available on all stages in their life cycle. As my results suggest, the relatively short emergence and eclosure stage can be influenced by a variety of aquatic and terrestrial factors. By incorporating the consideration of each stage of the dragonfly life cycle, and by investigating species-specific and location-specific responses to habitat, the usefulness of odonates as bioindicators will be strengthened.

The emergence zone: response of nymphal dragonflies to changes in riverbank habitat

Due to their unique life history strategies, dragonflies present the ideal lens through which ecologists can view the impact of habitat change. While the majority of the available literature focuses on the impact of temperature and substrate on the nymphal stage, research has extended into the impact of these factors on adult dragonflies as well (McGeoch and Samways 1991). Unfortunately, there has been little focus on the transitional emergence stage. This stage represents a vital connection between the aquatic nymphal stage and the aerial adult stage.

Riverbank ecotone habitats are composed of a combination of both aquatic and terrestrial characteristics and are frequently influenced by human activities. Adjacent forest cover and land use determine the geomorphic evolution of the riverbank, and can create the potential for erosion (Weins 2002). Aquatic characteristics also impact the riverbank ecotone. For example, seasonal flooding and altered hydroperiods (e.g., due to dams and reservoir releases), affect sediment deposition and transport, can lead to increased erosion of riverbanks, and ultimately may modify habitats for benthic macroinvertebrates (McClelland and Brusven 1980, Naiman and Dēcamps 1997, Magilligan and Nislow 2001). Recreational boat activity, and subsequent generation of boat wakes, adds additional stress to the riverbank ecotone (Schorr 2000).

Riverbank ecotones are typically highly complex areas with a variety of emergent and established vegetation, alluvial deposits, and woody debris, and provide refuge for many species of insects (van Looy et al. 2005). The interface zone between two landscape types, termed an edge, has long been the focus of researchers (Samways and Steytler 1996, Urbine-Cardona 2006). Edges provide unique habitats that often are not found in either of the adjacent areas. While the importance of terrestrial edges has been well documented, little focus has been placed on the unique edge that exists between aquatic and terrestrial landscapes (Homan et al. 2004, Gamble et al. 2006). In river systems, the riverbank ecotone comprises this edge and represents the final hurdle that riverine odonates must cross before they emerge as adults.

The placement of rocks (riprap) along the lower edge of riverbanks is commonly used to control bank erosion and restore the structural stability of the slope. Typically riprap slope stabilization is applied at the lowest section of the bank, and extends into the water line. While riprap does control rate of erosion, it creates highly variable, potentially hazardous ecotones for dragonflies. Characteristics of this ecotone include an extensive interstitial space (which may provide refugia for the nymphs), relative substrate stability, and permanent exposure to wave action.

The exposed rock toe may also create a thermal barrier to eclosing nymphs. Although research has been conducted on the colonization of these areas by plants and invertebrates

(Tockner 1991), little attention has been centered on the important role this biotope plays on species that must transverse it. Presence of riprap can alter morphologic evolution of the river through natural changes in energy flow (organic nutrients), physical characteristics, and plant succession (Fischenich 2003), and the size of the riprap alters habitat for many fish species (Lister et al. 1995, Beamer and Henderson 1998). Riprap can also affect the hydrologic balance causing changes in river slope or profile, barriers between surface, subsurface, and benthic waters, or alteration of flow (Fischenich 2003). The riprap can also absorb heat causing micohabitat changes in temperature. For both *G. vastus* and *S. spiniceps* the presence of riprap strikingly reduced the distance they traveled to their eclosure location. By eclosing closer to the water, both species are at increased risk of being drowned by rising water levels, or boat wakes.

Recently, there has been an interest in tracking the effects of climate change, most notably change in ambient air temperature, on populations of dragonflies. Work by Braune et al. (2008), suggests that increasing air temperatures may potentially cause a change in the voltinism, or number of generations produced in one year. Braune et al. (2008) also suggests that with increasing ambient temperatures the phenology of higher altitudinal species will change more rapidly than lower altitudinal species. Although many species have been shown to exhibit faster cycles as the result of temperature increases (Root et al. 2003), at least one species of dragonfly, *Orthetrum albistylum speciosum*, has demonstrated delayed emergence as the result of increases in ambient temperature (Doi 2008). Air temperature has also been demonstrated to have an effect on nymphal development rates (Pritchard 1989), and adult activity patterns (De Marco and Peixoto 2004).

The effect of water temperature on instream nymphal density and distribution has been well documented, but its effect on nymphal emergence has only recently been addressed (Hassall et al. 2007, Richter et al. 2008). In a long term study of flight season (a reflection of nymphal emergence) in Britain, Hassall et al. (2007) documented a three day increase in flight season per 1^{o} C increase in water temperature. As water temperatures increased, nymphal emergence occurred earlier in the season. This disjunction of emergence timing and prey emergence could potentially limit the availability of sufficient food sources for the newly emerged adults. This finding has subsequently been supported by Richter et al. (2008) who studied the impact of increasing water temperatures on emergence onset in *Gomphus vulgatissimus*. Emergence began fourteen days earlier than normal when water temperature was increased by 2^{o}C, and thirty-six days when the temperature was increased by 6^{o}C (Richter et al. 2008).

Size of near-bank sediment has also been linked to presence or absence of certain species of dragonfly nymphs (Marczak et al. 2006). Nymphs of *Cordulegaster dorsalis* were more abundant in zones that had sediment between 1.5mm, and 2.5mm (Marczak et al. 2006), while nymphs of *Gomphus flavipes* can be found in a sediment ranging from 0.07 – 4.76mm (Galetti and Ravizza 1977).

Although the effect of near shore water velocity on nymphal density has not been well documented, water velocity has been identified as an important factor in determining instream nymphal survival and distribution (Carchini and Rota 1985, Hoffmann and Mason 2005). In a study of a modified river in England, Hofmann and Mason (2005) recorded that *Calopteryx splendens* nymphs were tolerant of high velocities, while nymphs of *Libellula folia* and *Platycnemis pennipes* were restricted to areas of moderate velocities.

How bank stabilization impacts dragonflies

The aim of restoration ecology is to restore damaged systems to their natural state (Ewel 1987, Hull and Robertson 2000) but in reservoir systems, where “natural” banks are often highly eroded, the goal of restoration may be modified; restoration goals may instead focus on one particular aspect (i.e. bank stability) of the system. Determination of what constitutes a natural, or “ecologically-stable” state depends upon the ability of ecologists to define the physical and ecological processes present in the system (Hull and Robertson 2000). In the Turners Falls Reservoir system emphasis has historically been placed on physically stable riverbanks that have the greatest capacity to minimize bank erosion. Over the years, stabilization has involved placement of large boulders (which although successful in achieving the stated goal, are not well-suited to mimicking a “natural” system), various size riprap, and most recently vegetation-only bank stabilization methods (which have not been highly successful at minimizing bank erosion).

The United States Army Corps of Engineers recognizes four basic stabilization methods using riprap: (1) armor, (2) flow deflection, (3) slope stabilization and (4) energy reduction (Fischenich 2003). Armoring involves the placement of large stones, which completely cover the surface of the riverbank, while flow deflection involves the creation of stone jetties that protrude into the river and deflect the force of the water away from the bank (Fischenich 2003). Slope stabilization techniques typically involve placement of riprap along the bank edge (toe). Energy reduction methods incorporate a variety of techniques that are focused on reducing the energy and erosional capability of the river.

While all four categories of stabilization are applied along the Connecticut River, slope stabilization is the most prevalent method in the Turners Falls Reservoir, and may have the greatest potential impact on emerging dragonflies (McLain et al. 2004, 2006).

The placement of rocks (riprap) along the lower edge of riverbanks is commonly used as a restoration technique to control bank erosion. Typically riprap slope stabilization is applied at the lowest section of the bank, and extended into the water line. While riprap does control rate of erosion, it also creates a unique, though not necessarily beneficial, ecotone for dragonflies. While this ecotone may provide a source of beneficial refugia for the nymphs, through increased interstitial space, the riprap zone (within which many nymph species eclose), is subject to extensive wave action.

The exposed rock toes may also create a thermal barrier to eclosing nymphs. Although research has been conducted on the colonization of these areas by plants and invertebrates (Tockner 1991), little attention has been centered on the important role this biotope plays on species that must transverse it. Presence of riprap can alter morphologic evolution of the river through natural changes in energy flow (organic nutrients), physical characteristics, and plant succession (Fischenich 2003). The size of the riprap alters habitat for many fish species (Lister et al. 1995, Beamer and Henderson 1998). Riprap can also affect the hydrologic balance causing changes in river slope or profile, barriers between surface, subsurface, and benthic waters, or alteration of flow (Fischenich 2003). The riprap can absorb heat, causing changes in micohabitat temperature. Vegetation is often unable to successfully recolonize riprapped areas.

In an effort to mitigate some of these negative effects, restoration managers have begun to integrate vegetation into their restoration plans. In the Turners Falls Reservoir system of the Connecticut River, several riprap and vegetative bank stabilization methods have been applied with varying success.

Regardless of which type of bank restoration is ultimately implemented, it is imperative that ecologists focus not only on the whole system outcomes of restoration, but also on individual components (both abiotic and biotic) of the river system. It is the connections between these myriad components that form the basis of a healthy (in terms of structure and function) riverbank ecosystem. In their quest to create restored systems quickly, ecologists or resource practitioners risk devaluing, or even missing, integral components. Inclusion of these components and their "health" may ultimately determine whether or not a restored system continues to function in a sustainable manner.

Summary of prior research in the Turners Falls Reservoir

While the overall body of literature on emergent nymphal density is sparse, the Turners Falls Reservoir has been the site of a long-term monitoring project focused on the impacts of restoration on nymphal density. Beginning in 2001, a team of researchers (McLain et al. 2004, 2006, McLain 2008), began tracking the density of several species (Table 4) at sites that were scheduled to be stabilized. In subsequent years, the team returned to the sites to record species response to the restoration. While their data seem to suggest the potential benefits of stream bank stabilization (restoration) on nymphal density, McLain (2008) is quick to point out that they observed extreme fluctuations in nymphal densities from year to year.

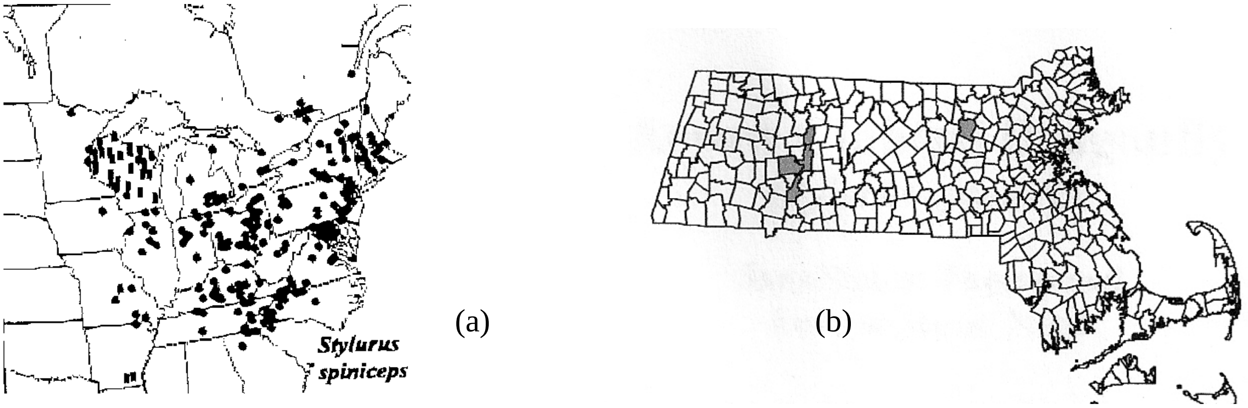
Table 4. Mean exuvia density per 30.48m at selected pre-stabilization and post-stabilization sites within the Turners Falls Reservoir (modified from McLain 2008).

	Pre-stabilization (2001)	Post-stabilization (2002-2008)
G. vastus	30	375
N. yamaskanensis	3	17
S. spiniceps	4	9
M. illinoiensis	3	35
D. spinosus	3	5

Selected species of the Turners Falls Reservoir

In my study I chose to track the densities of seven species which are found within the Turners Falls Reservoir of the Connecticut River in Gill, MA. Species chosen for this study were selected based on two criteria: (1) density (the most frequently encountered species recorded during a 2006-2007 pilot study of the area), and (2) Massachusetts Endangered Species Act listing status (MESA).

Stylurus spiniceps is listed as listed as a threatened species under MESA. Nymphs can first be seen to emerge from rivers beginning in late June or early July (Nikula et al. 2003). *S. spiniceps* has historically been found in the Lawrence and Concord rivers, but recent records indicate their presence in the Sunderland, MA – Deerfield, MA section of the Connecticut River (Fig. 3).

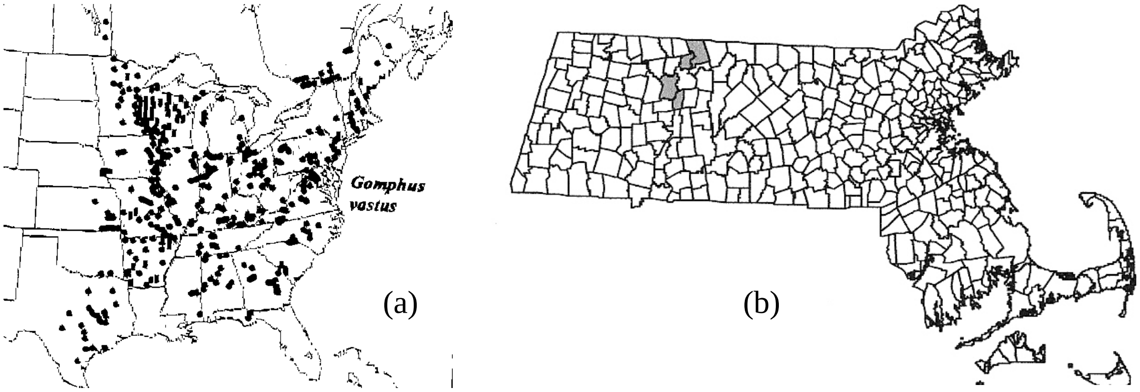


(a)

(b)

Figure 3. Distribution of *S. spiniceps* in (a) the United States (modified from Donnelly 2004) and (b) Massachusetts (modified from MA Natural Heritage and Endangered Species database) shaded polygons indicate presence of species.

Gomphus vastus is listed as a species of special concern under MESA. Historic records have shown populations of *G. vastus* in the Merrimack River, but more recent records indicate their presence in the Connecticut River (Fig 4). Nymphs begin emerging in early June and typically continue through mid-late July (Nikula et al. 2003).



(a)

(b)

Figure 4. Distribution of *G.vastus* in (a) the United States (modified from Donnelly 2004) and (b) Massachusetts (modified from MA Natural Heritage and Endangered Species database) shaded polygons indicate presence of species.

Neurocordulia yamaskanensis is also listed as species of special concern under MESA. Currently, the only known location of *N. yamaskanensis* in Massachusetts is the Connecticut River (Fig 5).

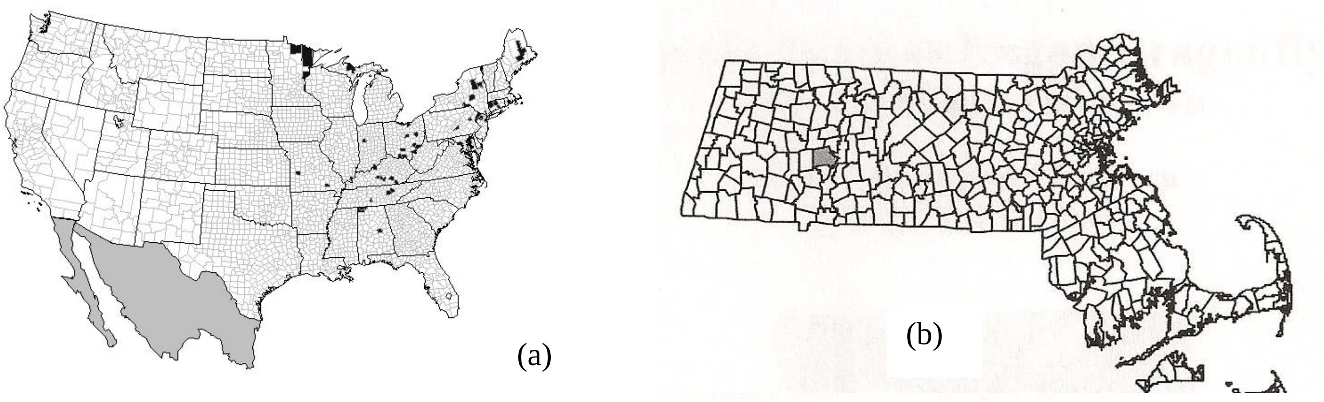
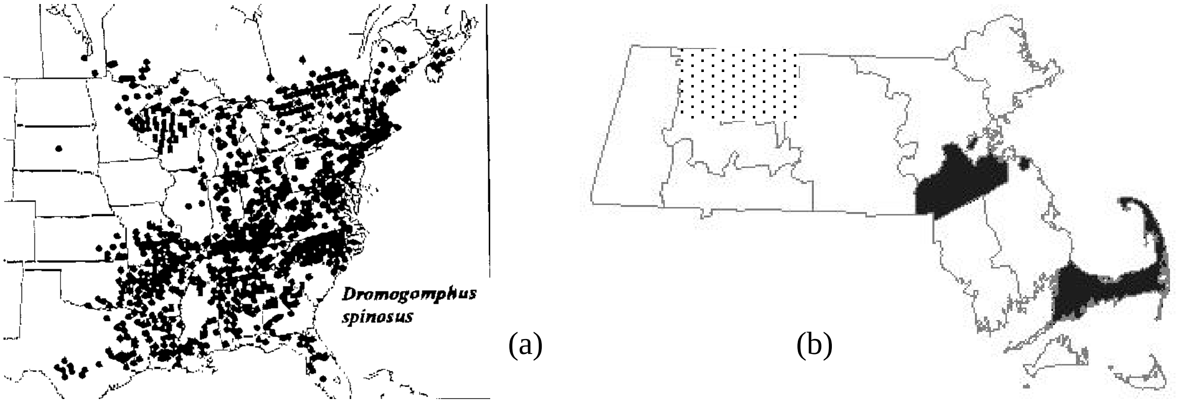


Figure 5. Distribution of *N. yamaskanensis* in (a) the United States (modified from Kondratieff, 2000), and (b) Massachusetts (modified from MA Natural Heritage and Endangered Species database) shaded polygons indicate presence of species.

