Aquatic Macrophytes at the
Henrys Fork of the Snake River during 2009

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ABSTRACT

Aquatic macrophytes at the Henrys Fork of the Snake River, Idaho provide important habitat for over-wintering juvenile rainbow trout (*Oncorhynchus mykiss*) and macroinvertebrates, including Trichoptera, Ephemeroptera, and Diptera, that are important food resources for rainbow trout. Aquatic macrophytes also provide important components of summer habitat for adult and sub-adult trout, including increased water depths for a given flow, increased channel complexity, decreased water velocity, and overhead concealment. Loss of habitat is one of the factors attributed to the decline in the trout fishery at the Henrys Fork, but no data on aquatic macrophytes has been collected since 2001. Aquatic macrophytes at the Henrys Fork were periodically sampled from 1958 to 2001. Different sampling methods make comparisons of historical results challenging. However, decreasing trends in vegetation abundance have been attributed to influx of large quantities of sediment from draw downs at Island Park Reservoir, extremely high spring flows, low winter flows and associated ice scour, and foraging by trumpeter swans and other waterfowl. The objectives of this study are to: 1) assess the current condition of aquatic macrophytes; 2) compare the current condition of aquatic macrophytes to historical surveys; 3) assess river substrate and river channel cross sections at ten transects within and adjacent to Harriman State Park; and 4) explore preliminary relationships between aquatic macrophytes and physical stream characteristics. During 2009, percent cover of aquatic macrophytes averaged 64% and was significantly lower than 1993, 1994, and 1999. Percent cover of aquatic macrophytes did not significantly differ between 1988 and years following the 1992 sediment release from Island Park Reservoir. Sampling transects were dominated by gravel substrates with the occurrence of sand and silt generally increasing in downstream transects. River bed profiles at Last Chance, Big Bend, and Millionaire’s Pool showed gradual fluctuations between “mounds” and “shallow channels.” Transects at Harriman East were characterized by a “deep” channel where water depths exceeded 132 cm. Water velocity at 80% of the water column was inversely correlated with percent cover of vegetation. However, correlations were weak, likely due to variation in density and morphology of vegetation and biomass of algae among plots. The abundance of aquatic macrophytes in riverine systems is influenced by complex interactions of abiotic and biotic conditions. Quantitative assessments of the abiotic and biotic factors affecting aquatic macrophytes are needed to provide information on the factors affecting aquatic macrophytes at the Henrys Fork. Additional macrophyte surveys are needed to determine if the lower percent cover observed during 2009 is a long-term decreasing trend in aquatic macrophytes or if it is part of the natural variability of aquatic macrophyte life cycles.
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INTRODUCTION

Rainbow Trout, Aquatic Macrophytes, and Macroinvertebrates

The Henrys Fork of the Snake River was a notable fishery as early as the late 19th century (Van Kirk and Benjamin 2000) and is currently an important fishery for wild rainbow trout (Oncorhynchus mykiss) in Idaho (Idaho Department of Fish and Game 2007). According to many anglers, the fishery in the Henrys Fork has declined since the 1970s and has been described to have fewer trout, less robust insect hatches, and a loss of habitat (Henry’s Fork Foundation 2008). The Henry’s Fork Foundation initiated the Caldera Project during 2008, aimed at assessing factors that affect rainbow trout and identifying habitat improvement actions that benefit: 1) over-wintering juvenile trout (age 0); and 2) summer habitat for sub-adult and adult trout within Harriman State Park and adjacent waters.

Over-winter survival of juvenile trout is related to several factors, including winter flows out of Island Park Reservoir (Mitro et al. 2003) and abundance of aquatic macrophytes (Griffith and Smith 1995). Juvenile rainbow trout will conceal themselves in beds of aquatic macrophytes, undercut banks, and submerged stems and leaves of emergent vegetation during the early winter (Reihle and Griffith 1993). Juvenile fish may shift to cobble/boulder habitats during the late winter as aquatic macrophytes decompose and decrease in cover (Simpkins et al. 2000; Griffith and Smith 1995).

The role of aquatic macrophytes in providing cover for adult rainbow trout is less clear. However, important components of summer habitat for sub-adult and adult trout are influenced by aquatic macrophytes. Aquatic macrophytes increase water depths for a given flow (Vinson et al. 1992), provide increased foraging opportunities (Van Kirk and Martin 2000), decrease water velocity (Horvath 2004, Sand-Jensen and Mebus 1996, Gregg and Rose 1982) and increase channel complexity (e.g., variations in stream bed) by trapping fine sediment and particulate matter (Horvath 2004, Gregg and Rose 1982, Barko et al. 1991) and accelerating flows around macrophyte patches compared to within patches (Sand-Jensen and Mebus 1996). Anecdotal evidence suggests that macrophytes provide important cover and increased foraging opportunities on the Henrys Fork, however, the role of aquatic macrophytes in providing summer cover for adult and sub-adult rainbow trout on the Henrys Fork has not been studied (Van Kirk and Martin 2000).

Aquatic macroinvertebrates, including species from the orders Trichoptera (caddisflies), Ephemeroptera (mayflies), and Diptera (true flies), are important food resources for rainbow trout on the Henrys Fork during the summer (Angradi and Griffith 1990). Throughout the world, aquatic macrophytes provide important habitat for macroinvertebrates in large and small rivers (Humphries 1996, Cogerino et al. 1995, Wright 1992, Gregg and Rose 1985). Aquatic macrophytes slow water velocities and provide attachment sites and forage for aquatic macroinvertebrates, and cover from predators. Abundance and species richness of macroinvertebrates is higher in aquatic macrophyte beds than on mineral substrates (Wright 1992, Gregg and Rose 1985) and also varies by species and/or morphology of aquatic vegetation (Cheruvelil et al. 2000, Humphries 1996, Gerrish and Bristow 1979, Krull 1970).

The density of macroinvertebrate taxa that are important food resources for rainbow trout on the Henrys Fork has been correlated to the biomass of aquatic macrophytes in other riverine
systems (Goulart and Callisto 2005, Collier et al. 1999). Therefore changes in the abundance and species composition of aquatic macrophytes on the Henrys Fork likely influences the density of macroinvertebrates. The abundance of macroinvertebrates is also influenced by water chemistry and physical characteristics of river reaches.

**Aquatic Macrophytes at the Henrys Fork**


<table>
<thead>
<tr>
<th>Sampling Method</th>
<th>Year</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Visual estimates</td>
<td>1958</td>
<td>Detailed methods are not known for 1958. For all other years, percent cover was visually estimated for each species and bare ground using a 10 x 10 in (25 x 25 cm) plexiglass sampling quadrat at 25–30 points along 10 transects (Shea et al. 1996, this study).</td>
</tr>
<tr>
<td></td>
<td>1993–1997</td>
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<td></td>
<td>1999</td>
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<td>2001</td>
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<td></td>
<td>2009</td>
<td></td>
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<tr>
<td>Frequency of occurrence</td>
<td>1989</td>
<td>Vegetation was recorded as present or absent at sampling locations; percent cover is estimated by dividing the number of plots with vegetation by the total number of plots sampled (Shea et al. 1996, Vinson et al. 1992, see also Platts et al. 1983). Transects sampled were the same transects used for visual estimates during 1993–2009.</td>
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<tr>
<td></td>
<td>1990</td>
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<td>1991</td>
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<td>1992</td>
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<tr>
<td>Point-intercept</td>
<td>1988</td>
<td>A rectangular frame with 20 grid points was used to sample vegetation at 20 plots along 68 stratified random transects; the species of vegetation (or bare ground) present under each grid point was recorded for each plot (Snyder 1991). This method was repeated by Shea et al. (1996). See Elzinga et al. (2001) for a summary of point-intercepts for measuring cover.</td>
</tr>
<tr>
<td></td>
<td>1994</td>
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<tr>
<td>Wet-weight biomass</td>
<td>1977</td>
<td>Vegetation was collected along 10 transects using a Hess sampler, washed of debris and invertebrates, wrung out by hand, sorted to species, and weighed (Hampton 1981). This method was repeated by Angradi and Contor (1989). During 1977 vegetation was collected with a forceps like-like device that cut and held the vegetation (Shea et al. 1996). Transects differed than those used for visual estimates of percent cover.</td>
</tr>
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<td></td>
<td>1979</td>
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<td>1986</td>
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<td>1987</td>
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</table>
During 1958, the Henrys Fork was described as having an “abundance of food to sustain the present population of trumpeter swans” (Hansen 1959). During 1977, aquatic macrophytes were again described as “prolific” with “dense mats of macrophytes [reaching] the water surface” in several locations (Shea et al. 1996). Productivity of aquatic macrophytes during the mid 1980s (1.77 kg/m² during 1986 and 2.42 kg/m² during 1987) declined between 43 and 59% compared to the late 1970s (4.09 kg/m²), based on wet weight biomass. This decline in aquatic macrophytes followed low-elevation draw downs from Island Park Reservoir during 1979 that mobilized and transported large quantities of sediment into the Henrys Fork (Gregory 2008) and maximum annual flows from Island Park Dam that occurred early in the growing season during May 1982, 1984, and 1986 (Shea et al. 1996).