Dromogomphus spinosus, *Ophiogomphus rupinsulensis*, *Macromia illinoiensis*, and *Epitheca princeps* are not currently listed under MESA. *D. spinosus* is commonly found throughout Massachusetts (Fig 6), and is typically found in slow moving streams and rivers (Nikula et al. 2003). Nymphs begin emerging in mid June and continue through late August (Nikula et al. 2003).



(a) (b)

Figure 6. Distribution of *D. spinosus* in (a) the United States (modified from Donnelly 2004) and (b) Massachusetts (modified from MA Natural Heritage and Endangered Species database) shaded polygons indicate presence of species, dots represent species accounts from McLain and Morrison (2004, 2006), Martin (2007), and McLain (2008)

Ophiogomphus rupinsulensis is found throughout the Northeastern United States (Fig 7); in Massachusetts it is found in small numbers in the northeastern, central, and western parts of the state (Nikula et al. 2003). Nymphs typically begin emerging in early June and continue through late July (Nikula et al. 2003).

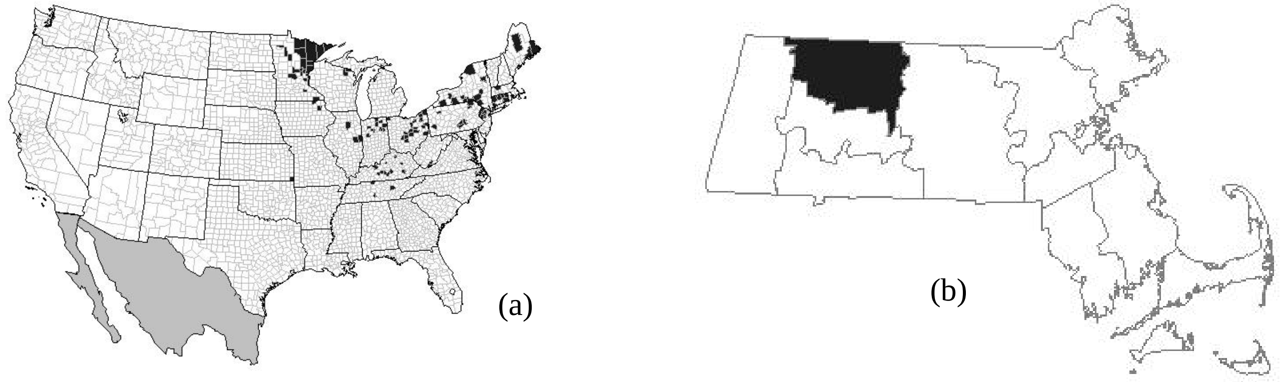


Figure 7. Distribution of *O. rupinsulensis* in (a) the United States, and (b) Massachusetts (modified from Kondratieff, 2000) shaded polygons indicate presence of species.

Macromia illinoiensis is found fairly commonly throughout the Eastern United States, and typically is present in lake or river systems (Fig 8). Nymphs of *M. illinoiensis* begin emerging in early June, and continue emergence until late August or early September (Nikula et al. 2003). Although Kondratueff (2000) (Fig 7), does not show the presence of *M. illinoiensis* in Connecticut River, exuviae were frequently collected by McLain and Morrison (2004, 2006), and McLain (2008).

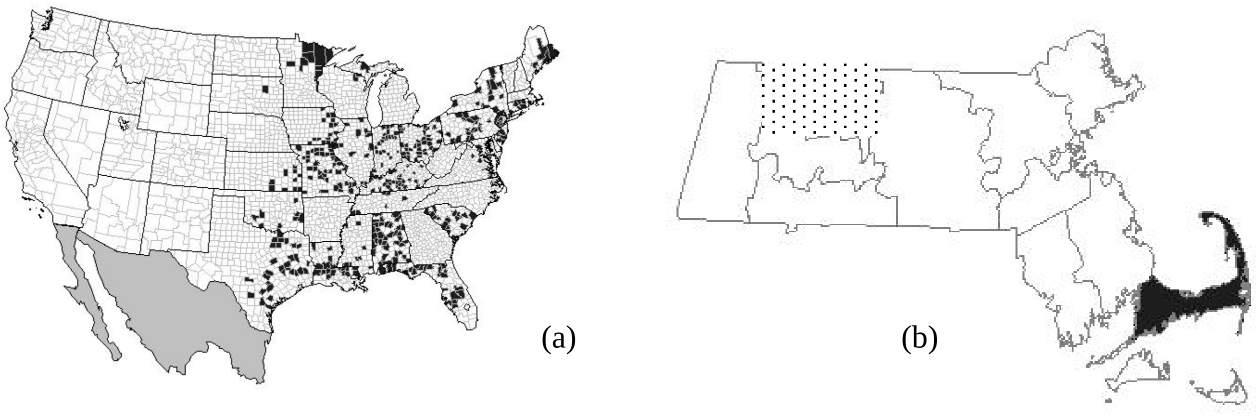


Figure 8. Distribution of *M. illinoiensis* in (a) the United States and (b) Massachusetts (modified from Kondratueff 2000) shaded polygons indicate presence of species, dots represent species accounts from McLain and Morrison (2004, 2006), Martin (2007), and McLain (2008).

E. princeps is found commonly in several counties within Massachusetts (Fig 9), and can be found along quiet rivers, lakes, and ponds (Nikula et al. 2003). McLain and Morrison (2004, 2006), and McLain (2008) reported finding *E. princeps* within the Turners Falls Reservoir. Nymphs typically emerge beginning in early July, and emergence can continue until mid-August (Nikula et al. 2003).

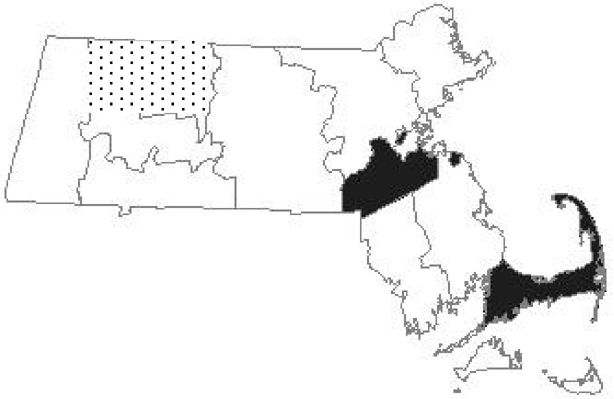


Figure 9. Distribution of *E. princeps* in Massachusetts (modified from Kondratueff 2000) shaded polygons indicate presence of species, dots represent species accounts from McLain and Morrison (2004, 2006), Martin (2007), and McLain (2008).

Site selection

Restoration measures, particularly within those areas that are downstream of reservoirs created for hydropower production, have been demonstrated to have significant effects on sedimentation, nutrient flow, and benthic habitat (Petts 1984, Johansson and Nilsson 2002, Campbell and Novelo-Gutierrez 2007, Fette et al. 2007). For this reason I chose to conduct my research within the Turners Falls Reservoir (Gill, MA: Franklin Co.) section of the Connecticut River (Fig 10).

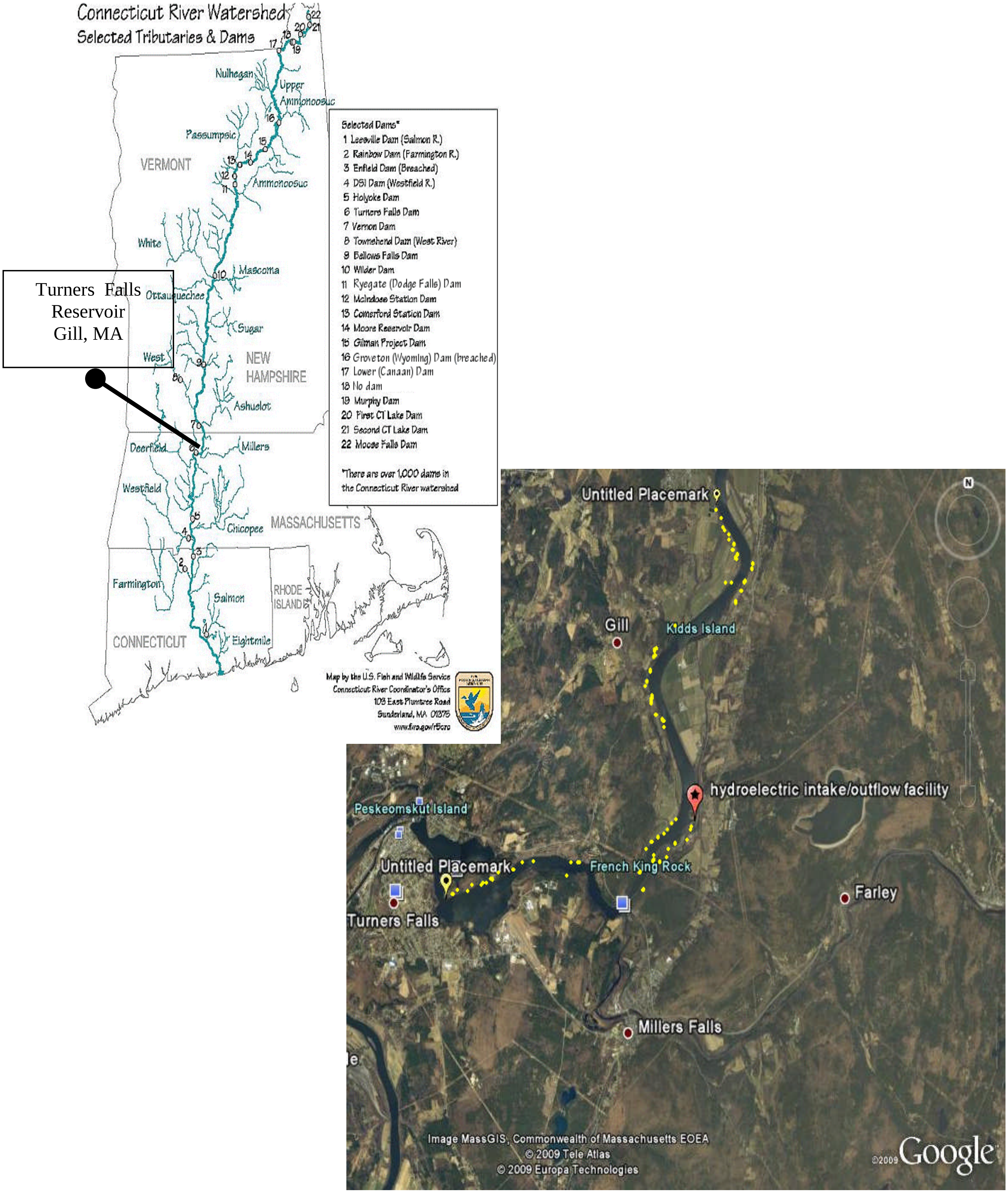


Figure 10. Connecticut River, Turners Falls Reservoir. Gill, Ma: Franklin Co., dots represent relative transect locations.

The Turners Falls Reservoir provides the ideal location to observe and document multiple stressors on emerging dragonfly nymphs. The reservoir is an ideal living laboratory, because it has well defined boundaries (Vernon, VT dam to the north and Turners Falls, MA dam to the south). Since this section of the Connecticut River functions as a reservoir for hydroelectric generation, water levels fluctuate frequently throughout the day. Due to the differences in water release and uptake at the Northfield Mountain Pump Storage facility (location has been designated as intake/outflow in this study), water level fluctuations in the Turners Falls Reservoir average as much as 1.07m per day (U.S. Army Corps 1991). Discharge data (ft^3/sec) for the time period of the study was obtained from the USGS gauge station 01170500 located on the Connecticut River at Montague City, MA: Franklin Co. (Fig. 11).. During the period of the study, water velocity was relatively stable, with the exception of two time periods, 21 July – 30 July, and 2 August-15 August, where discharge rates were higher than average (16,911ft^3/sec) (Fig. 11).

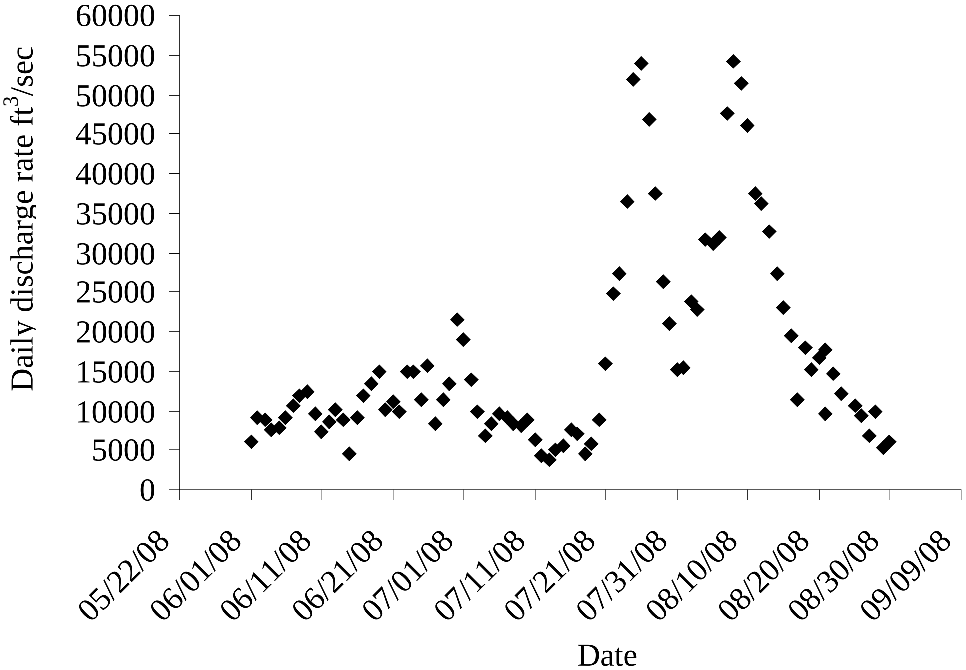


Figure 11. Daily discharge (ft^3/sec) rates for Turners Falls Reservoir area of the Connecticut River, data from USGS stream gauge 01170500 (USGS Water Resources)

Near-shore water velocities are also highly variable. Water level changes and boat activities also increase bank erosion, and sedimentation flux in near-shore areas. In 1991, an extensive review of streambank conditions along the Connecticut River in New Hampshire and Massachusetts revealed that 32% of the shoreline within the Turners Falls Reservoir was in a state of active erosion (U.S. Army Corps 1991). Riprap has been extensively used to stabilize banks, but many shoreline slopes remain severely undercut.

The Connecticut River is the main hydrological feature of the Turners Falls Reservoir, but since the construction of the Turners Falls dam (1904), Vernon, VT dam (1910) and the Northfield Mountain Pump Storage Project (1972), the reservoir can be considered as a managed system. A variety of streambank restoration methods (from concrete blocks to large boulders) were implemented soon after the hydroelectric facility began operations.

Methods

At the Turners Falls Reservoir, the Northfield Mountain Pump Storage intake/outflow facility designated the center of my study area. Fifty transects (25 control sites (eroded), and 25 riprap sites) were located upriver and fifty transects (25 control sites (eroded), and 25 riprap sites) were located downriver of this location. A random numbers table was used to establish the distance from the hydroelectric intake/outflow facility to the location of the furthest upriver and downriver transect. Each transect covered 1.5m (5ft) of shoreline and extended 12.19m (40ft) vertically up the slope of the bank (Fig 12).

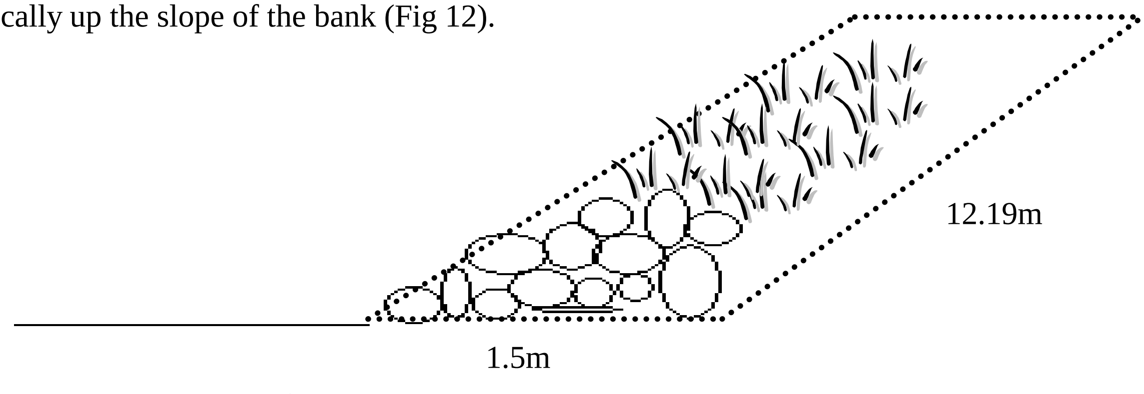


Figure 12. Depiction of survey transect.

This distance is slightly longer than the maximum observed emergence distance recorded by McLain et al. (2004), McLain et al. (2006) and in my 2006 pilot study (Martin 2006a).

Distances to subsequent sampling locations were also determined using a random numbers generator. Three numbers that were randomly generated were multiplied together in order to determine the distance from the original transact to the next location. Each additional transact was established in the same manner. Odd numbers resulted in transects being located on the left bank (facing downriver), while even numbers resulted in transects being established on the right bank (facing downriver). Each transact was visited at least twice a week beginning on June 5th, 2008, and ending on August 14th, 2008. Sites were visited between the hours of 10:00AM-3:00PM.

Physical parameters

Slope was determined from the waterside edge of the riprap toe to 40ft (12.19m) up the bank. Three river-bed sediment samples were taken 2ft (0.61m) beyond the edge of the bank; these samples were combined and percent composition of sand, silt, and clay was determined. The width of the riprap toe (stabilization using rocks) was recorded, and individual riprap circumference (cm) was ascertained by measuring 100 random samples with a Wildco (95 Botsford Place, Buffalo, NY) gravelometer (Potyondy and Hardy 1993, Bunte and Abt 2001). Water, air, and substrate surface temperature were recorded at the beginning of each sampling session using a digital thermometer. Water level was recorded using a staff gauge (USGS), which was placed vertically in the water 1.22 m (4ft) from the shoreline. Water level was recorded at ten-minute intervals throughout the monitoring period. Water velocity was recorded 1.22m (4ft) from the shoreline using a Wildco flowmeter.

Species density and emergence distance

Species density per transect (60.98m^2 (200ft^2)) for *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, *E. princeps*, and the distance traveled from the edge of the water to the eclosure site for *G. vastus* and *S. spiniceps* was determined through either (1) direct observation of eclosing nymphs, or (2) presence of attached exuviae (Chovanec and Waringer 2001, Corbet 2004, Foster and Soluk 2004).

Specimens were stored in individually labeled 250ml containers (one container per site per sampling session), preserved in 50% isopropyl alcohol, and labeled with the site name, date, and time. Specimens were identified to the species level using a northeastern Anisoptera key (Soltesz 2002).