By the late 1980s, percent cover estimates (77% during 1988) and frequency of occurrence (85% during 1989) suggested that productivity of aquatic macrophytes may have increased from the mid 1980s. However, long-time researchers and fishermen had perceived a decline in macrophytes by the late 1980s (Shea et al. 1996, Paini and Stiehl 1993). Low winter flows and extensive winter ice, followed by an increase in flows to melt ice and free food resources to prevent increased swan mortality during the winter 1988–1989 have been attributed to the perceived decrease in aquatic macrophytes (Shea et al. 1996). Increased winter flows opened ice-free areas for trumpeter swans and other waterfowl; however, increased flows and associated ice scour also ripped up and dislodged aquatic vegetation (Harrop 2004).

Aquatic macrophytes experienced a “massive” decline during the winter of 1989–1990 (Shea et al. 1996) when aquatic macrophytes ranged from 1 to 5% cover on each transect by March 1990. This decline coincided with record high numbers of swans at Harriman State Park during winter 1989–1990 and low flows with ice free conditions that created large areas of suitable foraging habitat for wintering swans, geese, and ducks (Shea et al. 1996). During October 1990, aquatic macrophytes occurred at 67% of sample sites, recovering from the winter decline, but at a frequency below October 1989 (85%).

During 1990–1992, the frequency of aquatic macrophyte occurrence increased from 67% to 83%. Increased productivity was also inferred from increased vegetation height during this time period (12 cm [5 in] during 1990 compared to 28 cm [11 in] during 1992; Shea et al. 1996). Visual estimates of percent cover during 1993–2001 ranged from 67 to 83%, with percent cover significantly higher during 1993 and 1999 than most other years (this study). During March 1995, percent cover of aquatic macrophytes during winter had increased (range < 5% to 65%; Shea et al. 1996) compared to the low estimates during March 1990.

Changes in productivity measures of aquatic macrophytes on the Henrys Fork has been attributed to low winter flows with associated build up of thick ice, high variation between winter and spring flows, influx of silt from draw downs at Island Park Reservoir (e.g., 1966, 1979, and 1992), and winter foraging by trumpeter swans and other waterfowl (Shea et al. 1996). Aquatic macrophytes are also influenced by light, temperature, water depth and chemistry, current velocities and wave action, nutrients, and physical and chemical properties of sediment (Barko et al. 1991, Barko and Smart 1981, Anderson 1978, Westlake 1967).

Aquatic macrophytes on the Henrys Fork were last surveyed during 2001. The current condition of aquatic macrophytes and how it relates to past conditions is not known. This study updates our current understanding of aquatic macrophytes on the Henrys Fork and compares data collected during October 2009 to historical surveys.
Objectives

The objectives of this study are to:


2. Compare the current condition of aquatic macrophytes along on the Henrys Fork of the Snake River to historical surveys.

3. Assess river substrate and river channel cross sections along 10 transects at the Henrys Fork of the Snake River.

4. Explore preliminary relationships between aquatic macrophytes and physical stream characteristics.
STUDY AREA

The Henrys Fork of the Snake River is located in eastern Idaho. Its headwaters lie within the western edge of Yellowstone National Park and along the Continental Divide. Gregory (2008) described seven stream reaches of the Henrys Fork from Island Park Dam to Mesa Falls (Fig. 1). This 28 mile section of the Henrys Fork drains through the “Island Park Caldera”, an area of collapsed volcanoes that was once located over the “hot spot” now currently under Yellowstone National Park. This section of the Henrys Fork encompasses the area referred to as the Caldera Section (Henry’s Fork Foundation 2008).

Water flows in the Caldera Section of the Henrys Fork are primarily controlled by releases from the Island Park Dam as well as inputs from springs and six tributaries: the Buffalo River, Blue Spring Creek, Antelope Park Creek, Big Bend Creek, Thurmon Creek, and Fish Creek. Daily mean discharge rates at the Island Park Dam ranged from 356 to 366 ft³/s (10.1 to 10.4 m³/s) during the sampling period in October 2009 (USGS 2009). Discharge from the spring-fed Buffalo River is estimated at 212 ft³/s (6 m³/s) except during the spring snowmelt period (Mitro et al. 2003). Therefore, total discharge at Harriman State Park during the sampling period was approximately 573 ft³/s (16.2 m³/s). The gradient of the Henrys Fork decreases from Box Canyon downstream to Harriman East (Gregory 2008). Field work was conducted within the Caldera Section of the Henrys Fork from Last Chance to the inlet of Fish Creek. All transects, except ABar, are located within the boundary of Harriman State Park (Fig. 2).

Fig. 1. The Henrys Fork of the Snake River from Island Park Dam to Upper Mesa Falls. Stream reaches (shown in white) are described by Gregory (2008).
Fig. 2. Transects used to sample aquatic macrophytes during 2009 at the Henrys Fork of the Snake River, Idaho.
METHODS

Aquatic Macrophytes

Data Collection

Aquatic macrophytes were sampled during October 13–17, 2009 using methods described by Shea et al. (1996), in order to replicate historical aquatic macrophyte surveys conducted during 1993–2001. Data were collected at 25–30 plots along transects established during 1989 (1A, 1B, 2A, 2B, 3, 4A, 4B, 5A, and 5B) in areas historically used by wintering trumpeter swans and stream reaches suitable for hydraulic modeling (Shea et al. 1996, Vinson et al. 1992; Fig. 2). Transect ABar, established during 1991, was also sampled. Nomenclature for scientific names of aquatic and wetland plants follows Crow and Hellquist (2000a, 2000b).

Transect end points and bearings were re-established based on historical field notes updated during September 2009 (R. Shea, personal communication). Wooden stakes were installed on river left for transects ABar, 1A, 1B, 2A, and 2B and river right for transects 3, 4A, 4B, 5A, and 5B. Wooden stakes were installed on the opposite side of the river channel after sampling was completed for each transect during October 2009.

Sampling plots at each transect were established by estimating the width (m) of the river channel using a range finder and dividing by 30 to determine the distance between sampling points to the nearest 0.5 m (1.6 ft). The width was divided by 30 in order to establish between 25 and 30 sampling plots along each transect, similar to historical survey methods (R. Shea, personal communication). The distance between plots was paced and the number of plots sampled on each transect ranged from 25 to 31 (Table 2). This two-stage sampling design enables efficient estimates of cover using small sampling quadrats while crossing the variability of the population allowing for more precise estimates of means (Elzinga et al. 1998). Using this design, the transect is the primary sampling unit (n = 10).

At each sampling plot, a 25 x 25 cm (10 x 10 in) sampling quadrat was haphazardly placed upstream of the observer’s legs without looking at the vegetation or substrate so as to not bias the sampling plot location. Percent cover of each species present and bare ground were visually estimated to the nearest 5% (nearest 1% if total coverage was < 5%). Vegetation height and water depth were measured to the nearest 1 cm at the center of the upstream side of the quadrat using a ¾ inch PVC pipe marked in 1 cm increments. When a sampling plot contained vegetation covered with periphyton (algae attached to a substrate) percent cover of algae was recorded to the nearest 5% before carefully removing the algae in order to identify aquatic macrophytes underneath. Percent cover estimates of periphyton were made independent of cover estimates for macrophytes and bare ground (e.g., a quadrat with 100% cover algae could have between 0 and 100% cover of macrophytes). Percent cover of bare ground and aquatic macrophytes equaled 100%.

The sampling quadrat was constructed from ¼ inch (0.64 cm) marine grade plywood and ⅛ inch (0.32 cm) plexiglass (Fig. 3). A plywood frame was constructed by forming four lap joints along the edges of two 10.5 x 6 inch (26.7 x 15 cm) pieces of plywood. These pieces were fit onto two shorter 10 x 6 inch (25.4 x 15 cm) pieces of plywood to form a 10 x 10 in (25 x 25 cm) square. Before assembly, a ½ inch (0.32 cm) dado groove was cut 0.25 in (0.64 cm) above the longest edge of the plywood. The plexiglass (10.25 x 10.25 in; 26 x 26 cm) was glued in the
dado groove in the plywood with waterproof glue, forming the bottom of the quadrat. The plywood lap joints were glued and secured with wood screws. Watertight seals were assured by sealing all joints and screw holes with silicone sealant. The quadrat was divided in quarters using a waterproof marker.

Table 2. Channel width, interval between plots, and number of plots sampled for each transect during October 2009 at the Henrys Fork of the Snake River, Idaho.