Statistical analysis

Analysis was divided into three parts, (1) comparison of abiotic features and location, (2) species density, and (3) emergence distance. The effect of location on abiotic factors was analyzed using the Mann-Whitney U test. The influence of location on species density was also compared using Mann-Whitney U. Mann- Whitney U is a non-parametric test that does not require data to follow a normal distribution. Further analysis of the correlation between species density and abiotic factors was conducted using Pearson correlation. The Pearson correlation measures the degree to which two variables are associated. The strongest correlations result in a value of +/-1.

Initial analysis of the effects of both location on *G. vastus* and *S. spiniceps* emergence distance was conducted using Mann-Whitney U. Pearson correlation was used to highlight connections between location, abiotic features and emergence distance.

In order to more clearly encompass the complexities of the Turners Falls Reservoir, backwards elimination multiple regression (BEMR) was conducted on the relationships between (1) location, abiotic factors, and *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. princeps* densities, and (2) location, abiotic factors, and emergence distance of *G. vastus* and *S. spiniceps*. The general goal of multiple regression analysis is to predict which independent variables (abiotic factors) affect the dependent variable (density and emergence distance). BEMR removes the least significant independent factor(s) each time the regression is run. This results in a model that reflects the independent variables which have the highest level of significance. BEMR analysis resulted in the generation of several significant models. One "best-fit" model was chosen based on the greatest number of significant contributions by individual abiotic factors and the highest possible r^2 value. For the density analysis, r^2 values>0.50 were considered to be biologically significant. Due to the small scale effects of habitat change on emergence behavior all r^2 values were considered to be biologically significant.

Observations and hypothesis

Based on my observations of nymphal dragonflies within the Turners Falls Reservoir from 2006, and species habitat preference data provided by Corbet (2004) I have made several predictions as to how dragonfly density and emergence distance will be affected by location and selected abiotic features.

Density hypothesis one: Density of *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. princeps* will not be affected by location (upriver versus downriver).

<u>Density hypothesis two</u>: Density of *S. spiniceps*, *G. vastus, N. yamaskanensis, D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. princeps* will not be affected by air temperature, water temperature, substrate temperature, or slope.

<u>Density hypothesis three</u>: Density of *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis, M. illinoiensis*, and *E. princeps* will be influenced by water velocity, water level change, presence of riprap, width of the riprap zone, circumference of the rocks within the riprap zone, and composition of sediment (% sand, % silt, % clay).

<u>Emergence hypothesis one:</u> Location (upriver or downriver) will have no effect on *G. vastus or S. spiniceps* travel distance.

<u>Emergence hypothesis two</u>: *G. vastus* and *S. spiniceps* emergence distance will not be affected by air, water, or substrate temperature.

<u>Emergence hypothesis three</u>: *G. vastus* and *S. spiniceps* emergence distance will be affected by water velocity, water level change, presence of riprap, width of the riprap zone, circumference of the rocks within the riprap zone, and composition of sediment (% sand, % silt, % clay).

Results of density analysis

Mann-Whitney U analysis showed significant ($p<0.05$) differences between several abiotic features at upriver versus downriver locations (Table 5). Air temperature was an average of 6.62 degrees (F) warmer at the downriver locations (below the intake/outflow facility) as compared to the upriver sites (above the intake/outflow facility) ($p<0.05$). Substrate temperature was also warmer at downriver sites ($p<0.05$) (Table 5). Water velocity (m/sec) was slightly slower at the upriver locations (mean= 1.59, sd=0.50) as compared to the downriver locations (mean= 1.63, sd= 0.37) ($p<0.05$). The riprap zone was significantly wider ($p<0.05$) at the downriver locations (mean= 1.20, sd= 1.62) than the upriver sites (mean= 1.48, sd= 1.48). Circumference of the rocks within the riprap zone varied considerably at both the upriver and downriver locations (Table 5), with downriver sites having slightly larger rocks. The composition of sediment located in the near riverbank zone differed between upriver and downriver locations ($p<0.05$) (Table 5). The percentage of sand found within the zone was 6.17% higher at the downriver locations, while both percentage of silt and percentage of clay were higher at the upriver locations (Table 5).

Table 5. Mann-Whitney U analysis of abiotic features present at upriver (n=50) and downriver (n=50) locations. Significant associations ($p<0.05$) are marked with*, standard deviations are given in parenthesis.

Abiotic feature	Upriver	Downriver
Air temperature*	68.12 (6.22)	74.74 (4.76)
Water temperature	71.26 (3.96)	72.26 (5.15)
Substrate temperature*	71.39 (4.17)	73.24 (4.46)
Water velocity (m/sec)*	1.59 (0.50)	1.63 (0.37)
Water level change (cm)	3.41 (2.29)	3.69 (3.25)
% slope	32.86 (8.15)	33.13 (5.77)
Width of riprap (m)*	1.20 (1.63)	1.48 (1.48)
Circumference of riprap (cm)*	44.99 (56.32)	63.18 (56.53)
% sand*	34.07 (11.67)	40.24 (9.04)
% silt*	36.61 (7.46)	32.63 (4.33)
% clay*	29.07 (5.98)	28.36 (24.62)

Higher densities of *S. spiniceps, G. vastus, M. illinoiensis,* and *E. princeps* were found on non-riprapped sites ($p>0.05$), while *N. yamaskanensis*, *D. spinosus,* and *O. rupinsulensis* densities were higher on riprapped sites ($p>0.05$) (Table 6). No significant associations were found between absence or presence of riprap and densities of *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus, O. rupinsulensis, M. illinoiensis,* and *E. princeps* (Table 6). .

Table 6. Mann-U analysis of densities per 60.98m^2 of *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus, O. rupinsulensis, M. illinoiensis,* and *E. princeps* at non-riprapped versus riprapped riverbanks, standard deviations are given in parenthesis. No significant relationships were found.

Species	Non-riprap	Riprap
S. spiniceps	3.47(3.85)	3.09(2.50)
G. vastus	4.27(3.77)	3.85(4.40)
N. yamaskanensis	1.64(0.93)	2.53(2.10)
D. spinosus	1.69(1.18)	2.36(1.21)
O. rupinsulensis	1.45(0.69)	1.75(1.16)
M. illinoiensis	2.17(1.87)	1.47(0.87)
E. princeps	2.18(1.53)	1.78(1.17)

There were no significant differences in density of *S. spiniceps, N. yamaskanensis, D. spinosus, O. rupinsulensis*, or *E. princeps* at upriver versus downriver locations. *G. vastus* was found in significantly higher densities (299) at upriver sites than downriver sites (215) (Table 7), while *M. illinoiensis* was found in higher densities at downriver sites (35).

Table 7. Mann-Whitney U analysis of densities per 60.98m^2 of *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus, O. rupinsulensis, M. illinoiensis,* and *E. princeps* at upriver (n=50) and downriver locations (n=50). Significant associations (p<0.05) are marked with*, standard deviations are given in parenthesis.

Species	Upriver		Downriver	
	Total	Mean	Total	Mean
S. spiniceps	83	3 (3.45)	127	3 (3.11)
*G. vastus**	299	5 (3.84)	215	3 (4.33)
N. yamaskanensis	34	2 (2)	32	2 (1.33)
D. spinosus	36	2 (1.27)	32	2 (1.33)
O. rupinsulensis	15	2 (0.97)	15	2 (0.87)
*M. illinoiensis**	29	2 (1.75)	35	2 (1.20)
E. princeps	42	2 (1.71)	38	2 (0.98)

Pearson correlations conducted on the influence of individual abiotic factors on the seven selected species resulted in several significant ($p<0.05$) differences between upriver and downriver locations (Table 8).

Table 8. Pearson correlation on the significant ($p<0.05$) influences of selected abiotic factors at upriver ($n=50$) versus downriver ($n=50$) locations for density per $60.98 m^2$ of *S. spiniceps, G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, and *M. illinoiensis*. No significant correlations were found for *E. princeps*, --- represents no significant correlation.

Abiotic factors	Species	Upriver	Downriver
Air temperature	*O. rupinsulensis*	0.95	---
	M. illinoiensis	-0.44	---
	N. yamaskanensis	---	0.47
Water temperature	*O. rupinsulensis*	0.99	---
	M. illinoiensis	-0.59	---
Substrate temperature	*G. vastus*	-0.24	---
Water velocity	*M. illinoiensis*	0.70	0.62
Water level change (cm)	*S. spiniceps*	---	0.29
	O. rupinsulensis	-0.95	0.62
	M. illinoiensis	-0.64	---
Slope	*S. spiniceps*	-0.35	---
	D. spinosus	0.49	---
Width of riprap (m)	*O. rupinsulensis*	-0.87	---
Circumference of riprap (cm)	*O. rupinsulensis*	0.925	---
% sand	*S. spiniceps*	-0.48	---
	G. vastus	-0.28	-0.29
% silt	*S. spiniceps*	0.43	---
	G. vastus	0.30	---
% clay	*S. spiniceps*	0.48	---
	N. yamaskanensis	0.44	---
	M. illinoiensis	-0.49	---

Temperature

The density of three species, *N. yamaskanensis*, *O. rupinsulensis* and *M. illinoiensis*, was significantly ($p<0.05$) correlated with air temperature (Table 8). Models suggest that *N. yamaskanensis* density increases with each one-degree (F) increase in air temperature. *O. rupinsulensis* density at upriver sites was positively related to air temperature, while density of

M. illinoiensis was negatively related to air temperature. Although the correlations of *O. rupinsulensis*, *M. illinoiensis*, and *N. yamaskanensis* densities to air temperature were statistically significant ($p<0.05$), the resulting correlation coefficient (r^2) (Table 9) were not strong enough to represent biological significance ($r^2 >0.50$).

Table 9. Correlation coefficients (r^2) of *O. rupinsulensis*, *M. illinoiensis*, and *G. vastus* to temperature ($p<0.05$).

		Upriver
Air temperature	*O. rupinsulensis*	0.002
	M. illinoiensis	0.009
	N. yamaskanensis	0.222
Water temperature	*O. rupinsulensis*	0.002
	M. illinoiensis	0.002
Substrate temperature	*G. vastus*	0.0001

O. rupinsulensis and *M. illinoiensis* were significantly correlated ($p<0.05$) with water temperature within the emergence zone. As with air temperature, *O. rupinsulensis* density increased with each 1^0 F increase in water temperature, while *M. illinoiensis* density decreased (Table 4), but as with air temperature, correlations between density of *O. rupinsulensis* and *M. illinoiensis* were not strong, and do not adequately reflect biological significance (Table 5).

The only significant correlation between substrate temperature and nymphal density was found with *G. vastus* density at upriver locations. For every 1^0 F increase in substrate temperature, *G. vastus* decreased (Table 4). This correlation, however, is extremely weak ($r^2<0.50$) (Table 5), and cannot be considered as a strong representation of what is occurring within the system.

Water velocity and level

M. illinoiensis was positively correlated with water velocity (m/sec) at both the upriver and downriver locations, although density was negatively correlated with change in water level at the upriver locations (Table 8). *O. rupinsulensis* density was negatively correlated with changes in water level above the intake/outtake facility, but displayed a positive correlation at the downriver locations (Table 8). *S. spiniceps* density was also positively correlated with changes in water level at the downriver locations (Table 8). Although all of these correlations were statistically significant ($p<0.05$), their correlation coefficients were not sufficient ($r^2<0.50$) to illustrate biological significance (Table 10).

Table 10. Correlation coefficients (r^2) of O. rupinsulensis, *M. illinoiensis*, and *G. vastus* to water velocity and water level change ($p<0.05$), --- represents no significant correlation.

		Upriver	Downriver
Water velocity (m/sec)	*M. illinoiensis*	0.125	0.484
Water level change (cm)	*S. spiniceps*	0.081	---
	O. rupinsulensis	0.008	0.028
	M. illinoiensis	0.064	---

Riverbank features

S. spiniceps density at the upriver sites decreased as slope of the riverbank increased, while *D. spinosus* density displayed a positive correlation with percent slope (Table 8). Both correlations resulted in weak r^2 values, and are not strong indicators of biological significance.

The width of the riprap zone (m) and circumference of the rocks within the riprap zone (cm) were significantly correlated ($p<0.05$) with density of *O. rupinsulensis* at both upriver and downriver locations. An increase in the width of the riprap zone resulted in lower *O.*

rupinsulensis densities at the upriver locations (Fig 13). Downriver *O. rupinsulensis* densities also displayed a negative correlation with increased riprap zone width, but the association is based on an extremely small sample size (n=2). Rock circumference also had a significant effect on *O. rupinsulensis* densities; larger rock circumferences resulted in increased densities at both upriver and down river locations (Fig 13).

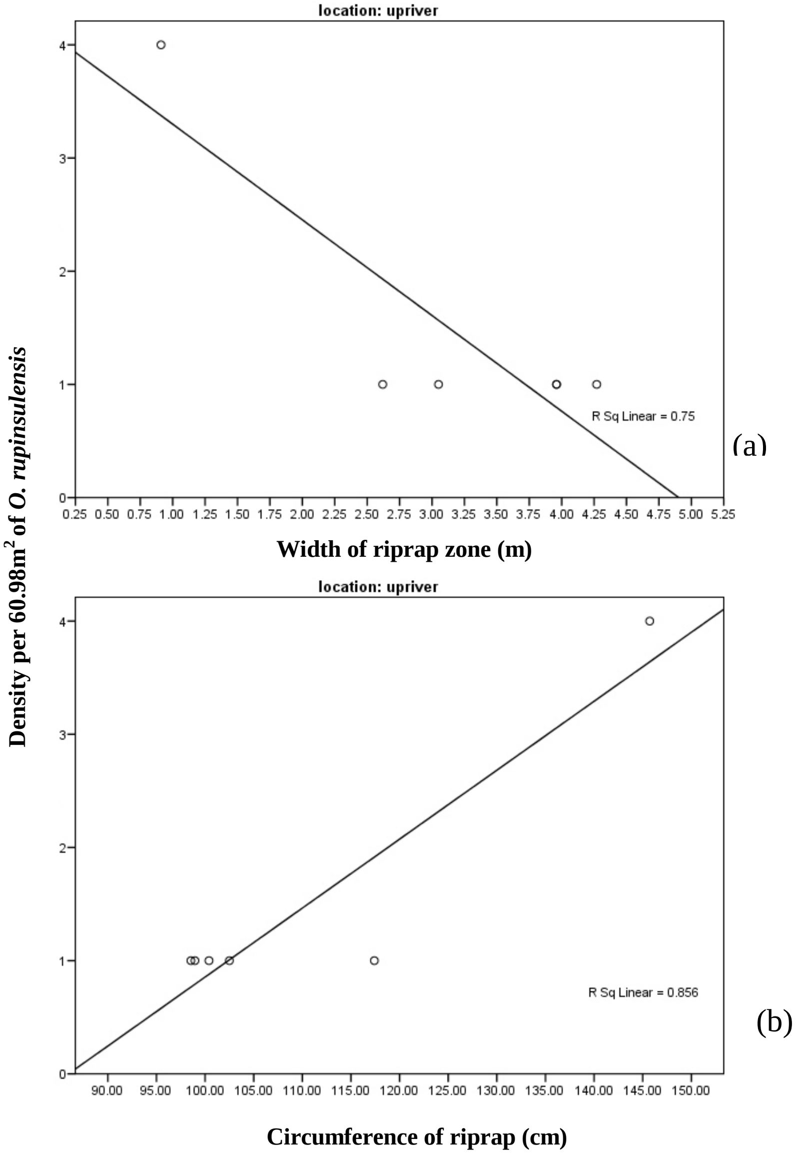


Figure 13. Pearson correlation of *O. rupinsulensis* to (a) width of riprap zone and (b) circumference of rocks within the riprap at upriver locations (Table 4).

Composition of sediment

The composition of the sediment within the emergence zone (0.61m, 2 ft) past the riprap (on the river side), was correlated with density of two species. *S. spiniceps* density displayed a strongly (r^2= 0.224) negative correlation (upriver locations only) with increasing % sand, while *G. vastus* (both upriver and downriver locations) displayed a slightly weaker negative association (Table 8). As the percentage of silt within the emergence zone increased, densities of both *S. spiniceps* and *G. vastus* increased (Table 8). Statistically significant correlations ($p<0.05$) between the percentage of clay in the sediment and nymphal density were found for *S. spiniceps*, *G. vastus*, and *M. illinoiensis*. Both *S. spiniceps* and *G. vastus* were positively correlated with the percentage of clay in the sediment, while *M. illinoiensis* displayed a negative correlation (Table 8). Since none of these correlations were very strong ($r^2<0.50$), they are not considered to be biologically significant (Table 11).

Table 11. Correlation coefficients (r^2) of *S. spiniceps, G. vastus, N*, yamaskanensis, and M. *illinoiensis* to composition of sediment ($p<0.05$), --- represents no significant correlation.

		Upriver	Downriver
% sand	*S. spiniceps*	0.224	---
	G, vastus	0.039	0.035
% silt	*S. spiniceps*	0.183	---
	G. vastus	0.062	---
% clay	*S. spiniceps*	0.228	---
	N. yamaskanensis	0.196	---
	M. illinoiensis	0.003	---

Model selection nymphal density

The Pearson correlations discussed in the previous section illustrate the effect of single abiotic factors on the density of selected dragonfly species. While they are illustrative of specific impacts within the Turners Falls Reservoir system, they do not adequately describe the complexities of the system. In an effect to more accurately describe the ecological system present in the Turners Falls Reservoir, I used backward elimination multiple regression (BEMR) to illuminate the intricate connections between multiple abiotic factors and dragonfly nymphal densities (Table 12).