<table>
<thead>
<tr>
<th>Transect</th>
<th>Channel Width (m)</th>
<th>Interval Between Plots (m)</th>
<th>No. of Plots</th>
</tr>
</thead>
<tbody>
<tr>
<td>ABar</td>
<td>100</td>
<td>3</td>
<td>29</td>
</tr>
<tr>
<td>1A</td>
<td>140</td>
<td>4.5</td>
<td>31</td>
</tr>
<tr>
<td>1B</td>
<td>204</td>
<td>6.5</td>
<td>31</td>
</tr>
<tr>
<td>2A</td>
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<td>7</td>
<td>31</td>
</tr>
<tr>
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<td>3</td>
<td>200</td>
<td>7</td>
<td>30</td>
</tr>
<tr>
<td>4A</td>
<td>120</td>
<td>4</td>
<td>31</td>
</tr>
<tr>
<td>4B</td>
<td>116</td>
<td>4</td>
<td>28</td>
</tr>
<tr>
<td>5A</td>
<td>110</td>
<td>3.5</td>
<td>29</td>
</tr>
<tr>
<td>5B</td>
<td>113</td>
<td>4</td>
<td>25c</td>
</tr>
</tbody>
</table>

Summary Statistics

- Minimum: 113, Interval: 3, No. of Plots: 25
- Maximum: 245, Interval: 8, No. of Plots: 31
- Mean: 155, Interval: NA, No. of Plots: 29
- Total: NA, Interval: NA, No. of Plots: 294

*a Channel width was measured with a range finder.
*b One plot from transect 1A, 1 plot from transect 2A, 3 plots from transect 3, and 1 plot from transect 4A were excluded from analyses due to thick mats of periphyton and phytoplankton (see Analyses section).
*c Four plots in the middle of transect 5B could not be sampled because of water too deep to wade.

Fig. 3. Sampling quadrat/plexiglass view box (25 x 25 cm) and PVC pipe (marked in 1 cm intervals) used to measure percent cover, height of aquatic vegetation, and water depth during October 2009 at the Henrys Fork of the Snake River, Idaho.
**Analyses**

Mean percent cover of aquatic macrophytes was calculated for each species, all vegetation combined, “Group 1” species, “Group 2” species, and over-wintering vegetation along each transect. Percent cover of algae was not included in calculations of total vegetative cover. Six sampling plots with thick masses of periphyton and phytoplankton encompassing the entire water column were not included in analyses because percent cover of rooted vegetation could not accurately be estimated. Plots excluded from analyses included: 1 plot from transect 1A; 1 plot from transect 2A; three plots from transect 3; and 1 plot from transect 4A. Aquatic macrophytes were present within these algal assemblages so it is unlikely that removal of these six plots biased total estimates of vegetative cover. Data on aquatic macrophytes prior to 2009 were summarized from Shea et al. (1996), Shea (1997), Shea (1999), and Shea (2001).

Aquatic macrophytes were grouped into two categories used by Shea et al. (1996) based on growth forms described by Chambers (1987) and Wilcox and Meeker (1991). Growth forms described as tall erect species are classified as Group 1 species and include *Stuckenia* spp. (includes *Potamogeton pectinatus* reported during historical surveys), *Potamogeton richardsonii*, *Elodea canadensis*, and *Myriophyllum* spp. Growth forms described as shorter bottom-dwelling mat-forming species are classified as Group 2 species and include *Callitriche* spp., *Zannichellia palustris*, and *Ranunculus aquatilis*. Shea et al. (1996) further described Group 1 species preferring slow to moderate flows, while Group 2 species can tolerate higher velocity flows. Over-wintering vegetation includes *Myriophyllum* spp. and *R. aquatilis* (Angradi 1991), and *E. canadensis* (Bowmer et al. 1995; R. Shea, personal communication).

All statistical analyses were performed using the statistics module in SigmaPlot 11.0 (Systat Software, Inc. 2008). A general linear model (GLM) analysis of variance (ANOVA) was used to quantify the effect of year on vegetative characteristics (Jager and Looman 1995, ter Braak and Looman 1995). A one-way repeated measures ANOVA was performed to test for differences in percent cover of all vegetation combined, Group 1 species, Group 2 species, and over-wintering species across years with the same sampling methods (1993–1997, 1999, 2001, and 2009). Differences were considered significant at \( P = 0.05 \). When results of the ANOVA were significant, all pairwise multiple comparisons were performed using Tukey adjustments (Neter et al. 1996). Results of the ANOVA cannot be used to draw inferences to the entire Caldera Section of the Henrys Fork because transects were not randomly established.

To explore differences in aquatic macrophytes before and after the 1992 sediment release from Island Park Reservoir, a one-way repeated measures ANOVA was performed to test for differences between 1988 and 1994 point-intercept data. Because additional comparisons of trends among 1988 and subsequent years with different sampling methods (point-intercept vs. visual aerial estimates) were of interest, unpaired t-tests were performed to test for differences in the two sampling methods on cover estimates of all vegetation combined, Group 1 species, and Group 2 species during 1994. Data for Group 2 species did not meet the assumption of equal variance, so the non-parametric Mann-Whitney Rank Sum Test was performed. Mean cover of all vegetation combined (\( t(25) = 0.50, P = 0.62 \)), Group 1 species (\( t(25) = 0.89, P = 0.38 \)), and Group 2 species (\( T = 151, P = 0.60 \)) did not significantly differ between sampling methods during 1994. Therefore, data from 1988 (point-intercept estimates) were compared to visual...
aerial estimates during 1993–1997, 1999, 2001, and 2009 using unpaired t-tests. If data did not meet the assumption of equal variance or were not normally distributed, a non-parametric Mann-Whitney Rank Sum Test was performed.

Average vegetation height and estimates of variance during 2009 and 1989–2001 (excluding non-sampled years) was calculated for each transect (n = 10) when data were available. Elevation of vegetation during 2009 was calculated by adding vegetation height to the river bed elevation at each sampling point. Elevation of vegetation height was included in graphs of river bed profiles. To assess the amount of vegetation encompassing the water column during 1989–2009, mean vegetation height was divided by mean water depth and multiplied by 100 for each transect. Data from each transect were averaged to calculate the percent of the water column occupied by vegetation for each year.

**Physical Characteristics of Transects**

**Data Collection**

Physical characteristics of transects were collected while sampling aquatic macrophytes, requiring three field personnel. Substrate was recorded by randomly placing a finger along the upstream edge of the sampling quadrat. The size of the substrate at that location was recorded as silt (< 2 mm, < 0.08 in), sand (< 2 mm, < 0.08 in), gravel (2.1–64 mm, 0.08–2.5 in), cobble (64.1–256 mm, 2.51–10 in), or boulder (> 256.1 mm, >10.1 in).

Water velocity (ft/s) was measured at the same location as vegetation height and water depth. Water velocities were collected at every sampling plot for the ABar transect and every other sampling plot for all other transects due to time constraints. Velocities were measured at 20, 50, and 80% of the water column using a Marsh-McBirney Flo-Mate Portable Velocity Flow Meter and a USGS topsetting wading rod. The velocity meter was held directly into the direction of flow and the data averaging function of the meter was used for calculating average velocity over a 10 second interval. Velocities were not collected when sampling plots were located within very thick algal masses of periphyton and phytoplankton because the meter could not be positioned within the masses.

Transect cross sections were measured using TopCon GPS-based survey equipment, including two HiPer GA receivers for a RTK base and rover. GPS coordinates (latitude, longitude, and elevation [ft]) were measured at the center of the upstream edge of every sampling plot (same location as water depth and vegetation height). GPS coordinates were also measured at transect end-points, bank slope breaks, and edges of water along river left and river right (looking downstream).

Only one known survey monument was in the vicinity of the transects, and due to time constraints we were not able to tie all transects into one base station. Therefore, GPS coordinates collected from different base stations have a 5–10 m ellipse error. Measurements taken from any one base station are accurate to 3 cm relative to each other. Base stations, transects, and transect end-point GPS coordinates are listed in Table 3. GPS coordinates were downloaded using Topcon Link v.7.2.3. Files were then converted to comma separated files and ArcGIS shapefiles for analyses. Each transect was completed using one base station set up, with the exception of transect 5A. Assuming the water level was constant, the elevation of plots 1–17 on transect 5A
were adjusted by 6.5 ft (2 m) to correspond to elevations taken from Base Station 6. Latitude and longitude measures for plots along transect 5A taken from the two different base stations appeared relatively accurate to each other.

**Table 3. GPS survey equipment (TopCon HiPer GA) RTK base stations used to measure cross sections along each transect and GPS coordinates (latitude and longitude) of transect end-points measured during October 2009 at the Henrys Fork of the Snake River, Idaho.**

<table>
<thead>
<tr>
<th>Base Station</th>
<th>Transect</th>
<th>Transect End-Point (River Left)</th>
<th>Transect End-Point (River Right)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Latitude</td>
<td>Longitude</td>
</tr>
<tr>
<td>1</td>
<td>ABar</td>
<td>44°21'55.46517&quot; N</td>
<td>-111°24'05.63861&quot; W</td>
</tr>
<tr>
<td>2</td>
<td>1A</td>
<td>44°21'27.43455&quot; N</td>
<td>-111°26'21.31637&quot; W</td>
</tr>
<tr>
<td>2</td>
<td>1B</td>
<td>44°21'31.18496&quot; N</td>
<td>-111°26'35.66106&quot; W</td>
</tr>
<tr>
<td>2</td>
<td>2A</td>
<td>44°21'25.49866&quot; N</td>
<td>-111°26'45.78092&quot; W</td>
</tr>
<tr>
<td>2</td>
<td>2B</td>
<td>44°21'22.56052&quot; N</td>
<td>-111°26'46.53681&quot; W</td>
</tr>
<tr>
<td>3</td>
<td>3 (plots 1-19)</td>
<td>No data</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>4A</td>
<td>44°19'02.18074&quot; N</td>
<td>-111°26'26.42661&quot; W</td>
</tr>
<tr>
<td>4</td>
<td>4B</td>
<td>44°19'04.38476&quot; N</td>
<td>-111°26'13.96180&quot; W</td>
</tr>
<tr>
<td>5</td>
<td>5A (plots 1-17)</td>
<td>n/a</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>5A (plots 18-29)</td>
<td>44°18'37.77689&quot; N</td>
<td>-111°26'04.06301&quot; W</td>
</tr>
<tr>
<td>6</td>
<td>5B</td>
<td>44°18'26.17892&quot; N</td>
<td>-111°26'06.24766&quot; W</td>
</tr>
</tbody>
</table>

**Analyses**

The frequency of each substrate types was calculated for each transect by dividing the number of plots for each substrate type by the total number of plots sampled. Mean water velocity at 20, 50, and 80% of the water column was calculated for each transect. Spearman Rank Order Correlations were performed to examine the relationship between water velocity at 80% of the water column and percent cover of vegetation. Linear regression lines were graphed for each transect.
RESULTS

Species Composition

During October 2009, we observed 10 genera from 9 different families of aquatic and wetland plants and 1 genus of plant-like algae (Table 4). All submerged aquatic macrophyte genera and/or species, with the exception of Callitriche spp. and Myriophyllum hippuroides, were observed on all transects. Callitriche spp. was observed on all transects except ABar. M. hippuroides was only observed at Harriman East in 6 plots on transects 4A and 5A. Due to the similarities of species in the genus Myriophyllum and the limited occurrence of M. hippuroides, it is included in Myriophyllum spp.

In the 288 sampled plots used for analyses, Stuckenia spp. was observed in 61% (n = 176) of plots, followed by Ranunculus aquatilis in 59% (n = 169) of plots. Other commonly observed species included Zannichellia palustris (47%; n = 135), Callitriche spp. (47%; n = 135), Elodea canadensis (41%; n = 119), Potamogeton richardsonii (25%; n = 71), and Myriophyllum spp. (24%; n = 69). Eleocharis sp., Alisma sp., Nitella sp., and Lemna trisulca all occurred in less than 7% (n ≤ 18) of plots.

At least two kinds of macroscopic periphyton were observed attached to aquatic plants and gravel/cobble substrates (Fig. 4). Long strands of green filamentous algae were observed on 197 (68%) of the sample plots. An epiphytic globular algae, brown in color was observed on 76 (26%) of sample plots and primarily occurred on vegetation along the shallowly-flooded edges of the river bed. Large thick floating masses of periphyton and phytoplankton extending through the entire water column were observed at six sample plots.

Fig. 4. Long strands of filamentous green algae on Elodea canadensis (left), and globular algae, brown in color (right), attached to submerged aquatic macrophytes and/or gravel/cobble substrates during October 2009 at the Henrys Fork of the Snake River, Idaho.

<table>
<thead>
<tr>
<th>Scientific Name</th>
<th>Notes with Common Names</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Submerged Aquatic Vegetation</strong></td>
<td></td>
</tr>
<tr>
<td>Potamogetonaceae (pondweed family)</td>
<td></td>
</tr>
<tr>
<td><em>Stuckenia</em> spp.</td>
<td>Long, narrow-leaved pondweeds with no floating leaves; includes <em>S. pectinata</em> (sago pondweed), <em>S. filiformis</em> (slender-leaved pondweed), and <em>S. vaginata</em> (sheathing pondweed). <em>S. pectinata</em> is synonymous with <em>Potamogeton pectinatus</em>, sago pondweed historically reported from the Henrys Fork during 1958–2001. No <em>Stuckenia</em> plants observed during October 2009 had fruits or flowers present to positively identify to the species level.</td>
</tr>
<tr>
<td><em>Potamogeton richardsonii</em></td>
<td>Richardson’s pondweed</td>
</tr>
<tr>
<td>Zannichelliaeae (horned pondweed family)</td>
<td></td>
</tr>
<tr>
<td><em>Zannichellia palustris</em></td>
<td>Horned pondweed</td>
</tr>
<tr>
<td>Ranunculaceae (buttercup family)</td>
<td></td>
</tr>
<tr>
<td><em>Ranunculus aquatilis</em></td>
<td>White water-buttercup, water crowfoot</td>
</tr>
<tr>
<td>Callitrichaceae (callitriche family)</td>
<td></td>
</tr>
<tr>
<td><em>Callitrichia</em> spp.</td>
<td>Water-starwort; includes <em>C. hermaphroditica</em> (northern or autumnal water-starwort), <em>C. heterophylla</em> (different-leaved water-starwort), and <em>C. verna</em> (spring water-starwort). <em>C. verna</em> and <em>C. hermaphroditica</em> were both historically reported from the Henrys Fork. Both of these species as well as plants with no fruits were observed during 2009.</td>
</tr>
<tr>
<td>Haloragaceae (water milfoil family)</td>
<td></td>
</tr>
<tr>
<td><em>Myriophyllum</em> spp.</td>
<td>Includes <em>M. sibiricum</em> (northern milfoil; synonymous with <em>M. exalbescens</em>) and <em>M. hippuroides</em> (western milfoil) observed during 2009. Both species, as well as <em>M. quitense</em> (Andean milfoil), were historically reported from the Henrys Fork; <em>M. verticillatum</em> has also been reported from Idaho. Individuals with no flowering stalks during 2009 limited identification of all plants to the species level.</td>
</tr>
<tr>
<td>Hydrocharitaceae (tape-grass family)</td>
<td></td>
</tr>
<tr>
<td><em>Elodea canadensis</em></td>
<td>Common waterweed</td>
</tr>
<tr>
<td><strong>Free-floating Aquatic Vegetation</strong></td>
<td></td>
</tr>
<tr>
<td>Lemnaceae (duckweed family)</td>
<td></td>
</tr>
<tr>
<td><em>Lemna trisulca</em></td>
<td>Star duckweed or ivy duckweed. Observed along the shallow river margins of transects 1B and 2A and in the middle of transect 5A.</td>
</tr>
<tr>
<td><strong>Emergent Vegetation</strong></td>
<td></td>
</tr>
<tr>
<td>Alismataceae (water-plantain family)</td>
<td></td>
</tr>
<tr>
<td><em>Alisma</em> sp.</td>
<td>Includes <em>A. triviale</em> , also referred to as <em>A. plantago-aquatica</em> (northern water plantain), and <em>A. gramineum</em> (narrow-leaf water plantain). Only observed along shallow edges of the river bed. <em>Alisma</em> spp. are similar to other basal ribbon leaved species such as <em>Sagittaria</em>. Observation of seeds confirmed the genus <em>Alisma</em> was present on the Henrys Fork during 2009.</td>
</tr>
<tr>
<td>Cyperaceae (sedge family)</td>
<td></td>
</tr>
<tr>
<td><em>Eleocharis</em> sp.</td>
<td>Spikerush; only observed along shallow edges of the river bed</td>
</tr>
<tr>
<td><strong>Plant-like Algae</strong></td>
<td></td>
</tr>
<tr>
<td>Characeae (stonewort family)</td>
<td></td>
</tr>
<tr>
<td><em>Nitella</em> sp.</td>
<td>Brittlewort. It’s similar to <em>Chara</em> sp. (muskgrass), another plant-like alga.</td>
</tr>
</tbody>
</table>
Percent Cover of Aquatic Macrophytes

Percent cover of most aquatic macrophytes varied considerably among transects and plots within a transect (Fig. 5). *Zannichellia palustris* and *Ranunculus aquatilis* showed the highest variation in percent cover among transects. Mean percent cover of *Z. palustris* ranged from 2 to 36%, and mean percent cover of *R. aquatilis* ranged from 2 to 32%. Mean percent cover of *Myriophyllum* spp. ranged from 0.5 to 25%, but cover on 9 of 10 transects averaged ≤ 14%. Mean percent cover of *Elodea canadensis* ranged from 0.4 to 9% from transect ABar downstream to transect 3; mean percent cover at Harriman East transects (4A, 4B, 5A, 5B) ranged from 16 to 23%. Mean percent cover of *Callitriche* spp. ranged from 1 to 12% on transects where it occurred. Mean percent cover of *Stuckenia* spp. varied from 17% on transect ABar to 3% on transect 1B, but when compared to other aquatic macrophyte groups percent cover of *Stuckenia* spp. was relatively consistent among transects averaging between 9 and 13% for 8 of 10 transects. Mean percent cover of *Potamogeton richardsonii* was also relatively consistent averaging between 0.2 and 3.5% for all transects.