Table 12. Significant models of the relationship of multiple abiotic factors to dragonfly nymphal density as generated by BEMR analysis. Significant individual contributions of abiotic factors within each model are denoted with (*)

	Upriver Features	**Standardized B**	**Downriver Features**	**Standardized B**
S. spiniceps	Air temperature*	-10.55	No significant model Mean = 3 sd = (3.11)	
	Water temperature*	0.88		
	Substrate temperature*	9.79		
	Circumference of riprap*	-0.86		
	R^2	**0.51**		
G. vastus	% silt in sediment*	0.43	% sand in sediment*	-0.29
	Substrate temperature*	-1.83		
	Water velocity*	0.34		
	Circumference of riprap	0.33		
	Air temperature*	1.65		
	R^2	**0.22**	**R^2**	**0.08**
N. yamaskanensis	Substrate temperature*	0.97	Air temperature*	5.71
	Water level change*	1.41	Water temperature*	-5.32
	Slope*	-0.96	Slope	-0.29
	% silt in sediment*	-0.57	% silt in sediment	0.44
	Width of riprap zone*	-0.74	Water velocity*	0.75
	Circumference of riprap*	1.44	Water level change*	0.72
	R^2	**0.89**	**R^2**	**0.92**
D. spinosus	% slope*	0.49	No significant model Mean = 2 sd = 1.33	
	R^2	**0.24**		
O. rupinsulensis	No significant model Mean = 2 sd = 0.97		Circumference of riprap	-0.87
			% clay in sediment*	-1.50
			% sand in sediment*	-1.51
			R^2	**0.87**
M. illinoiensis	Water temperature*	1.33	Water temperature*	-0.40
	Air temperature*	-1.23	Water level change*	-0.48
	Water velocity*	0.82	% clay in sediment	-0.34
	% slope*	0.79	% slope	-0.35
	Width of riprap zone*	-1.13		
	Circumference of riprap*	0.88		
	% silt in sediment*	0.91		
	R^2	**0.92**	**R^2**	**0.70**
E. princeps	No significant model Mean = 2 sd = 1.71		No significant model Mean = 2 sd = 0.98	

Results of travel distance analysis

G. vastus travels further from the waterline during emergence than does *S. spiniceps.* On non-riprapped riverbanks, *G. vastus* traveled an average of 4.36m (sd=1.14), while *S. spiniceps* only traveled 0.33m (sd=0.49) (Table 13). The presence of riprap (as a method of riverbank restoration and stabilization) reduced both *G. vastus* and *S. spiniceps* emergence distance. On sites with riprap, *G. vastus* traveled 2.52m (sd=1.19) from the waterline, while *S. spiniceps* traveled 0.15m (sd=0.08) from the waterline (Table 13).

Table 13. Mean distance traveled (m) from the waterline by *G. vastus* and *S. spiniceps* at non-riprapped and riprapped locations (* represents significance p<0.05), standard deviations are given in parenthesis.

	Non-riprapped	Riprap
Distance traveled (m) from the waterline by *G. vastus**	4.36(1.14)	2.52(1.19)
Distance traveled (m) from the waterline by *S. spiniceps**	0.33(0.49)	0.15(0.08)

Location also appeared to influence emergence distance, Mann-Whitney U analysis of *S. spiniceps* travel distance resulted in average distance traveled of 0.33m from the water at the upriver locations, and an average of 0.19m (p>0.05) at the down river locations. *G. vastus* also traveled further from the water at upriver sites than downriver sites (3.94m, 2.72m respectively) (p<0.05) (Table 14).

Table 14. Mean distance traveled (m) from the waterline by *G. vastus* and *S. spiniceps* at upriver and downriver locations (* represents significance p<0.05), standard deviations are given in parenthesis.

	Upriver	Downriver
Distance traveled (m) from the waterline by *S. spiniceps*	0.33 (0.55)	0.19 (0.09)
Distance traveled (m) from the waterline by *G. vastus**	3.94 (1.49)	2.72(1.17)

A total of eight abiotic factors (Table 15) were correlated with the distance *S. spiniceps* traveled from the water. All eleven abiotic factors were significantly correlated with the distance *G. vastus* traveled from the water.

Table 15. Pearson correlation on the relationship of selected abiotic factors at upriver versus downriver locations to distance traveled (m) from the waterline by *S. spiniceps* and *G. vastus* ($p<0.05$), --- represents no significant correlation.

Abiotic factors	Species	Upriver	Downriver
Air temperature	*S. spiniceps*	0.35	0.43
	G. vastus	0.18	-0.18
Water temperature	*S. spiniceps*	0.40	0.43
	G. vastus	0.24	-0.19
Substrate temperature	*S. spiniceps*	0.28	0.49
	G. vastus	0.10	-0.27
Water level change	*S. spiniceps*	0.22	---
	G. vastus	-0.61	0.10
Width of riprap (m)	*S. spiniceps*	---	-0.46
	G. vastus	0.22	-0.15
Circumference of riprap (cm)	*S. spiniceps*	---	0.36
	G. vastus	---	-0.18
Water velocity (m/sec)	*G. vastus*	0.40	0.17
Slope	*G. vastus*	-0.30	0.51
% sand	*S. spiniceps*	---	0.21
	G. vastus	-0.16	0.59
% silt	*S. spiniceps*	-0.23	---
	G. vastus	0.19	-0.49
% clay	*G. vastus*	0.19	-0.35

Temperature

Air temperature was positively correlated with distance traveled by *S. spiniceps* at both upriver and downriver locations. For every one-degree (F) increase in air temperature, *S. spiniceps* nymphs traveled an additional 0.35m up the riverbank (at upriver transects), and an additional 0.43m up the riverbank (at downriver transects) (Fig 14). Although the correlations are not very strong, they are illustrative of the effect of small-scale changes on *S. spiniceps* travel distance.

For every one-degree (F) increase in air temperature above the intake/outtake facility, *G. vastus* traveled an additional 0.18m from the water. Below the intake/outtake facility, the 1^0 F air temperature increase resulted in *G. vastus* reducing its travel distance by 0.18m (Table 15, Fig 15).

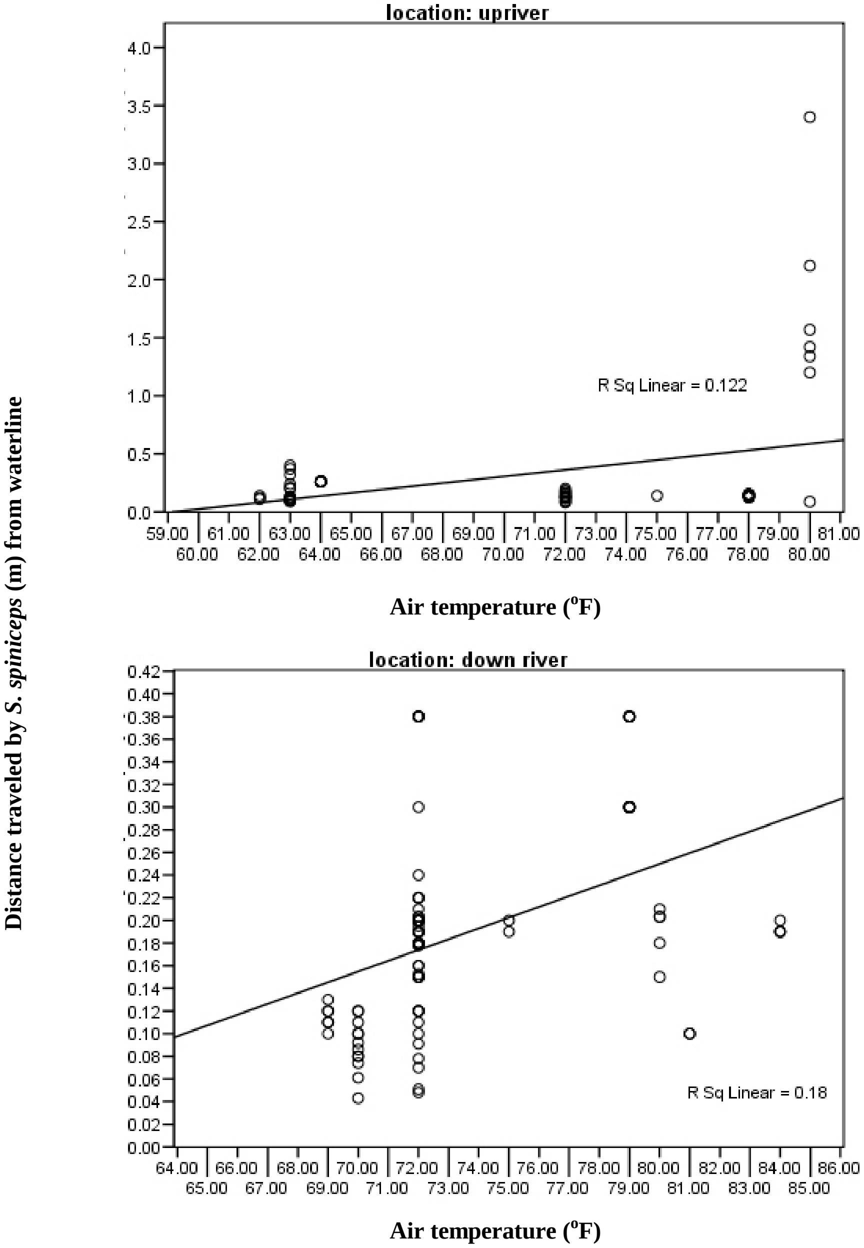


Figure 14. Distance traveled (m) from waterline by *S. spiniceps* correlated with air temperature.

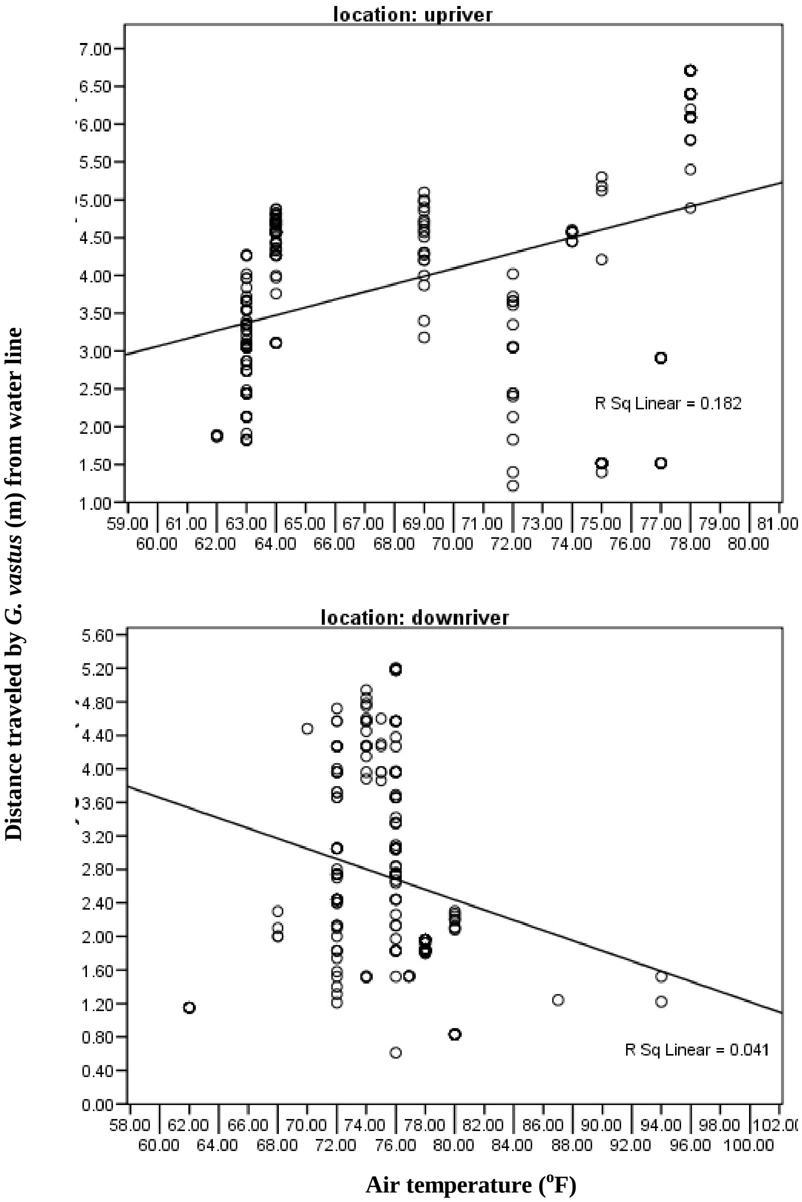


Figure 15. Distance traveled (m) from waterline by *G. vastus* correlated with air temperature

Water temperature was also significantly correlated ($p<0.05$) with the distance traveled by *S. spiniceps* nymphs. A 1^0 F increase in water temperature resulted in a 0.40m increase in

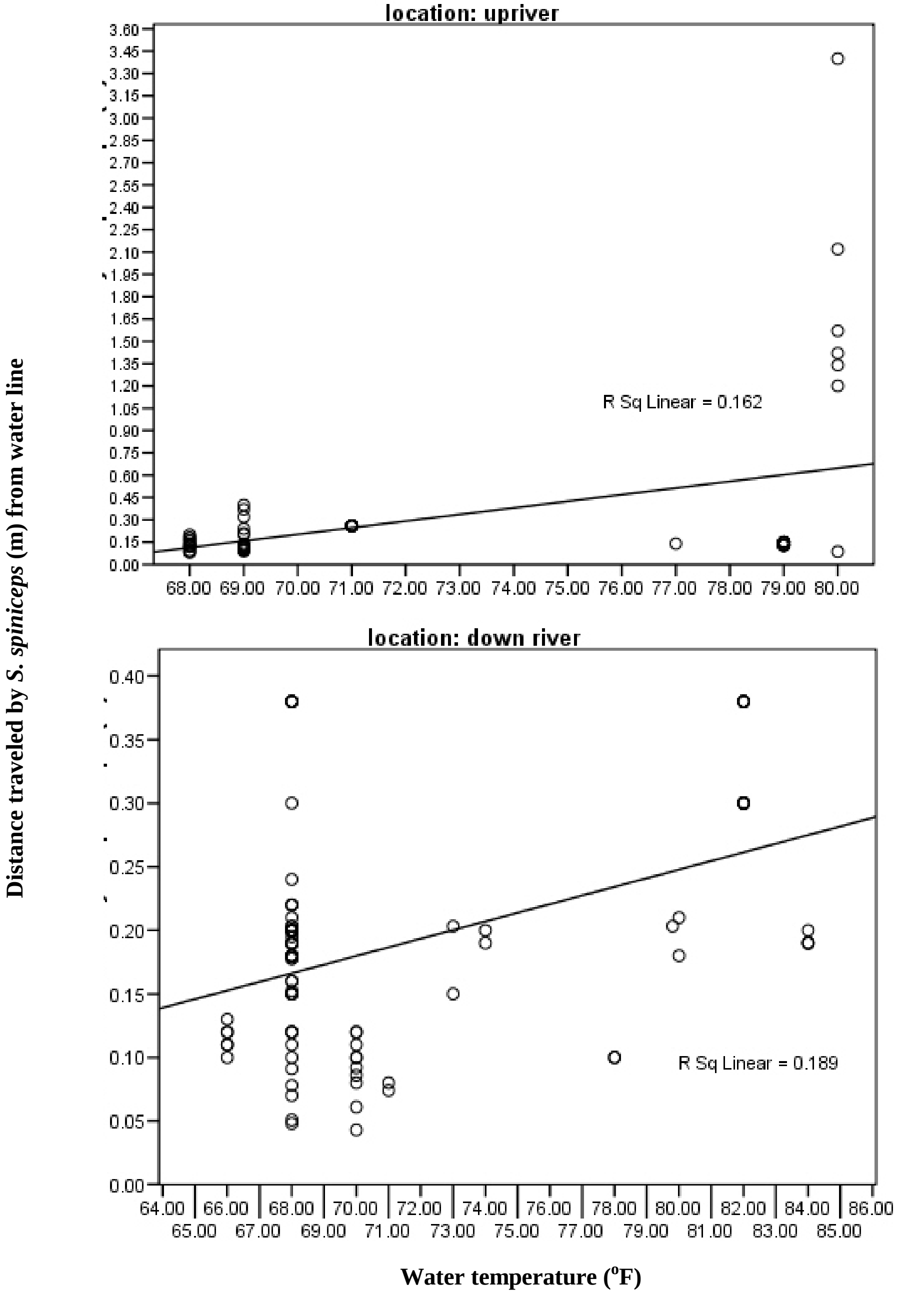


Figure 16. Relationship of water temperature on distance traveled (m) from waterline by *S. spiniceps.*

distance traveled from the waterline by *S. spiniceps* at both upriver and downriver locations (0.28m upriver, 0.49m downriver) (Fig 16). *G. vastus* traveled an additional 0.10m at upriver locations and decreased its distance traveled by 0.27m at downriver locations (Table 15, Fig 17).

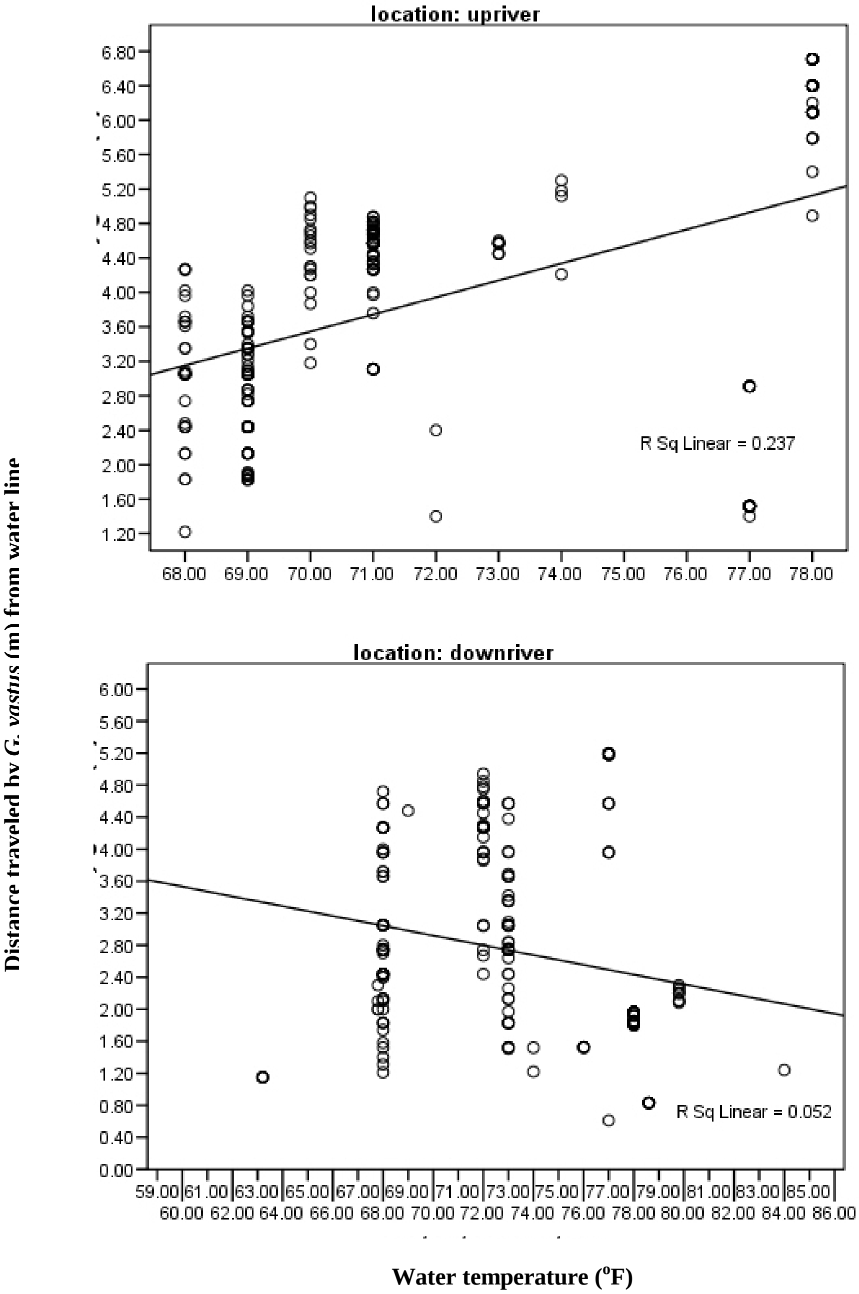


Figure 17. Relationship between water temperature (^{o}F) and the distance traveled (m) from waterline by *G. vastus*.

A 1^0 F increase in substrate temperature was related to an increase in distance traveled by *S. spiniceps* at both upriver and downriver locations (0.28m upriver, 0.49m downriver) (Fig. 18). *G. vastus* traveled an additional 0.10m at upriver locations and decreased its distance traveled by 0.27m at downriver locations (Table 15, Fig. 19).

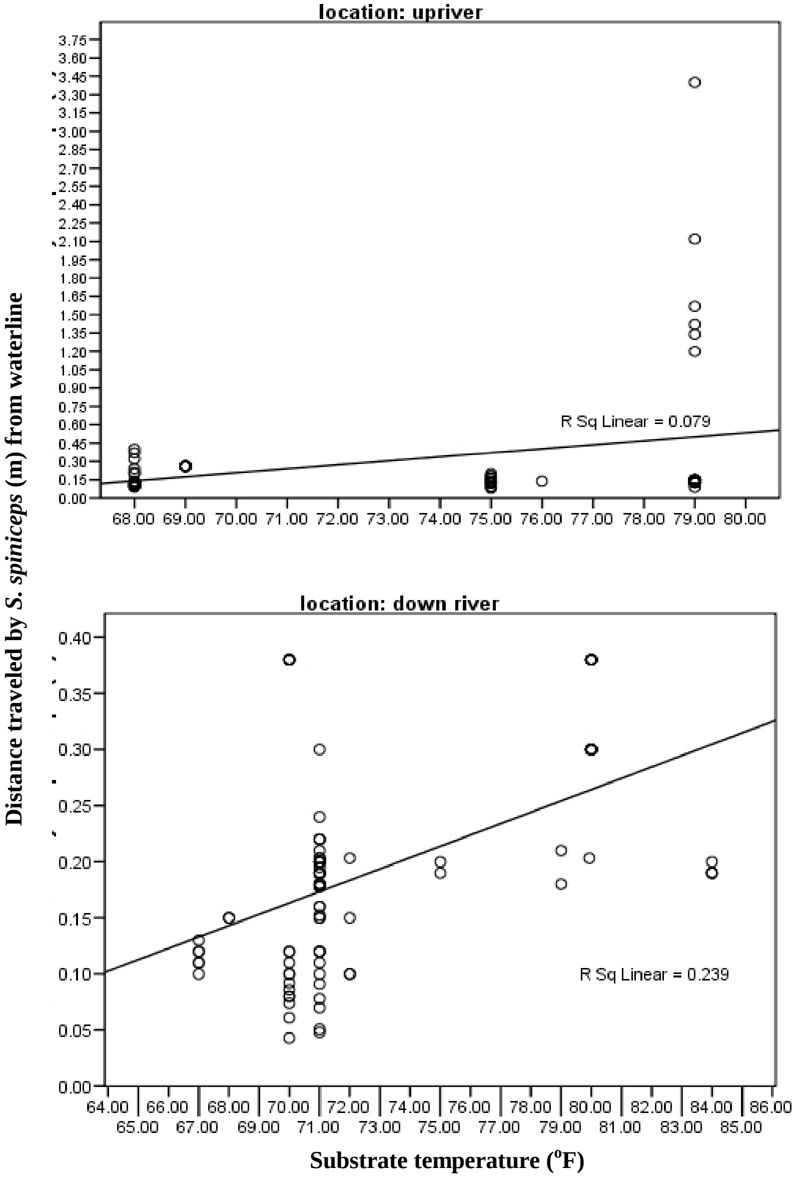


Figure 18. Relationship between substrate temperature (^{o}F) and the distance traveled (m) from waterline by *S. spiniceps*.

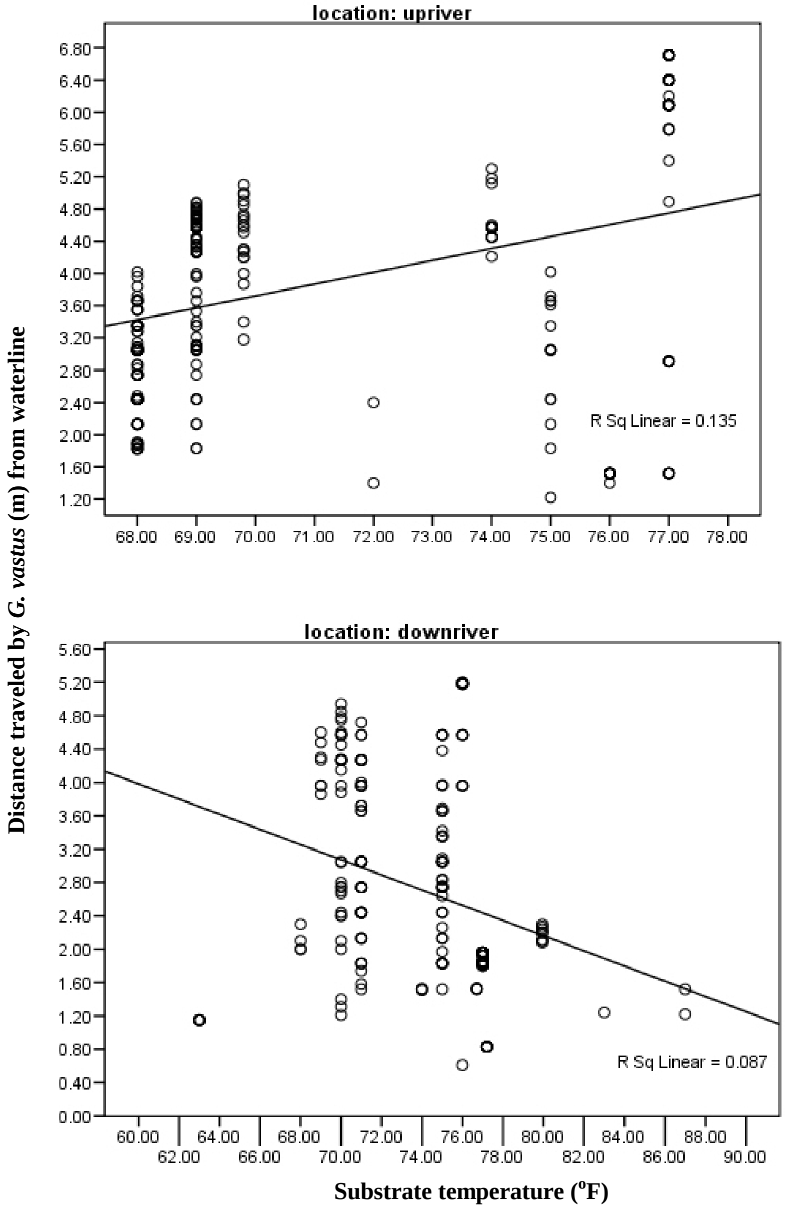
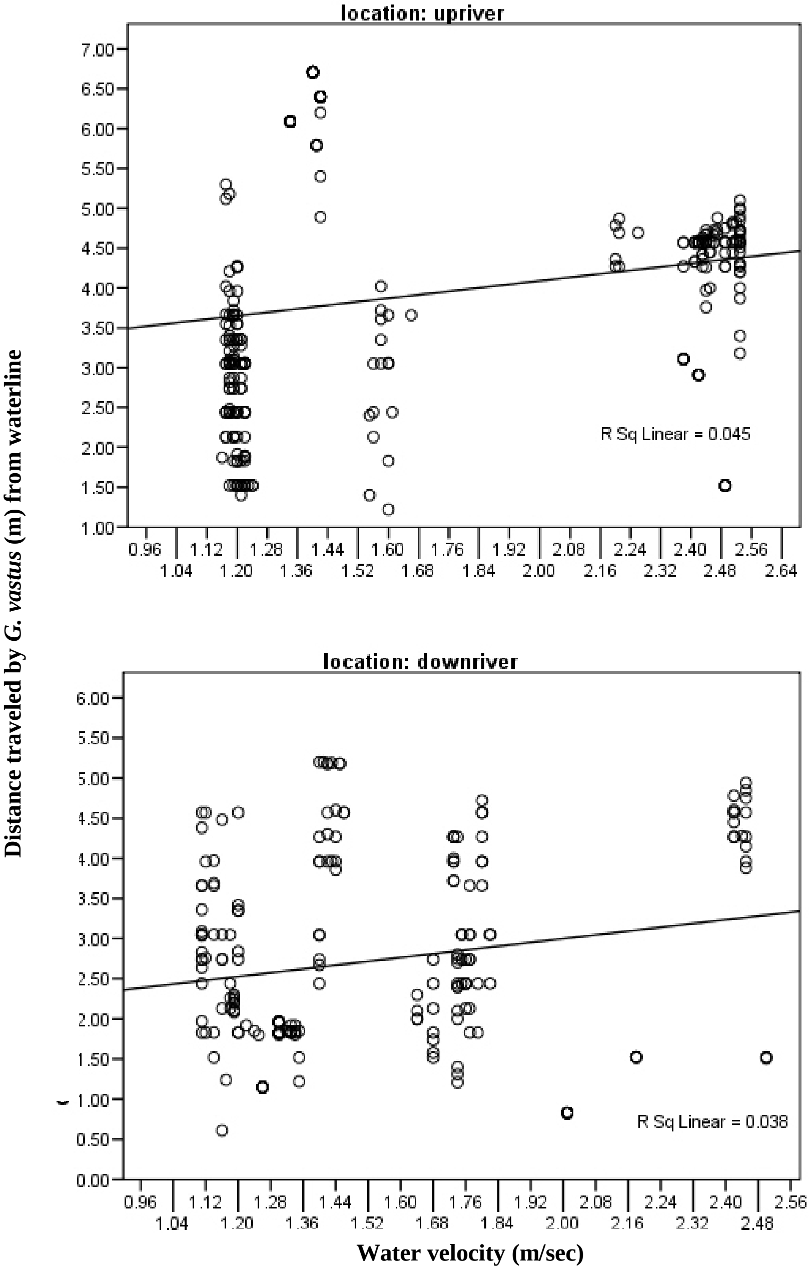


Figure 19. Relationship between substrate temperature (°F) and distance traveled (m) from waterline by *G. vastus.*

Water velocity and level

Water velocity was correlated with an increase in distance traveled by *G. vastus* at both upriver and downriver locations. A 1m/sec increase in water velocity was related to an increase in distance traveled from the waterline (0.40m upriver, 0.17m downriver) (Fig 20).



location: upriver

location: downriver

Figure 20. Relationship between water velocity (m/sec) and distance traveled (m) from waterline by *G. vastus*

Water velocity was not significantly correlated with distance traveled by *S. spiniceps*.

A 1cm/hour increase in water level was related to an increase in the distance *S. spiniceps* traveled from the water at upriver locations (0.22m) (Table 15, Fig 21). *G. vastus* responded to the same rate of water level increase differently depending on whether it was above or below the intake/outtake facility. At the upriver sites, a 1cm/hr water level rise resulted in *G. vastus* reducing its travel distance by 0.40m. At the downriver sites, *G. vastus* increased its travel distance by 0.10m (Table 15, Fig 21).

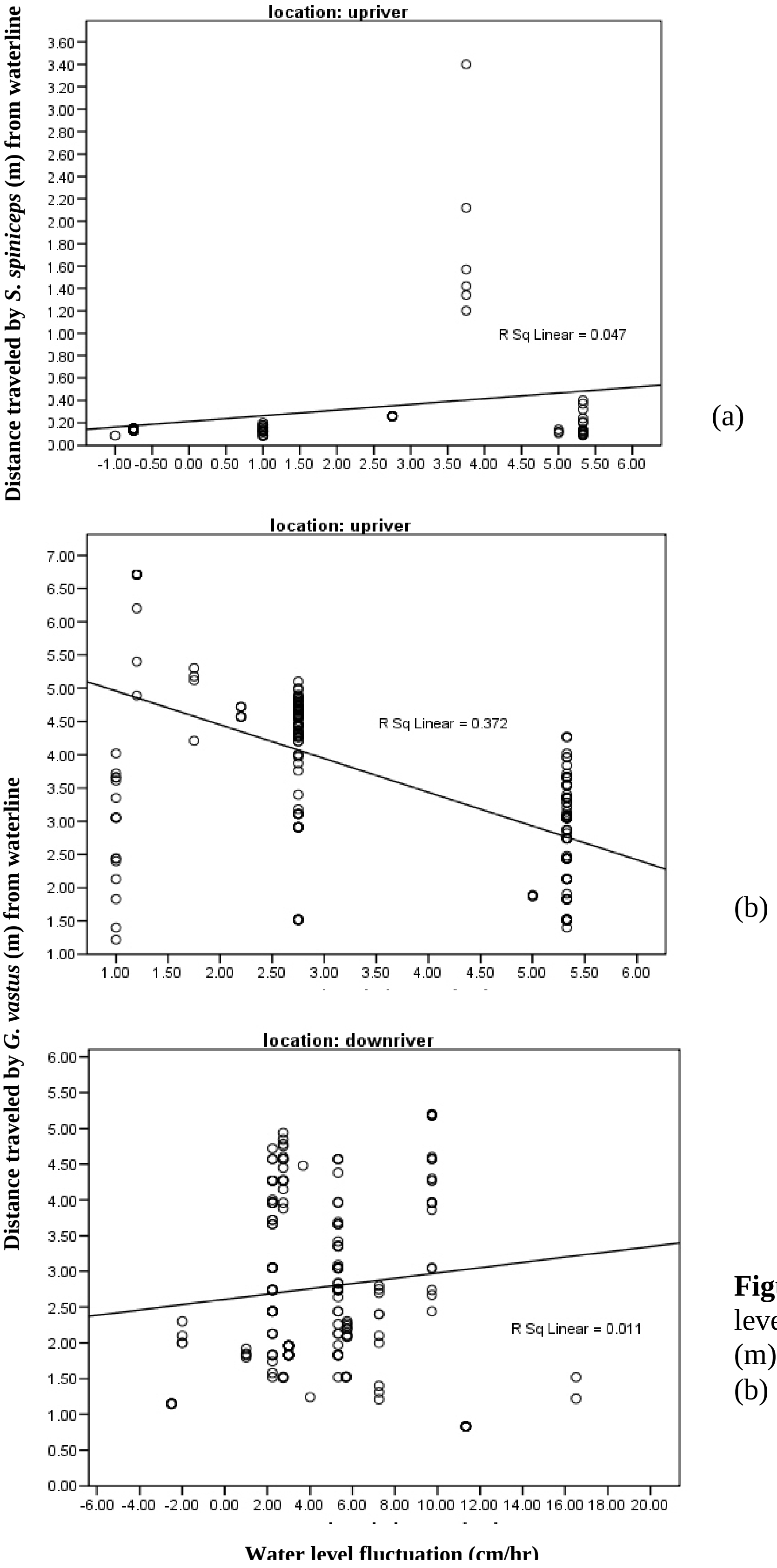


Figure 21. Relationship between water level change (cm/hr) and distance traveled (m) from waterline by (a) *S. spiniceps* and (b) *G. vastus*..

Riverbank features

Both width of the riprap zone, and the circumference of rocks within the zone were negatively correlated with *G. vastus* emergence distance but positively correlated with *S. spiniceps* emergence distance.. For every 1m increase in riprap zone width, *S. spiniceps* increased its travel distance by 0.38m, while *G. vastus* reduced its travel distance by 0.18m (Table 15, Fig 22). For every 1cm increase in rock circumference within the riprap zone, *S. spiniceps* decreased its travel distance by 0.46m (downriver), while *G. vastus* increased its travel by 0.22m (upriver) (Table 15, Fig 23).

Slope was a significant feature for *G. vastus* travel distance. At upriver sites, a 1% increase in slope resulted in a 0.30m reduction in distance traveled, while the same slope increase at downriver locations caused *G. vastus* to travel an additional 0.51m from the water. (Table 15, Fig 24).

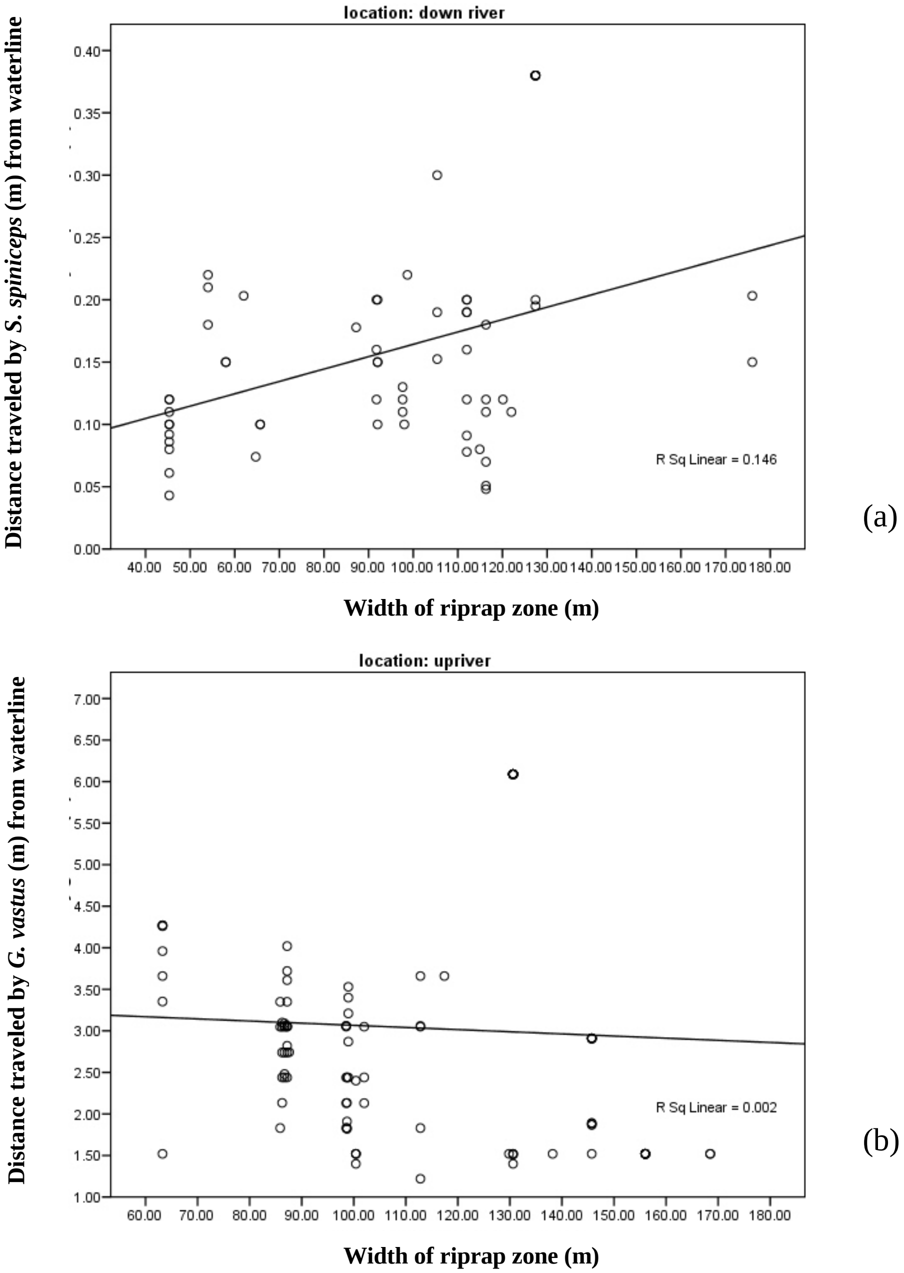


Figure 22. Relationship between width of riprap zone (m) and distance traveled (m) from waterline by (a) *S. spiniceps* at downriver locations and (b) *G. vastus* at upriver locations.