Percent cover of all vegetation combined for all transects (n = 10) averaged 64% ± 11 (SD; range 41–81%). Species with the highest mean percent cover included *R. aquatilis* (14% ± 11 SD), *Z. palustris* (14% ± 12 SD), *Stuckenia* spp. (11% ± 4 SD), and *E. canadensis* (11% ± 9 SD; Fig. 6). Percent cover of *Myriophyllum* spp., *Callitriche* spp., and *P. richardsonii* along each transect averaged ≤ 6% for each species.

Mean percent cover of Group 1 (30% ± 12 [SD], range 5–45%) and Group 2 (33% ± 18 SD, range 13–50%) species were similar (Fig. 6). Species known to over-winter at the Henrys Fork, *Myriophyllum* spp., *E. canadensis* and *R. aquatilis*, averaged 31% ± 16 (SD; range 14–52%) among transects.
Fig. 5. Percent cover of aquatic macrophytes during October 2009 at the Henrys Fork of the Snake River, Idaho. Boxes represent 25 and 75% quartiles, whiskers represent 10 and 90% quartiles, and solid dots represent outliers; median percent cover is the solid horizontal line within boxes and mean percent cover is the dotted line within boxes. *Potamogeton richardsonii*, a species included in Group 1 (Shea et al. 1996) is not shown because percent cover an all transects averaged $\leq 3\%$. 
Fig. 6. Mean percent cover ± SD of seven species of aquatic macrophytes, bare ground, all vegetation combined, group 1 and group 2 species, and over-wintering species for all transects (n = 10) sampled during October 2009 at the Henrys Fork of the Snake River, Idaho. Group 1 species include *Stuckenia* spp., *Potamogeton richardsonii*, *Myriophyllum* spp., and *Elodea canadensis*; Group 2 species include *Ranunculus aquatilis*, *Zannichellia palustris*, and *Callitriche* spp. (Shea et al. 1996). Over-wintering species include *Myriophyllum* spp. and *R. aquatilis* (Angradi 1991), and *E. canadensis* (Bowmer et al. 1995; R. Shea, personal communication).
Vegetation Height

Overall vegetation height (n = 10 transects) averaged 14.9 cm ± 2.8 SD; (5.9 in ± 1.1 SD; Fig. 7) and occupied 27% of the water column. Vegetation height rarely exceeded 40 cm (15.7 in) during 2009. Tall stands of aquatic macrophytes were observed at Harriman East, reaching up to 80 cm (31.5 in).

Fig. 7. Vegetation height along 10 transects during October 2009 at the Henrys Fork of the Snake River, Idaho. Boxes represent 25 and 75% quartiles, whiskers represent 10 and 90% quartiles, and solid dots represent outliers; median vegetation height is the solid horizontal bar and mean vegetation height is the dotted horizontal line within boxes.
Percent Cover of Periphyton

Percent cover of periphyton varied considerably among transects and plots within a transect (Fig. 8). Percent cover of algae averaged 51% ± 20 (SD) across all transects and ranged from 22% ± 28 (SD) on transect ABar to 81% ± 26 (SD) on transect 4A. Upstream of Silver Lake Outlet, percent cover of algae averaged 39% compared to 68% at Harriman East.

Fig. 8. Mean ± SD of periphyton during October 2009 at the Henrys Fork of the Snake River, Idaho.
Comparison with Recent Historical Aquatic Macrophyte Data

Species Composition and Percent Cover 1993–2009

All of the primary species and/or genera reported from historical surveys of the Henrys Fork were observed during October 2009. Three species, infrequently observed during historical surveys were not observed during 2009. *Myriophyllum quitense*, observed in the Henrys Fork during 1988 (Angradi 1991) and 1995 (Shea et al. 1996) was not observed during this study, however the lack of flowering stalks present during October 2009 prevented identification of most *Myriophyllum* to the species level. *Potamogeton pusillus*, observed during 1993, and *Chara* sp., a plant-like algae observed during 1997, were also not observed during this study. Another similar plant-like alga, *Nitella* sp., not previously reported from the Henrys Fork, was observed during 2009.

Mean percent cover of all vegetation combined significantly differed between years (*P* < 0.001; Fig. 9). Percent cover of all vegetation was significantly lower during 2009 (64% ± 4 SE) and 1995 (67% ± 3 SE) than 1993 (83% ± 3 SE; *P* ≤ 0.001) and 1999 (83% ± 2 SE; *P* ≤ 0.007). Percent cover was also lower during 1996 (71% ± 4 SE; *P* = 0.03) than 1999. Percent cover during 2009 did not significantly differ from percent cover during 1994–1997 and 2001.

Mean percent cover of Group 1 species also significantly differed between years (*P* < 0.001; Fig. 10). Percent cover of Group 1 species was significantly lower during 2009 (30% ± 4 SE) than 1993 (59% ± 5 SE; *P* < 0.001), 1994 (47% ± 5 SE; *P* = 0.002), 1995 (44% ± 5 SE; *P* = 0.03), 1996 (46% ± 4 SE; *P* = 0.006), and 1999 (53% ± 4 SE; *P* < 0.001). In addition to differing from 2009, percent cover of Group 1 species during 1993 was also higher than 1995 (*P* = 0.03), 1997 (42% ± 4 SE; *P* = 0.008), and 2001 (37% ± 2 SE; *P* < 0.001). Mean percent cover of Group 2 species did not differ among years (*P* = 0.06; Fig. 10).

Mean percent cover of over-wintering species significantly differed between years (*P* < 0.001; Fig. 10). Percent cover of over-wintering species during 2001 (25% ± 4 SE) was significantly lower than 1993 (39% ± 5 SE, *P* = 0.04), 1997 (41% ± 5 SE, *P* < 0.001), and 1999 (38% ± 5 SE, *P* = 0.01). Percent cover during 1997 was also higher than 1995 (28% ± 4 SE, *P* = 0.02). Percent cover of over-wintering species during 2009 did not significantly differ from other years.

Percent Cover Before and After 1992 Sediment Release

Percent cover estimates of total vegetation between 1988 (77% ± 6 [SE]) and 1994 (79% ± 4 [SE]) did not statistically differ (*P* = 0.7). Percent cover during 1988 did not significantly differ from visual cover estimates following the 1992 sediment release (i.e., 1993–1997, 1999, 2001, and 2009; Fig. 9). Percent cover Group 1 species, Group 2 species, and overwintering species during 1988 was similar to cover estimates during 1993 (Fig. 10).
Vegetation Height 1989–2009

During 1989–2009 (excluding non-sampled years), mean vegetation height for all transects combined peaked during 1989 at 38 cm compared to 11 cm during 1995 (Fig. 11). Mean vegetation height during 2009 was the 4th lowest average height reported for 12 years of sampling during 1989–2009. Aquatic macrophytes occupied up to 71% of the water column during 1989 and as low as 20% of the water column during 1995 (Fig. 12). During 2009, vegetation height was the 3rd lowest percentage of the water column (27%) reported for 12 years of sampling during 1989–2009.

Fig. 9. Mean percent cover ± SE of all vegetation combined during 1988–1997, 1999, 2001, and 2009 at the Henrys Fork of the Snake River. Historical data (1988–2001) are summarized from Shea et al. (1996), Shea (1997), Shea (1999), and Shea (2001). Sampling methods for point intercept cover estimates followed Snyder (1991); visual aerial cover estimates and presence/absence estimates followed Shea et al. (1996). Transects 2A and 4A were not sampled during 1993 and transect ABar was not sampled during 1988–1990. Years of visual aerial estimates of cover with different letters are significantly different (df = 7, F = 6.2, P < 0.05). Point intercept cover estimates did not significantly differ between 1988 and 1994 or from other years, with the exception that 1994 point-intercept estimate of cover was higher than the 2009 visual aerial estimate of cover (t(25) = 2.4, P = 0.02). Standard error estimates are not available for 1989. Years with presence/absence data (1989–1992) were not statistically analyzed.
Fig. 11. Mean vegetation height ± SE and mean water depth ± SE for 10 transects during October 1989–1997, 1999, 2001, and 2009 at the Henrys Fork of the Snake River, Idaho. Estimates of SE are not available for 1989. Discharge flows at Island Park Dam (USGS gauge station 13042500) were averaged for the sampling period each year. For years with no vegetation sampling, discharge flows were averaged from October 7–11.

**Physical Characteristics of Transects**

**Substrate**

Gravel substrates dominated on transects ABar, 1A, 1B, 2A, and 5A occurring at \( \geq 60\% \) of plots sampled on each transect (Fig. 13). Gravel substrates occurred on approximately half of the plots (48–52\%) on transects 2B and 5B and less than 41\% of plots on transects 3, 4A, and 4B. The occurrence of sandy substrates increased from transect 1B (23\%) downstream to transect 3 (53\%). Sandy substrates recorded on transects at Harriman East occurred on 43\% (4A), 50\% (4B), 17\% (5A) and 40\% (5B) of plots sampled on each transect. Silt was recorded on 5 of 10 transects and occurred on 6\% (1B), 20\% (4A), 4\% (4B), 7\% (5A), and 12\% (5B) of plots sampled on each transect. Silt and sand substrates were not recorded on the most upstream transects sampled, ABar and 1A. Boulder and cobble substrates were recorded on transects ABar, 1A, and 4B and occurred on < 17\% of plots sampled on those transects.

![Fig. 13. Frequency of occurrence of silt (< 2 mm; < 0.08 in), sand (< 2 mm; < 0.08 in), gravel (2.1–64 mm; 0.08–2.5 in), cobble (64.1–256 mm; 2.5–10 in), and boulder (>256 mm; > 10 in) substrates during October 2009 at the Henrys Fork of the Snake River, Idaho.](image-url)
**Water Velocity**

Velocity of water generally decreased from the top to the bottom of the water column (Fig. 14 and Fig. 15). For all transects combined, water velocity averaged 1.25 ft/s (0.38 m/s) at the top (20%) of the water column, 0.93 ft/s (0.28 m/s) at the middle of the water column, and 0.29 ft/s (0.088 m/s) at the bottom (80%) of the water column. Mean top of water column flow on the 3 most upstream transects (ABar, 1A, and 1B) ranged from 1.90 to 1.44 ft/s (0.58 to 0.44 m/s). Mean top of water column flow on all other transects except 2B ranged from 0.93 to 1.23 ft/s (0.28 to 0.37 m/s). Top of water column flow on transect 2B averaged 0.75 ft/s (0.23 m/s).

Water velocity at the bottom (80%) of the water column was negatively correlated with percent cover of aquatic macrophytes for four of the ten transects sampled (Fig. 16 and Fig. 17). Transects 1B \( r_s = -0.564, P = 0.02 \), 2B \( r_s = -0.572, P = 0.03 \), 4A \( r_s = -0.630, P = 0.01 \), and 5B \( r_s = -0.604, P = 0.04 \) showed significant, but weak inverse relationships between percent cover and water velocity. Other transects also showed inverse relationships between water velocity and aquatic macrophytes, but correlations were not significant.

**Transect cross sections**

River bed profiles at Last Chance (ABar; Fig. 18), Big Bend (1A, 1B, 2A, 2B; Fig. 20), and Millionaire’s Pool (3; Fig. 19) had 1 or 2 areas characterized by elevations with gradual fluctuations between “mounds” and “shallow channels.” Harriman East transects (4A, 4B, 5A, and 5B) had a “deep” channel toward the middle or left bank (Fig. 21 and Fig. 22). Maximum water depth within the deep channels at Harriman East transects ranged from 92 to > 132 cm (36 to > 52 in). Maximum water depths on transects upstream from the Outlet of Silver Lake ranged from 51 to 75 cm (20 to 30 in).
Fig. 14. Water velocity measured at 20% (uppermost dots), 50% (middle dots), and 80% (lower dots) of the water column along six transects upstream from Silver Lake Outlet during 2009 at the Henrys Fork of the Snake River, Idaho. Velocities were not measured at plots 15 and 19 on transect 3 because of thick algae.
Fig. 15. Water velocity measured at 20% (uppermost dots), 50% (middle dots), and 80% (lower dots) of the water column along four transects at Harriman East during 2009 at the Henrys Fork of the Snake River, Idaho. Velocities were not measured at plot 19 on transect 4A because of thick algae; velocities were not measured at plots 13–18 on transect 5B because the water too deep to wade.
Fig. 16. Water velocity at 80% of the water column and percent cover of vegetation at sampling plots along six transects upstream from Osborne Bridge during 2009 at the Henrys Fork of the Snake River, Idaho. Spearman Rank Order Correlation Test results and the linear regression line are shown for each transect. Statistically significant results ($P < 0.05$) are in bold.
Fig. 17. Water velocity measured at 80% of the water column and percent cover of vegetation at sampling plot along Harriman East transects during 2009 at the Henrys Fork of the Snake River, Idaho. Spearman Rank Order Correlation Test results and the linear regression line are shown for each transect. Statistically significant results ($P < 0.05$) are in bold.
Fig. 18. River bed, vegetation height, and water surface elevations (ft) along transect ABar during October 2009 at the Henrys Fork of the Snake River, Idaho. Vegetation height and water surface elevations were calculated by adding measured vegetation height and water depth to river bed elevations. LEW = left edge of water and REW = right edge of water looking downstream. Zero is at the left edge of water.

Fig. 19. River bed, water surface, and vegetation height elevations along transect 3 during October 2009 at the Henrys Fork of the Snake River, Idaho. Vegetation height and water surface elevations were calculated by adding measured vegetation height and water depth to river bed elevations. RB = right river bank looking downstream. Data from the LB of transect 3 was not collected due to problems with the equipment. Zero is at the right edge of water.
Fig. 20. River bed, vegetation height, and water surface elevations along transects 1A, 1B, 2A, and 2B during October 2009 at the Henrys Fork of the Snake River, Idaho. Vegetation height and water surface elevations were calculated by adding measured vegetation height and water depth to river bed elevations. LB = left river bank and RB = right river bank looking downstream. Zero is at the left edge of water.
Fig. 21. River bed, vegetation height, and water surface elevations along transects 4A and 4B during October 2009 at the Henrys Fork of the Snake River. Vegetation height and water surface elevations were calculated by adding measured vegetation height and water depth to river bed elevations. LB = left river bank and RB = right river looking downstream. Zero is at the right edge of water.
Fig. 22. River bed, vegetation height, and water surface elevations along transects 5A and 5B during October 2009 at the Henrys Fork of the Snake River. Vegetation height and water surface elevations were calculated by adding measured vegetation height and water depth to river bed elevations. LB = left river bank and RB = right river looking downstream. Data from the middle of transect 5B was not collected because the water was too deep to wade. Zero is at right edge of water.
DISCUSSION

Overall Trends of Aquatic Macrophytes

Variability and Inference

The distribution of aquatic macrophytes was highly variable within transects as well as among transects. This observation is similar to differences in wet weight biomass of aquatic macrophytes among transects (Hampton 1981) and the clustered distribution of macrophyte species (Snyder 1991) for the same study area. The variable and clustered distribution of aquatic macrophytes supports establishing a stratified random sampling design in order to infer results to the target population of interest (the Henrys Fork from Last Chance to Harriman East).

Results from this study and 1989–2001 should not be inferred to the entire reach of the Henrys Fork from Last Chance to Harriman East because transects were not randomly distributed. However, because transects were placed in areas historically used by foraging trumpeter swans and other waterfowl, these surveys will likely detect changes in vegetation resulting from biological impacts of foraging waterfowl, as well as from physical characteristics of sampled stream reaches (see Vinson et al. 1992).

Percent cover of aquatic macrophytes prior to 1988 cannot directly be compared to more recent data due to differences in sampling methods. Percent cover estimates from 1958 can provide a general historical baseline to compare with later studies. However, bare ground was not recorded during 1958 so cover estimates of aquatic macrophytes may be high compared to recent estimates because it is unlikely (although not impossible) that aquatic vegetation covered 100% of each sample. Percentage estimates of vegetation based on wet-weight biomass also do not provide a measure of bare ground.

Recent Historical Trends 1988–2009

Statistical analyses of aquatic macrophyte cover during 1988 compared to estimates of cover following the sediment release during 1992 did not detect any significant differences. However, the statistical power comparing data from two different sampling methods during 1994 and from the same methods during 1988 and 1994 was low (power = 0.05) indicating that it is less likely to detect a difference when one actually exists. Despite low statistical power, cover estimates from 1988 to 1994 were similar and relatively high, ranging from 77 to 85%, with the exception of 1990 when aquatic macrophyte cover averaged 67%. The lower cover during 1990 was likely a result of increased wintering trumpeter swans and increased availability of optimal foraging habitat (e.g. shallow water and ice-free conditions) during the preceding winter (Shea et al. 1996).

During 1989 aquatic macrophytes on the Henrys Fork were described as “dense” and averaged 85% cover (Vinson et al. 1992). Experienced researchers on the Henrys Fork noted that the abundance of aquatic macrophytes during the early 1990s appeared to be much less than during 1988 (Shea et al. 1996). If the subjective observation of decreased abundance is correct, it appears that cover estimates did not detect changes between 1988 and the early 1990s. Changes in total abundance of aquatic macrophytes may not be detected by percent cover
estimates, however, compared to density and frequency, percent cover estimates are the most closely related to biomass or annual production (Elzinga et al. 2001).