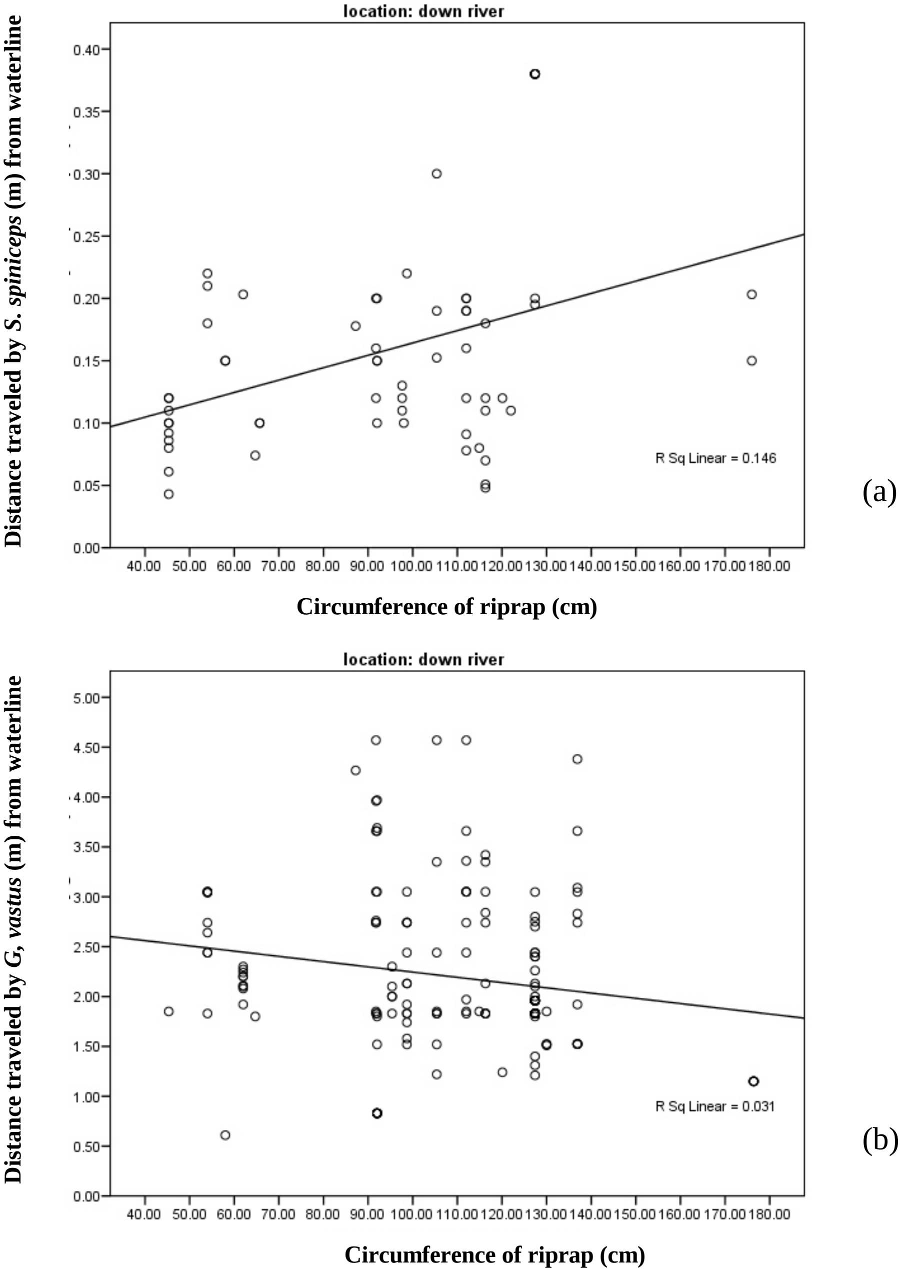


Figure 23. Relationship between riprap circumference (cm) and distance traveled by (a) *S. spiniceps* at downriver locations and (b) *G. vastus* at downriver locations.

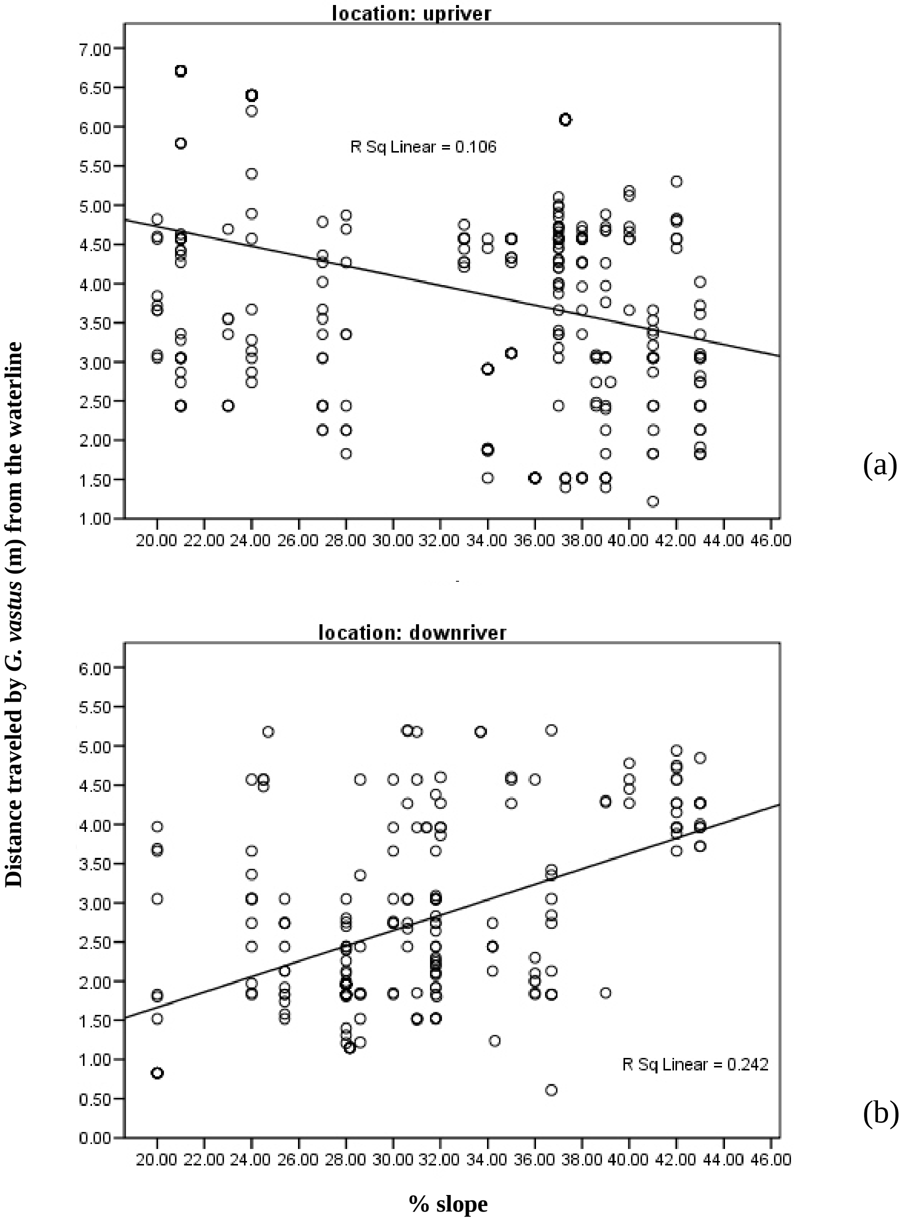


Figure 24. Relationship between slope (%) and the distance traveled from the waterline by *G. vastus* at (a) upriver and (b) downriver locations.

Composition of sediment

The composition of the sediment (sand, silt, clay) within the emergence zone, had a significant impact on the distance traveled by *G. vastus.* For every 1% increase in sand within the sediment, *S. spiniceps* increased its travel distance by 0.21m (downriver sites). An 1% increase in sand led to a 0.16m reduction in the distance traveled by *G. vastus* at upriver sites, and a 0.59m increase in the distance traveled at downriver sites (Table 15, Fig 25). A 1% increase in silt caused a 0.23m reduction in the distance traveled by *S. spiniceps* (upriver sites), while *G. vastus* increased its travel distance by 0.19m at upriver sites, and reduced its travel distance by 0.49m at down river sites (Table 15, Fig 26). For every 1% increase in clay, *G. vastus* traveled 0.19m further from the water at upriver sites, but reduced its distance by 0.35m at downriver sites (Table 15, Fig 27).

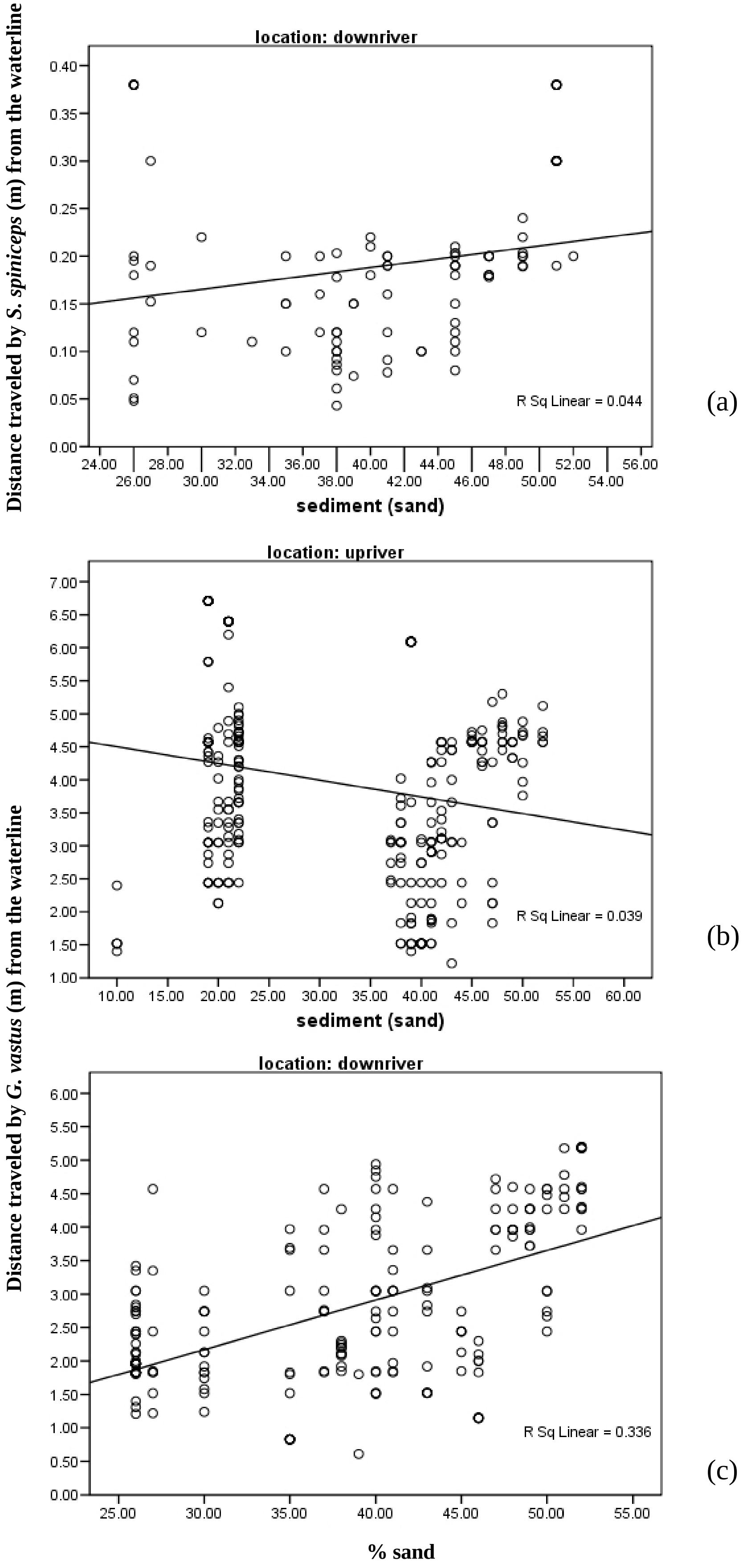


Figure 25. Relationship between the percentage of sand within the sediment and the distance traveled by (a) *S. spiniceps* at upriver locations, (b) *G. vastus* at upriver locations, and (c) *G. vastus* at downriver locations.

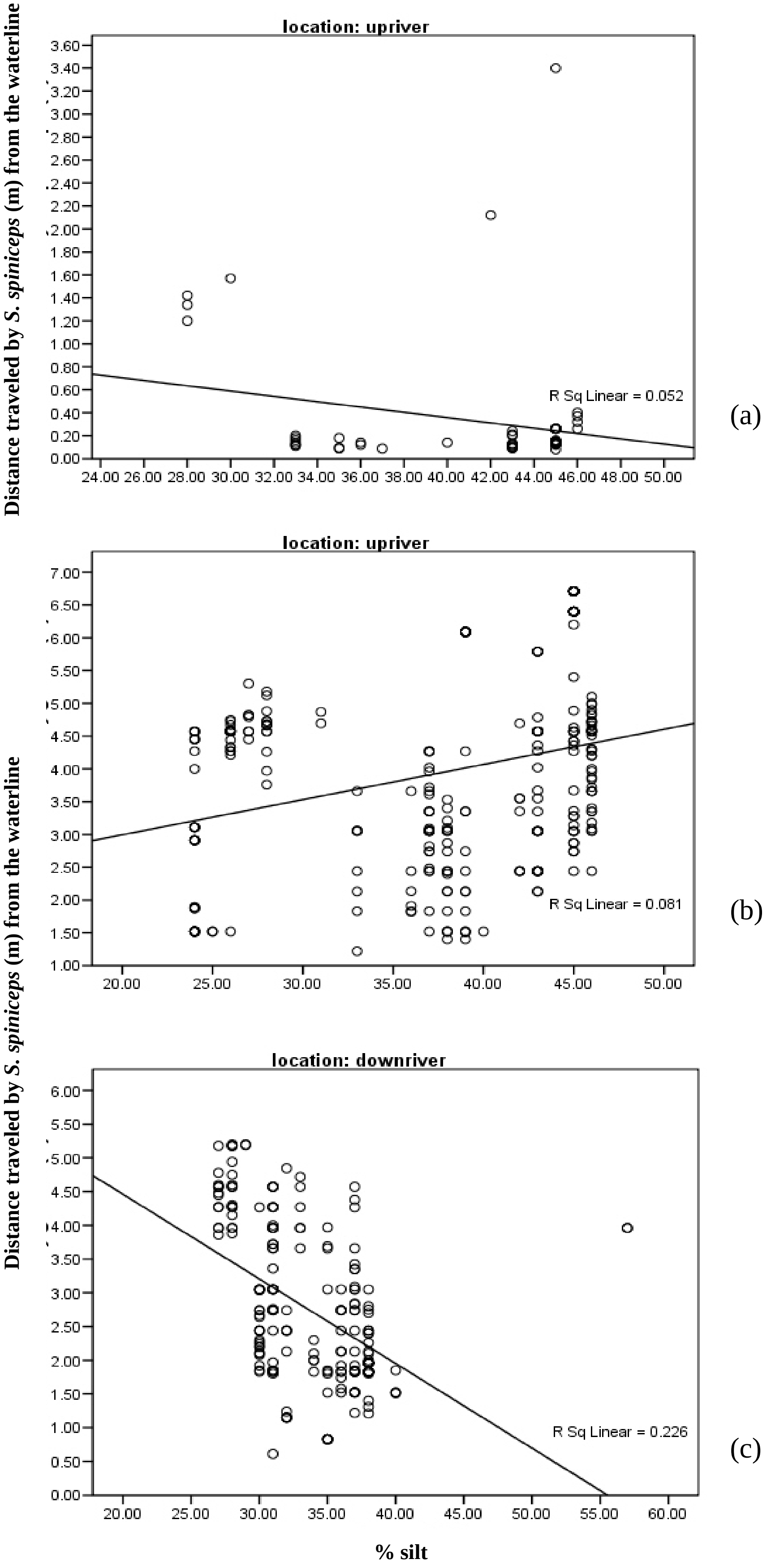


Figure 26. Relationship between the percentage of silt within the sediment and the distance traveled by (a) *S. spiniceps* at upriver locations, (b) *G. vastus* at upriver locations, and (c) *G. vastus* at downriver locations.