Percent cover of aquatic macrophytes during 2009 did not significantly differ from 4 of the 7 years of sampling during 1993–2001, but was significantly lower than 1993, 1994, and 1999. Due to a gap in data from 2002–2008, it cannot be determined if the aquatic macrophytes are experiencing a long-term decline or if the current abundance is part of the natural variability (e.g. cyclical) of aquatic macrophytes on the Henrys Fork.

Differences may also be related to observer bias and/or annual variation in phenology of vegetation. Percent cover of aquatic macrophytes during 2009 was approximately 20% lower than 1993 and 1999. Some authors suggest that differences in cover estimates need to be greater than 20–25% before they can be attributed to factors other than observer bias and annual variation (Kennedy and Addison 1987, Hope-Simpson 1940). The 20% difference observed between this study and peak estimates during 1993 and 1999 may, in part, be attributed to observer bias. However, estimates of cover are most variable at moderate levels (40–60%; Hatton et al. 1986) and the changes detected in this study were between 84 and 64%.

Differences in cover may be related to differences in phenology of vegetation between years. Timing of peak growth and level of senescence by October likely varies between years depending on environmental conditions (e.g. temperature, timing of peak spring flows, etc). Biomass of aquatic macrophytes at the Henrys Fork generally peaked during September–October 1987 and 1988, but also varied by species and year (Angradi 1991). For example, Ranunculus aquatilis maintained relatively high biomass during September and October 1988, whereas biomass of Zannichellia palustris dropped by almost 40% from September to October 1987. Myriophyllum quitense showed a bimodal peak with the highest biomass in June 1988 and another smaller peak during October of both years (Angradi 1991). Most species also exhibited a rapid decline in biomass following peak estimates, suggesting that cover estimates late in the growing season may be more susceptible to variation in annual growth patterns. To reduce the variation related to phenology of vegetation, cover measurements should be made during the same stage of the growing season, which will probably not occur on the same calendar date due to variation in annual weather (Elzinga et al. 2001).

Peak growing season during 2009 may have been earlier than in other years. For example, vegetation rarely reached the water surface during October 2009. However, dense mats of vegetation were observed on the water surface at Millionaire’s Pool (near the Ranch Houses) during mid-August 2009. By October the vegetation along transect 3 at Millionaire’s Pool had senesced, occupying on average 28% (range 9–56%) of the water column.

**Trends by Species’ Groups**

Since 1988, it appears that the cover of Group 1 species has gradually declined, likely due to declining cover estimates of *Elodea canadensis* and *Stuckenia* spp. (Fig. 23). A peak in Group 1 during 1997 may be the result of increasing cover of *Myriophyllum* spp. from 1993 to 1997. No significance difference in the abundance of Group 1 species during 2009 compared to 2001 suggests that the decline of Group 1 species may have stabilized. However, any trend analysis should be considered cautiously due to the data gap between 2002 and 2008.

Species within each of the growth form groups have different morphologies, habitat niches, and palatability to waterfowl. Aquatic macrophytes with different plant morphologies (e.g. finely dissected leaves versus broad leaves) have different abundances of macroinvertebrates (Cheruvelil et al. 2000). Individual species also have different affects on river processes (e.g., current velocity, sedimentation patterns). Due to this variability, species within a group likely respond differently to environmental factors and have different effects on biological processes. Therefore, trends of species and/or closely related species (e.g., fine-leaved pondweeds) should also be considered in the analyses of river processes.

**Trends by Species**

**Stuckenia spp.**

It appears that the abundance of *Stuckenia* spp. (includes *Potamogeton pectinatus* reported throughout historical surveys of the Henrys Fork) has gradually declined from 1958 to 2009 (Fig. 23). Despite the limited sample size during 1958, *Stuckenia* spp. was described as widespread and “abundant,” averaging 40% (Hansen 1959). Percent cover of *Stuckenia* spp. averaged 16–26% during the mid to late 1990s (Shea 1999, Shea et al. 1996). *Stuckenia* spp. was widespread during this study occurring in 61% of sampled plots; however, percent cover only averaged 10%. This decline (30%) is greater than the 20–25% difference that can be attributed to observer bias (Kennedy and Addison 1987, Hope-Simpson 1940).

*Stuckenia pectinata* is tolerant of a wide range of abiotic conditions and its occurrence and biomass in wetlands, shallow eutrophic lakes, and rivers have been related to many factors. *S. pectinata* competes well with *Myriophyllum sibiricum* (synonymous with previously reported *M. exalbescens*; Moen and Cohen 1989), a species that shares a similar niche (French and Chambers 1996), so its occurrence on the Henrys Fork is likely related to abiotic conditions, waterfowl herbivory, or a combination of both.

*S. pectinata* was most abundant at current speeds between 0.0–0.2 m/s (0.0-0.7 ft/s), water depths < 1.5 m (4.9 ft), and in silt sediments at the Nechako River in British Columbia (French and Chambers 1996). The biomass of *S. pectinata* at Badfish Creek in Wisconsin did not significantly differ between substrates, but tended to be higher in silt and gravel substrates compared to sand (Madsen and Adams 1989). Biomass significantly differed among stream reaches and correlations with individual physical stream attributes were weak, suggesting that *S. pectinata* responds to many environmental factors (Madsen and Adams 1989). Other substrate characteristics, including organic matter (and the associated availability of nitrogen released during the mineralization of organic matter), available soil phosphorus, and available soil potassium are important factors affecting distribution and/or biomass of *S. pectinata* (van Wijck et al. 1992, Anderson 1978).

Germination and early growth of *S. pectinata* is primarily affected by spring water temperature and to a lesser degree light (Scheffer et al. 1992, Madsen and Adams 1988, Spencer 1986). Lack of calcium and/or magnesium in solution may also inhibit or reduce the growth of
S. pectinata (Barko et al. 1986). Increased current velocity over the range of 0.17–0.73 m/s (0.56–2.4 ft/s) during in situ experiments at the Pembina River in Alberta coincided with decreased biomass of S. pectinata regardless of sediment type (Chambers et al. 1991). In greenhouse studies, the growth rate of S. pectinata from small tubers (< 10 mg) was less than from larger tubers (> 31 mg) and biomass decreased with increased planting depth and decreased tuber size (Spencer 1987). Therefore, abundance of S. pectinata at the Henrys Fork may negatively influenced by large quantities of sediment from draw downs of Island Park Reservoir and/or high current velocities during the early to mid–growing season. However, percent cover of Stuckenia spp. did not appear to be affected by the 1992 sediment release from Island Park Reservoir (Fig. 23).

Waterfowl herbivory can also impact submerged aquatic macrophyte populations, particularly in shallow sheltered areas (Idestam-Almquist 1998). S. pectinata tubers are an important food source for trumpeter swans in the Greater Yellowstone Area where tubers accounted for 23.4% of their diet during the winter (Squires 1995). Biomass of S. pectinata was reduced by 17% in the Lauwersmeer, a man-made lake in The Netherlands (Van Wijk 1988), and density was reduced 10–80 times in the Baltic Sea (Idestam-Almquist 1998) by foraging waterfowl. S. pectinata may exhibit compensatory growth responses to foraging by waterfowl. Trumpeter swans in the Canadian subpopulation of the Rocky Mountain population reduced the biomass of tubers and rhizomes by 24% during the spring, but shoot density and biomass of S. pectinata did not differ the following summer (LaMontagne et al. 2003).

Myriophyllum spp.

Percent cover estimates for Myriophyllum spp. were relatively similar between 1958, 1988, 1993–2001, and 2009 ranging from 6 to 17% (Fig. 23). Based on wet weigh biomass, Myriophyllum sibiricum (previously reported as M. exalbescens) accounted for 25% of the total biomass during 1979 and increased to 42% during 1986 (Shea et al. 1996). However, during this time biomass of M. sibiricum decreased from 10.1 to 7.5 kg/m². Its decline was less than other species, therefore accounting for a higher percentage of the total biomass in 1986 compared to 1979 (Shea et al. 1996). By 1987, total biomass (9.6 kg/m²) was similar to 1979 (Shea et al. 1996).

Limited information is available on the factors affecting native species of Myriophyllum. M. sibiricum was most abundant at slow current speeds (0.0–0.2 m/s; 0.0–0.7 ft/s), water depths < 1.5 m (4.9 ft; but 1/3 of the biomass occurred in water 1.5–3.0 m [4.9–9.8 ft]), and in silt sediments on the Nechako River, British Columbia (French and Chambers 1996). M. sibiricum occupied similar niches as S. pectinata, C. hermaphroditica, E. canadensis, although it was able to occupy deeper habitats when compared to S. pectinata (French and Chambers 1996).