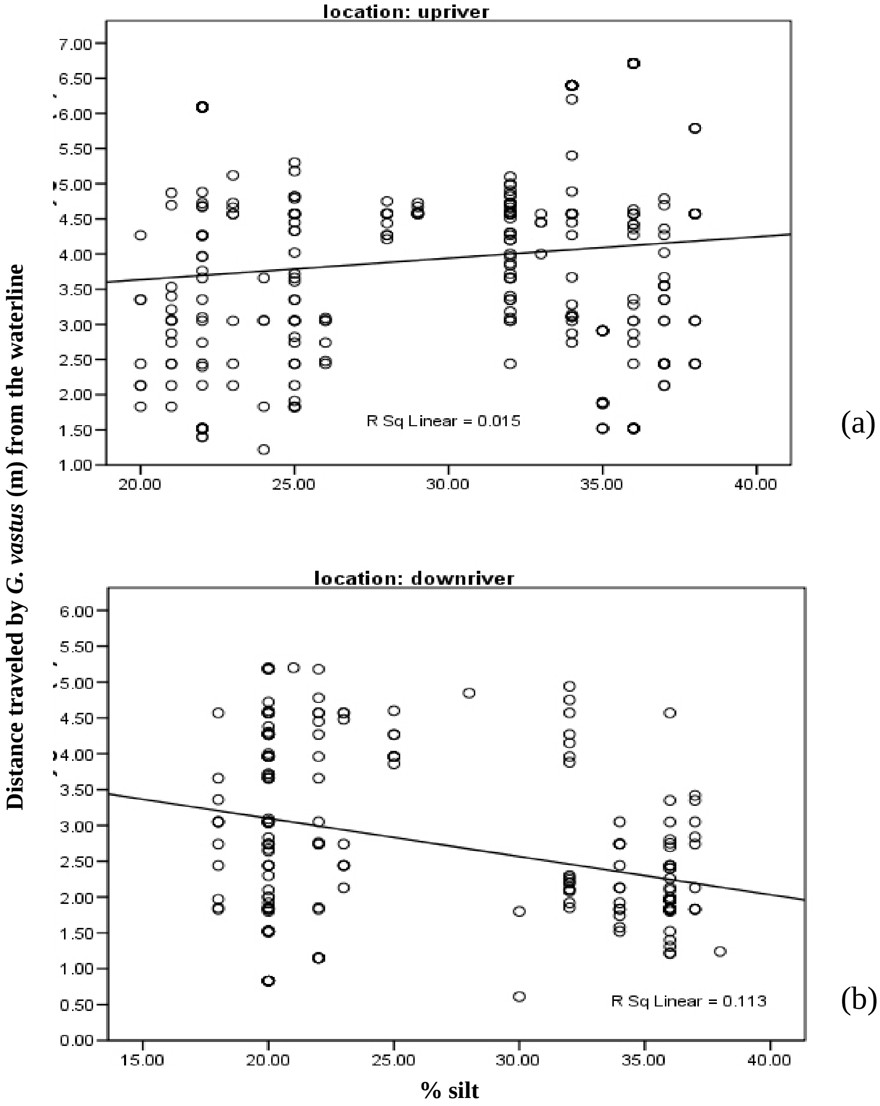


Figure 27. Relationship between the percentage of silt within the sediment and the distance traveled by *G. vastus* at (a) upriver locations, and (b) downriver locations.

Model selection emergence distance

The Pearson correlations discussed in the previous section highlights the effect of single abiotic factors on how far *S. spiniceps* and *G. vastus* traveled from the water. While the correlations are illustrative of specific impacts within the Turners Falls Reservoir system, they do not adequately describe the complexities of the system. In an attempt to more accurately describe the natural system present in the Turners Falls Reservoir, I used backward elimination multiple regression (BEMR) to illuminate the intricate connections between multiple abiotic factors and dragonfly emergence distance.

BEMR analysis of the distance traveled by *S. spiniceps* resulted in two different models. The downriver model contained more significant ($p<0.05$) factors than the upriver model. Overall, the upriver model accounts for 88% of the variance shown within the distance traveled by *S. spiniceps*, while the downriver model accounts for 54% of the variance (Table 16).

Table 16. Significant models of the relationships between multiple abiotic factors and distance traveled from the waterline by S. *spiniceps* and *G. vastus* as generated by BEMR analysis. All abiotic factors are significant ($p<0.05$).

	Upriver		**Downriver**	
	Features	**Standardized B**	**Features**	**Standardized B**
S. spiniceps	Air temperature	2.15	Air temperature	-0.59
	Water velocity	0.58	Water temperature	1.76
	Water level change (cm)	2.05	Water level change (cm)	0.38
	Width of riprap (m)	2.40	Width or riprap (m)	-1.06
	Circumference of riprap (mm)	-1.93	Circumference of riprap (mm)	0.54
	% silt in sediment	0.69	Water velocity (m/sec)	0.37
			Substrate temperature	-0.81
	R^2	**0.88**	**R^2**	**0.54**
G. vastus	Water temperature	0.67	Air temperature	-0.50
	Substrate temperature	-0.48	Substrate temperature	0.35
	Water velocity	-0.09	Water velocity	-0.23
	Water level change	-0.61	Water level change	0.16
	Width of riprap	0.42	Width of riprap	-0.45
	Circumference of riprap	-0.73	Circumference of riprap	-0.19
	% sand in sediment	0.59	% slope	0.31
	% silt in sediment	0.47	% silt in sediment	-0.24
	% clay in sediment	0.30		
	R^2	**0.77**	**R^2**	**0.67**

Air temperature was significant in both models. For every 1^0 F increase in air temperature at the upriver sites, *S. spiniceps* traveled an additional 2.15m. At the downriver sites a 1^0 F increase in air temperature was correlated with a 0.59m reduction in the distance traveled by *S. spiniceps*.

A 1^0 F increase in water temperature at the downriver sites resulted in a 1.76m increase in distance traveled by *S. spiniceps*. Water temperature was not included in the upriver model. Substrate temperature was only included in the downriver model; for every 1^0 F increase substrate temperature, *S. spiniceps* reduced its travel distance by 0.81m.

Water velocity was included in both models. A 1m/sec increase in water velocity was correlated with a subsequent increase in distance traveled (0.58m upriver, 0.37m downriver). A 1cm/hour change in water level was correlated with increases in *S. spiniceps* travel distance at both upriver and downriver locations (2.05m, 0.38m respectively). Increasing the width of the riprap zone by 1m was correlated with an increase (2.40m) in travel distance at the upriver locations, and a decrease (1.06m) in travel distance of *S. spiniceps* at the downriver locations. The affect of rock circumference within the riprap zone also varied between upriver and downriver locations. A 1cm increase in rock size at the upriver sites was correlated with a 1.93m reduction in travel distance, while the same rock size increase at downriver sites was correlated with a 0.54m increase in travel distance. Sediment composition (% silt) was only included in the upriver model. A 1% increase in silt was correlated with a 0.69m increase in distance traveled by *S. spiniceps*.

BEMR analysis of the distance traveled by *G. vastus* resulted in two significant models. The upriver model accounts for 77% of the variance shown within the distance traveled by *G. vastus*, while the downriver model accounts for 67% of the variance (Table 16). Air temperature was included only in the downriver models. For each 1^0 F increase in air temperature, *G. vastus* travel distance was reduced by 0.50m.

Substrate temperature was significant in both models. For every 1^0 F increase in substrate temperature at the upriver sites, *G. vastus* decreased its travel distance by 0.48m, while at the downriver sites, distance traveled increased by 0.35m. A 1m/sec increase in water velocity was correlated with a decrease in travel distance of 0.09m at the upriver sites, and a reduction in travel distance of 0.23m at the downriver sites. Water level change was negatively correlated with *G. vastus* travel distance at downriver sites, but positively correlated with the distance traveled at upriver sites. A 1cm/hour change in water level resulted in an increasing travel distance by 0.16m downriver, and decrease of 0.61m upriver.

Riprap characteristics (width, and size of rocks within the zone) were significant factors in both models. Increasing the width of the riprap zone by 1m resulted in a reduction in travel distance downriver (0.45m), while the same increase in riprap width upriver resulted in *G. vastus* increasing its travel distance by 0.42m. Increasing the circumference of the riprap by 1cm resulted in a decrease in travel distance in both upriver and downriver locations (0.73m upriver, 0.19m downriver).

The percentage of silt within the sediment was a significant factor in determining the distance traveled by *G. vastus* at both locations. Increasing silt composition by 1% decreased the distance *G. vastus* traveled by 0.24m at downriver sites, but increased the distance traveled by 0.47m at upriver sites. The percentage of sand within the sediment was a significant factor at upriver locations, a 1% increase led to an additional 0.59m being traveled by *G. vastus*. Slope was a significant component of the downriver model. A 1% increase in slope, increased the distance traveled by 0.31m.

Discussion

The purpose of this study was twofold, the first, was to examine links between the density of riverine nymphs and selected riverbank characteristics. The results of my research study support rejection or modification of my initial density and emergence distance predictions. The second purpose of my study was focused on the effects of riverbank restoration on the emergence distance of two MESA listed species, *S. spiniceps* and *G. vastus*. Since nymphs are highly vulnerable during the eclosure process, and high water levels and wakes may dislodge nymphs from their perching location, the distance these species travel from the river's edge reflects directly on the future adult populations.

Density hypothesis one: Density of *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. princeps* will not be affected by location (upriver versus downriver of hydroelectric intake/outflow facility).

The analysis of the data collected indicates that species react individually to site-specific abiotic factors. Individual abiotic features were components of several models, but their inclusion varied considerably between species, and location. While *S. spiniceps, N. yamaskanensis, D. spinosus, O. rupinsulensis, and E. princeps* densities did vary with location, they were not statistically different ($p>0.05$). Both *G. vastus* and *M. illinoiensis* densities were significantly ($p<0.05$) different between upriver and downriver locations, although each species responded differently to location (Fig 28). *G, vastus* densities were higher at upriver sites while *M. illinoiensis* densities were higher at downriver locations.

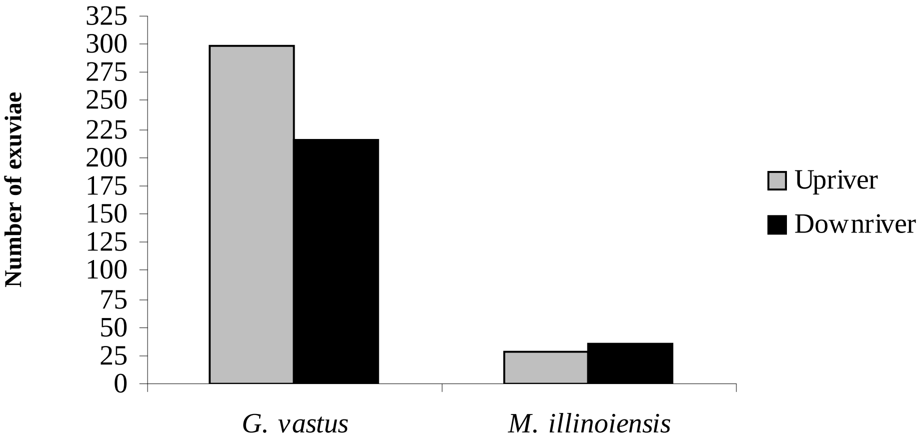


Figure 28. Total number of *G. vastus* and *M. illinoiensis* exuviae found at upriver versus downriver locations.

Density hypothesis two: Density of *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus, O. rupinsulensis*, *M. illinoiensis,* and *E. princeps* will not be affected by air temperature, water temperature, substrate temperature, or slope.

Temperature (air, water, and substrate) figured prominently in determining density, but there were some interesting differences between the three features. While decreases in air temperature were correlated with a reduction in *S. spiniceps* density, increases in both water and substrate temperature resulted in an increase in *S. spiniceps* density. Presence of riprap, more specifically the size of the rocks within the riprap zone, also had a negative effect on *S. spiniceps* density (Fig 29).

The models produced for *G. vastus* did not explain a lot of the variation in density. Although the upriver model included five abiotic features, it only accounted for 22% of *G. vastus* density. The downriver model, which only included one variable, was even less explanatory, encompassing only 8% of *G. vastus* density.

The models produced for *N. yamaskanensis* were very strong, explaining 89% of the variation found at upriver transects, and 92% of the variation found at downriver transects.

Interestingly, the two models had several abiotic features in common, although the effect of those features on *N. yamaskanensis* density was very different. Temperature (air and water) was more prominent in the downriver model, with air temperature having a positive effect on density, and water temperature having a negative effect. Water level change was included in both upriver and downriver models, and increases in water level (cm/hr) had a positive influence on density of *N. yamaskanensis*. Riprap (both width of the riprap zone, and circumference of the rocks within the zone) were only included in the upriver model. Increasing width of the riprap zone, was correlated with reductions in *N. yamaskanensis* density, but surprisingly, increases in rock size were correlated with increased *N. yamaskanensis* density. The percentage of silt within the sediment was included in both models, although it was only significant in the upriver model, where it was associated with a reduction in density.

The only significant association for *D. spinosus* density was percent slope (at upriver sites only). An increase in bank slope was associated with an increase in density.

For *O. rupinsulensis* the only significant model generated was for downriver sites. Although the model explained 87% of the variation found at the downriver sites, it only contained three abiotic factors (size of riprap, percentage of clay, and percentage of sand in the sediment). All three abiotic factors served to decrease nymphal density. Increases in both the percentage of sand and percentage of clay more strongly correlated with reductions in *O. rupinsulensis* density that increases in the circumference of rocks within the riprap zone.

M. illinoiensis density was positively associated with water temperature at upriver sites, but negatively associated at downriver sites. Water velocity, size of riprap, slope, and the percentage of silt in the sediment were all associated with increased nymphal density at upriver sites. Air temperature and width of the riprap zone at upriver sites were all associated with decreased nymphal density.

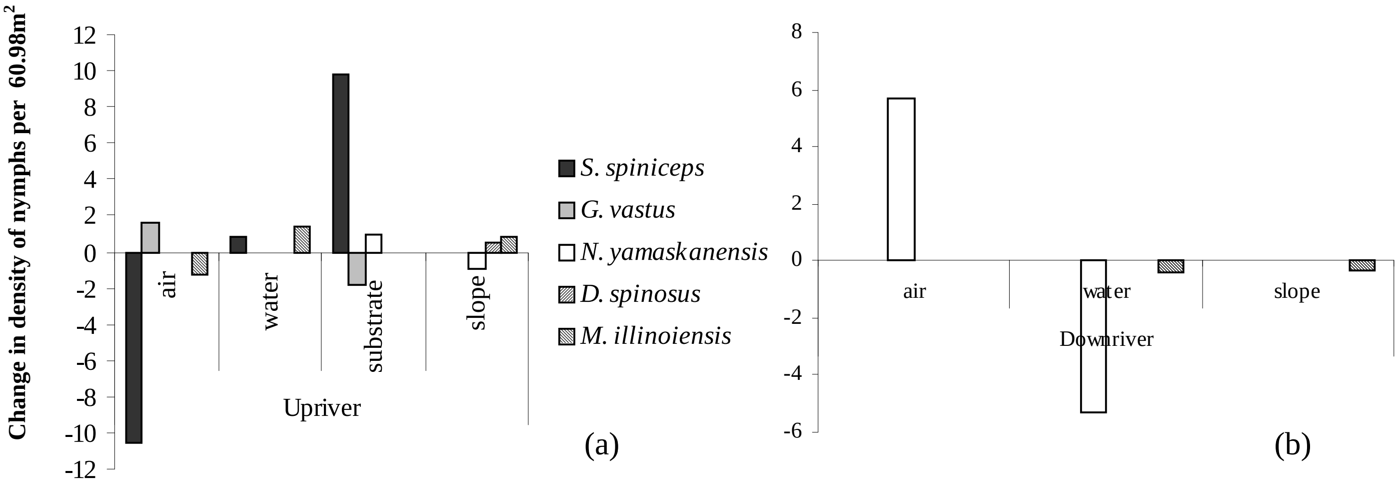


Figure 29. Impact of increasing air temperature, water temperature, substrate temperature, and slope on density of *S. spiniceps*, *G.vastus*, *N. yamaskanensis*, *D. spinosus*, and *M. illinoiensis* at (a) upriver and (b) downriver locations.

<u>Density hypothesis three</u>: Density of *S. spiniceps*, *G. vastus*, *N. yamaskanensis*, *D. spinosus*, *O. rupinsulensis*, *M. illinoiensis*, and *E. princeps* will be influenced by water velocity, water level change, presence of riprap, width of the riprap zone, circumference of the rocks within the riprap zone, and composition of sediment (% sand, % silt, % clay).

As with the previous two predictions, results indicate that species (1) respond differently to different abiotic factors and (2) response is influenced by location. The presence or absence of riprap had no significant effect on densities of any of the subject species. Hypothesis three held true for *S. spiniceps* (circumference of rocks), *G. vastus* (circumference of rocks and water velocity), *N. yamaskanensis* (circumference of rocks, width of riprap zone, water velocity, and water level change), *O. rupinsulensis* (% clay and % silt), and *M. illinoiensis* (circumference of riprap, width of riprap, water velocity, water level change, % silt), although individual species responded very differently at upriver sites than they did at downriver locations (Fig 30).

Ultimately, the influence of the selected abiotic factors on density *S. spiniceps, G. vastus, N. yamaskanensis, D. spinosus, O. rupinsulensis, M. illinoiensis,* and *E. princeps* was not constant. The overall effect of the features depended on (1) location, and (2) the species. Based on the relatively low r^2 values of several of the models, it is likely that other factors, whether abiotic or biotic, are influencing the density of these species.

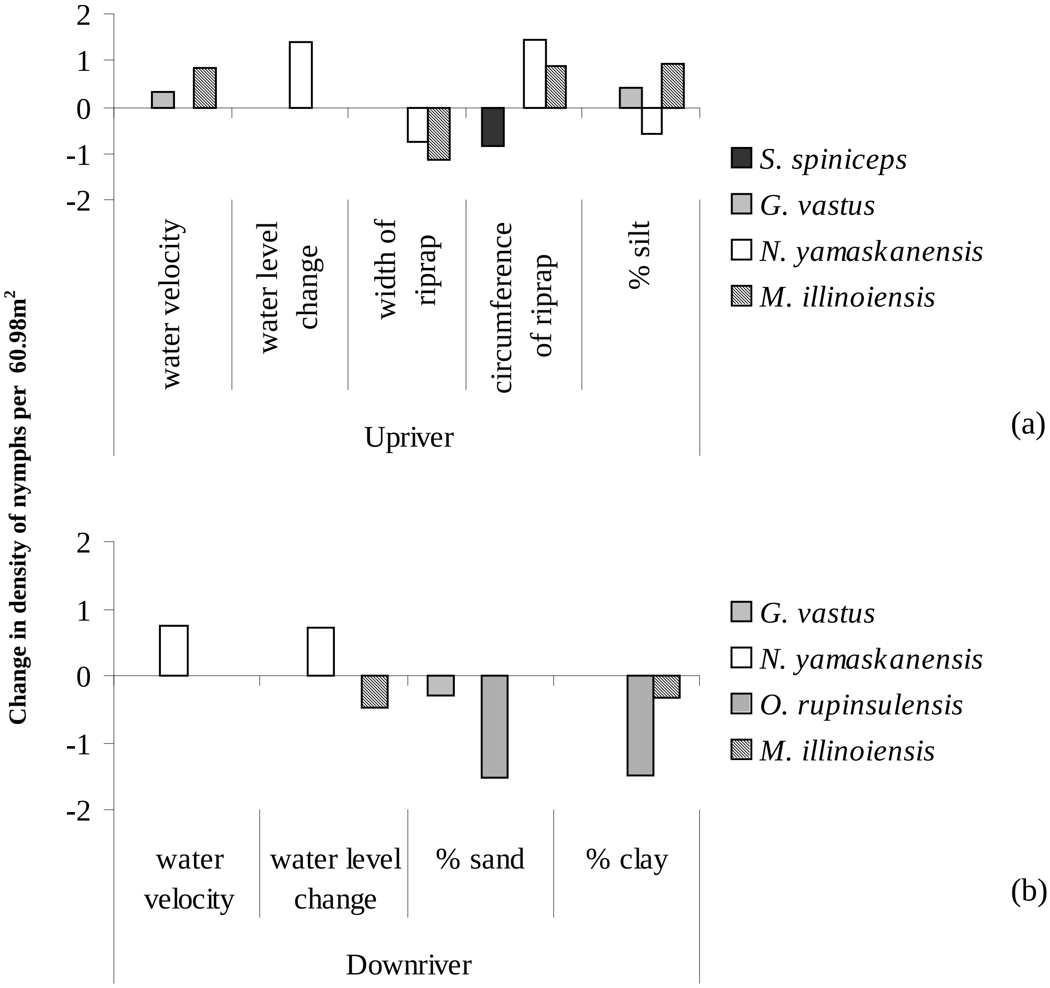


Figure 30. Relationship of increasing water velocity, water level change, width of riprap zone, circumference of rocks within riprap zone, and sediment composition (% sand, % silt, % clay) on density of *S. spiniceps, G.vastus, N. yamaskanensis, O. rupinsulensis* and *M. illinoiensis* at (a) upriver and (b) downriver locations.

Results of the emergence study are equally intriguing, while the reasons why *G. vastus* traveled significantly greater distances at the upriver sites are not known, the results of both the Pearson analysis and BEMR suggest some interesting trends.

Emergence hypothesis one: Location (upriver or downriver) will have no effect on *G. vastus or S. spiniceps* travel distance.

The distance traveled to eclosure site, in areas where there are no obvious limitations to travel, reflects species-specific preferences. The minimum height traveled for any species is at the water surface, or within 20 cm above the water surface (Corbet 1999). Gomphids generally travel between 25-50cm (Kurata 1971, Inoue 1979). In my study recorded travel distances for both *S. spiniceps* and *G. vastus* far exceeded the distances recorded by Kurata (1971) and Inoue (1979). The maximum distance traveled by *S. spiniceps* at any location was 0.88m (88cm). *G. vastus* traveled an impressive 3.94m (394cm) at upriver locations and 2.72m (272cm) at downriver locations. Due to the water level fluctuations within the reservoir, however, these distances may be slightly exaggerated. Results of my data analysis also suggest the need for modification of the three emergence distance predictions.

Emergence hypothesis two: *G. vastus* and *S. spiniceps* emergence distance will not be affected by air, water, or substrate temperature, or slope.

I hypothesized that air, water, and substrate temperatures would have no effect on the distance traveled by *S. spiniceps* and *G. vastus* at upriver and downriver locations. Based on my analysis, significant associations were found between all three temperatures and the distance traveled by *S. spiniceps* and *G. vastus*. Interestingly, the effects of air, water, and substrate temperature on distance traveled varied both with the species, and with location. In general, increasing temperature had a negative effect on *G. vastus* travel distance, specifically at downriver locations, but had a positive effect on *S. spiniceps* (Fig 31). The differences in effect due to location present several intriguing questions as to the effect of site-specific topography, vegetational features, and the impact of the intake/outtake facility. When correlated with other abiotic features, the three temperature variables remained significant, but their inclusion in the models was dependent on the location. The effect of location suggests that *G. vastus* and *S. spiniceps* are responding to site-specific features and processes. The variability at the ecotone seems to be a vital factor in whether or not these two species increase or decrease their emergence distances.

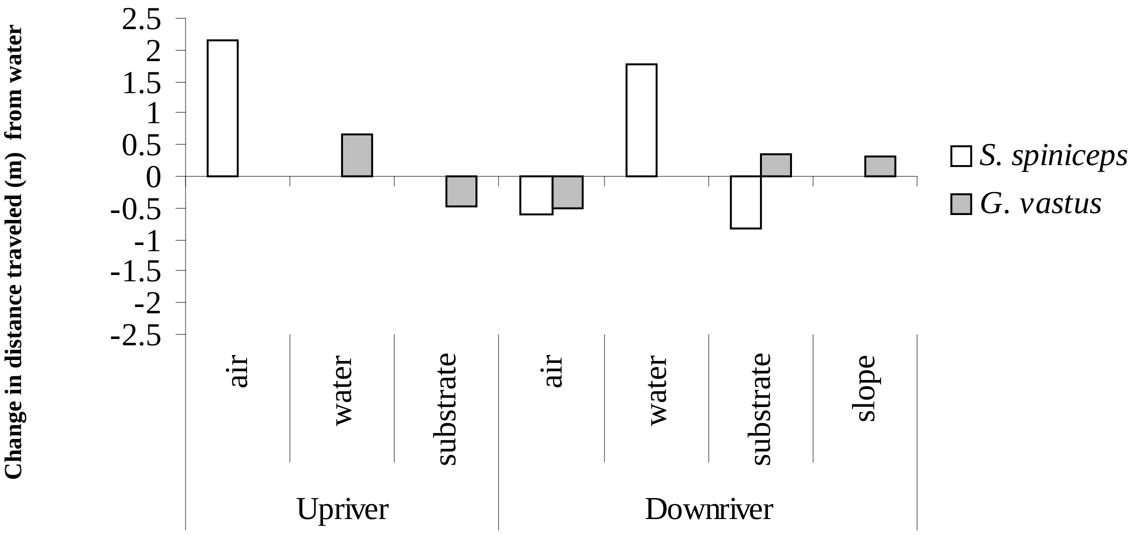


Figure 31. Effects of increasing air temperature, water temperature, substrate temperature and slope on distance traveled by *S. spiniceps* ($avg_{upriver}$=0.33m, $avg_{downriver}$=0.19m) and *G. vastus* ($avg_{upriver}$=3.94m, $avg_{downriver}$=2.72m).

<u>Emergence hypothesis three</u>: *G. vastus* and *S. spiniceps* emergence distance will be affected by water velocity, water level change, presence of riprap, width of the riprap zone, circumference of the rocks within the riprap zone, and composition of sediment (% sand, % silt, % clay).

Prior to this study, I hypothesized that certain bank characteristics (slope, presence of riprap, width of riprap zone, size of rocks within the riprap zone) would all affect the distance traveled by *S. spiniceps* and *G. vastus.* When considered independently, riprap had a significant impact on both *G. vastus* and *S. spiniceps* travel distance. Both species traveled significantly further from the waterline on riverbanks without riprap (Fig. 32).

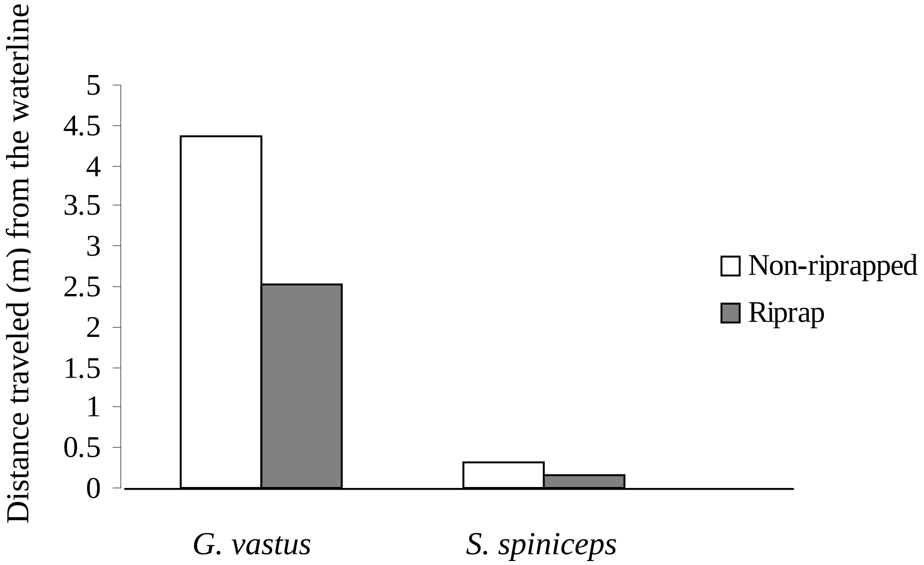


Figure 32. Mean distance traveled (m) from the waterline by *G. vastus* and *S. spiniceps* at non-riprapped and riprapped sites.

Species response to water velocity, water level change, width of the riprap zone, circumference of the rocks within the riprap zone, and composition of sediment (% sand, % silt, % clay) differed considerably depending upon location. Hypothesis three was supported for *G. vastus*, although response was clearly influenced by location. *S. spiniceps* did not respond as I had initially hypothesized, and significant relationships were only found at downriver locations (for width of riprap zone, and size of rocks within the zone). Interestingly, both *S. spiniceps* and *G. vastus* decreased their travel distances in areas with riprap. Both the width of the riprap zone and the circumference of the rocks within the zone decreased the distance traveled by both the species.

Since both species must cross sediment on their way up the bank, I hypothesized that they would both be affected by the percentage of sand, silt, and clay within the sediment. As with the previous abiotic features, *G. vastus* displayed the most correlations with sediment composition, although the actual impacts (increased or decreased travel distance) was variable depending upon location. *S. spiniceps* travel distance was not as clearly correlated with sediment composition.

I hypothesized that water velocity and water level change would both affect the distance traveled by the two species. As with the temperature analysis, *S. spiniceps* and *G. vastus* displayed different responses to the two abiotic variables. *S. spiniceps* was not significantly correlated with changes in water velocity at either the upriver or downriver locations, while *G. vastus* travel distance was positively correlated with water velocity (Fig 33). The influence of water velocity on *G. vastus* travel distance was also correlated with location. There was no uniformity in the effects of water level change on either *S. spiniceps* or *G. vastus*. While a 1cm/hr change in water level at upriver locations resulted in an increase in distance traveled by *S. spiniceps* it resulted in a decrease in distance traveled by *G. vastus*. *G. vastus* at downriver sites, traveled further when faced with the same water level change. These results suggest that there might be additional site-specific features that influence the behavior of these two species.

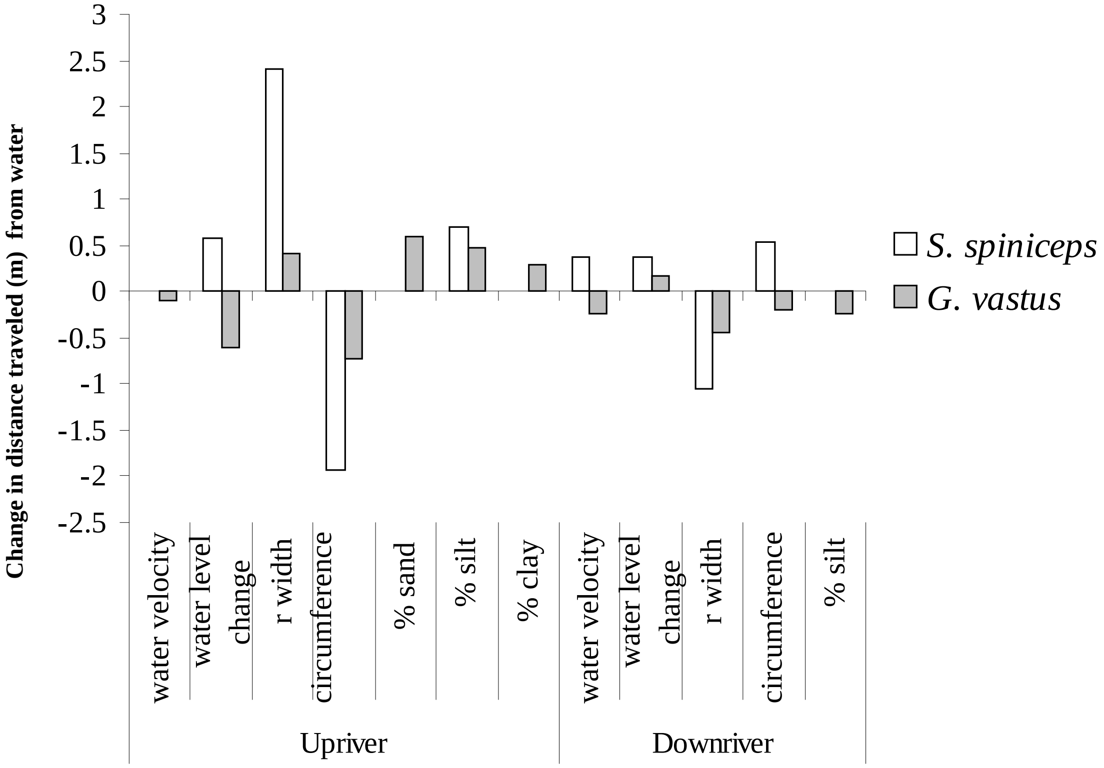


Figure 33. Relationship of water velocity, water level change, width of riprap zone (r width), circumference of rocks within the riprap zone (circumference), and composition of sediment (% sand, % silt, % clay) to the distance traveled from the waterline by *S. spiniceps* ($\text{avg}_{\text{upriver}}$=0.33m, $\text{avg}_{\text{downriver}}$=0.19m) and *G. vastus* ($\text{avg}_{\text{upriver}}$=3.94m, $\text{avg}_{\text{downriver}}$=2.72m).

While the individual correlations between the selected abiotic features and travel distance of *S. spiniceps* and *G. vastus* clearly indicate species-specific responses to environmental features, it is in consideration of multiple abiotic features that a clearer picture emerges. The resultant four models produced through BEMR analysis illustrate the pivotal impact that location has on both *S. spiniceps* and *G. vastus.* If location was not an influential feature, then the upriver and downriver models should be identical, and this was not the result. The upriver model for *S. spiniceps* travel distance included only six abiotic features (air temperature, water velocity, water level change, width of riprap zone, size of rocks within the riprap zone, and the percentage of silt i7n the sediment), yet it accounted for 88% of the impact on travel distance. The downriver model included seven features, yet only accounted for 54% of the impact on travel distance. In addition to the different assessment of abiotic impacts, individual abiotic features that were included in both models, did not have the same impact at upriver sites as they had at downriver sites. Since there were few universal impacts (occurring at both upriver and downriver locations) among the sediment features, the effect of microenvironment (i.e. Location of riverbank) appears to have a very influential role in determining how far *G. vastus* and *S. spiniceps* emerge.

The difference between models for *G. vastus* travel distance was not as distinctive as was the case for *S. spiniceps*, but there are still several interesting differences. The upriver model for *G. vastus* accounted for 77% of the impact on travel distance, while the downriver model accounted for only 67%. Only six abiotic factors (substrate temperature, water velocity, water level change, width of riprap zone, circumference of rocks within the riprap zone, and the percentage of silt in the sediment) were included in both models, yet the individual affect of these features varied widely between the models.

The results of this study suggest the powerful impact that microenvironments have on both the density and emergence distance of several New England riverine dragonfly nymphs. Many of the current biological assessment models which incorporate dragonfly ecology are focused on either a reach scale or even a landscape scale. Since results of my study support the conclusion that dragonfly density and emergence is intricately tied to ecological variations at the microscale level, it is imperative that these larger assessment models recognize the importance of these smaller scales and incorporate them into their framework. Overall, there were only a few abiotic features that resulted in similar densities or emergence distances at both upriver and downriver locations.

One of the strongest relationships highlighted by my research is the association between location, riprap, and emergence distance of *S. spiniceps.* During my previous observations in the Turners Falls Reservoir, I recorded *S. spiniceps* emergence distances of 0.37m on riverbanks without riprap, and 0.28m on riverbanks that had been stabilized with riprap. Results from my research study documented similar findings with *S. spiniceps* traveling 0.33m from the waterline on nonriprap stabilized banks, and 0.15m on riprap stabilized banks. Since water level fluctuations in the Turners Falls Reservoir average as much as 1.07m per day (U.S. Army Corps 1991), the shorter emergence distances evident as the result of the presence of riprap puts *S. spiniceps* at a high risk of injury or death.

The results of this study demonstrate the need for ongoing discourse on what factors are applied when determining significance of correlation data. The heavy reliance on p-values in determining significance often assigns undue importance to correlations that might not actually display any biological significance within the system. Although several of the correlations examined in this study were statistically significant ($p<0.05$), their r^2 values were extremely weak. Determination then, of what constitutes biological significance rests heavily on how well the system being studied is understood. In this study, correlations between abiotic factors and emergence distance of *S. spiniceps* and *G. vastus* were not overly strong. Based on extensive observations of these two species, however, it is apparent that even the most finite of connections is extremely influential on how far these species travel from the water's edge, which increases their likelihood of survival.

While many of the correlations and models are very strong ($r^2>0.50$), and could serve as valuable assessment tools for future bank restoration in the Turners Falls Reservoir, it is their site-specific nature that is the most revealing. Traditionally, species have been designated as stenotopic (species which are sensitive to changes in particular habitat conditions) or euryotopic (species that can withstand changes in habitat conditions) based on analysis of larger scales (reach or in some cases, watershed). The definition of what constitutes relative sensitivity or insensitivity to change is often highly variable depending upon (1) the unique characteristics of the system under investigation and (2) the species, or species assemblages being used as bioindicators. The results of my study highlight the impact that both of these variables have in designating relative resistance/ susceptibility to change. Stenotopic species are more useful as bioindicators, but in the Turners Falls Reservoir, it is difficult to assign any one particular species to the role of ideal stenotopic bioindicator.

Overall, results of this study show that (1) the correlation between nymphal density and abiotic features is not uniform, (2) relative location of the riverbank (in relation to intake/outflow facility) has a dramatic impact on both species density and emergence distance, (3) microscale (riverbank) features have significant impacts on nymphal dragonfly densities and emergence behavior, (5) the use of riprap stabilization as a form of restoration has a negative effect on emergence distance of *G. vastus* and *S.spiniceps*, and (4) statistical significance should be considered in its relation to biological significance.

In my particular study area, the effect of a finite group of abiotic variables had significantly different relationships to both nymphal densities and distance traveled from the waterline. Of all of the relationships that were highlighted by this study, the most intriguing, and potentially the most important to restoration ecologists, is the impact that riprap had on the distance traveled from the waterline by *G. vastus* and *S. spiniceps*. For *S. spiniceps* in particular, even the slightest decrease in the distance traveled from the water increases the risk of mortality due to water level increases or boat wake impacts. Although my study was focused on a particular system, and a finite number of species, the relationships highlighted generate questions about how other dragonfly species, in reservoir and non-reservoir systems might respond to changes at the small-scale interface zone between aquatic and terrestrial habitats.

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