Myriophyllum spp. is not considered an important food source for trumpeter swans. Trumpeter swans in the Greater Yellowstone area avoided consuming M. sibiricum (Squires 1995). Therefore the population of wintering trumpeter swans on the Henrys Fork probably does not directly affect the abundance of Myriophyllum spp. S. pectinata competes well with Myriophyllum sibiricum (Moen and Cohen 1989), but preferential foraging by trumpeter swans on S. pectinata or other preferred foods (e.g. Elodea canadensis) may give Myriophyllum spp. an indirect competitive advantage.
It appears that the distribution, frequency, and abundance of *Elodea canadensis* has fluctuated since 1958 when it was only observed at three locations from the Railroad Ranch Bridge downstream to Harriman Spring (likely the same location as Cold Spring). *E. canadensis* appears to have increased from 1958 to the late 1970s and then decreased during the mid 1980s (Fig. 23). During 1993–2009, percent cover of *E. canadensis* declined to a low of 7% during 1997 and appears to have slightly increased since then.

Its distribution from Last Chance to Harriman East has also shifted throughout the period of historical surveys. By 1979 *E. canadensis* occurred in 69% of samples from Last Chance to Harriman East (similar in distribution to this study) and accounted for 25% of total biomass with almost $\frac{1}{5}$ of the biomass collected from Last Chance (upstream of the Railroad Ranch Bridge). But by 1985 no *E. canadensis* was noted upstream from Osborne Bridge. This was confirmed with sampling during 1986 and 1987 when *E. canadensis* was observed from the Railroad Bridge downstream and accounted for $\leq 1\%$ of total biomass. During 2009, percent cover of *E. canadensis* appeared to be higher at Harriman East compared to transects upstream of Silver Lake Outlet and Osborne Bridge.

Percent cover estimates for *Zannichellia palustris* and *Ranunculus aquatilis* were relatively similar between 1958, 1988, 1993–2001, and 2009; *Z. palustris* ranged from 6 to 16% and *R. aquatilis* ranged from 7 to 16% (Fig. 24). However, neither species was detected in any biomass samples during 1977 (trace amounts of *R. aquatilis* were noted), 1979, or 1980, but both were present again in surveys during 1986–2009.

Given their distinct changes in distribution throughout the Henrys Fork and lack of occurrence during some sampling periods, growth and/or germination requirements for *E. canadensis*, *Z. palustris*, and *R. aquatilis* may be more specific than other species of aquatic macrophytes. During a survey of six wetlands on the Caribou-Targhee National Forest during 2002–2003, *E. canadensis* and *R. aquatilis* were each observed at one site while other aquatic macrophytes were found at multiple sites (Henry 2004).

*E. canadensis* can rapidly propagate through vegetative means, exhibits a life cycle that favors cool weather, is opportunistic when obtaining nutrients, and has several mechanisms to enhance photosynthetic efficiency (Nichols and Shaw 1986). Due to these characteristics, *E. canadensis* is often considered a nuisance species outside of its native North America, including region of Asia, Africa, Australia, and New Zealand (Bowmer et al. 1995). Conditions on the Henrys Fork may be limiting one or more of its growth requirements. Productivity of *E. canadensis* may be controlled by photorespiration (Simpson and Eaton 1986), dissolved inorganic carbon (Madsen and Sand-Jensen 1987), nitrogen (Ozimek et al. 1993), and the relationship among oxygen, pH and carbon dioxide (Simpson et al. 2006).

Shifting occurrence of *E. canadensis* may also be due to foraging by trumpeter swans (Paullin 1973) or due to a natural decline resulting from mineral depletion (Sculthorpe 1967). The relationship between number of wintering swans and percent cover of *E. canadensis* is not consistent, however, high numbers of wintering swans during 1994–1996 and 2000 (summarized by Van Kirk and Martin 2000) coincided with lower estimates of percent cover during the following growing season.

*E. canadensis* grew in relatively deep clear water with substrates low in silt at Red Rock Lakes National Wildlife Refuge (Paullin 1973). *E. canadensis* was also found in deeper water
(up to 3 m; 10 ft) in British Columbia and had a higher biomass in silt compared to sand substrates (French and Chambers 1996). Biomass of *E. canadensis* at Badfish Creek in Wisconsin was positively correlated with water depth and organic matter, negatively correlated with current velocity, and was higher in silt compared to sand and gravel substrates (Madsen and Adams 1989). Similar results were observed during this study. Percent cover of *E. canadensis* was higher along transects at Harriman East that had deeper water and higher silt content. The association of *E. canadensis* with silt substrates may be a result of its effect on sedimentation patterns (Sand-Jensen 2008) rather than a factor determining its distribution. Regardless of the causal relationship, *E. canadensis* likely benefits from this association.

*R. aquatilis* had a unique distribution compared with other species of aquatic macrophytes, occurring in moderately fast flowing currents (0.4–0.6 m/s; 1.3–2.0 ft/s) where other species were limited (French and Chambers 1996). In addition *R. aquatilis* was more abundant in water 1.5–3.0 m (4.9–9.8 ft) deep than in shallower water and on sandy substrates compared to silt substrates. Flow velocities within stands of *Ranunculus* at two river reaches in the United Kingdom were slow (< 0.1 m/s; 0.3 ft/s) and accelerated to 0.8 m/s (2.6 ft/s) in unvegetated areas (Cotton et al. 2006). Increase in area of *Ranunculus* spp. during the spring was positively correlated with mean discharge (Ham et al. 2006).

Little information is known about the abiotic factors affecting the growth of *Zannichellia palustris*; however it appears to be highly sensitive to sedimentation. *Z. palustris* decreased in the vicinity of construction zones in Alaska (Crow 1979) and disappeared from the upper portion of the Fall River, California where 0.6 to 1.2 m (1.3–3.9 ft) of sandy sediment accumulated (Spencer and Ksander 2002). Germination of *Z. palustris* in experimental studies was inhibited by as little as 2 cm (0.8 in) of sedimentation (Spencer and Ksander 2002).

This same relationship between *Z. palustris* and sedimentation is not apparent on the Henrys Fork. *Z. palustris* was not observed on any transects during 1977, two years prior to the 1979 sediment release from Island Park Dam (Fig. 24) suggesting that some other factor may have contributed to its decline since 1958. In addition, *Z. palustris* was present at 8 of 10 transects during 1993, one year following the 1992 sediment release from Island Park Dam.

**Callitriche spp. and Potamogeton richardsonii**

*Potamogeton richardsonii* and *Callitriche* spp. were commonly observed throughout historical surveys of aquatic macrophytes and contributed to species diversity and structural diversity; however, percent cover estimates since 1958 were \(< 10\%\). Exceptions to this include an estimate of 18% of the total biomass for *P. richardsonii* during 1979 (Shea et al. 1996) and 13% cover of *Callitriche* spp. during 2001 (visual estimate; Shea 2001).

*Callitriche* spp. are intolerant of shade from other aquatic macrophytes and riparian cover (NRCS 2004) and were observed in relatively low abundance on the Henrys Fork (this study), on wetlands in the Caribou-Targhee National Forest (Henry 2004) and at Red Rock Lakes National Wildlife Refuge (Paullin 1973). *Callitriche* spp. may not be as abundant as other species of aquatic macrophytes in wetlands and low gradient rivers in eastern Idaho and southwestern Montana because of shading from other aquatic macrophytes. At the Nechako River, British Columbia, *Callitriche hermaphroditica* was more abundant in low velocity water (0.0–0.2 m/s; 0.0–0.6 ft/s) and water < 1.5 m (4.9 ft), whereas abundance was similar on silt and sand
substrates (French and Chambers 1996). *P. richardsonii* followed a similar distribution with respect to low velocity water and water depth; however it was more abundant on silt substrates compared to sand substrates.

### Vegetation Height

Vegetation height is often used in conjunction with percent cover estimates to provide additional insights into productivity of aquatic macrophytes. Vegetation height and the amount of the water column is occupied by vegetation are important parameters of habitat available for fish and macroinvertebrates. However, vegetation height is strongly influenced by phenology, water flows, current velocity, and the interaction of these factors. Annual variation in these factors results in a complex relationship with vegetation height (see Fig. 11 and Fig. 12).

Aquatic macrophytes occupied the greatest percent of the water column (66–71%) during 1989 and 1992 when October flows from Island Park Dam were relatively low (2–195 ft³/s, 0.01–5.5 m³/s). Vegetation occupied < 30% of the water column during 1995–1997 when October discharge from Island Park Dam was relatively high (459–643 ft³/s, 13.0–18.2 m³/s). However, this relationship was not consistent among years. Mean vegetation height and mean water depth were similar during 1997 and 2009; however discharge from Island Park Dam was approximately 100 ft³/s (2.8 m³/s) higher during 1997 than 2009. Given the variability between flows and vegetation height and the interaction of vegetation and water depth (Vinson et al. 1992), this suggests that phenology of vegetation may account for some of the observed variability.

### Periphyton

Periphyton and phytoplankton were not identified to any taxonomic level during this study; however, they are important components of river processes. Percent cover of periphyton tended to be higher at East Harriman than on transects upstream of Silver Lake Outlet. Long strands of filamentous green algae were commonly observed associated with *Elodea canadensis* (see Fig. 4). Epiphytic algae (globular, brown in color) were attached to vegetation along the shallowly-flooded river margins.

During 2009, algae appeared to increase between August and October. In particular, the globular brown algae commonly attached to aquatic macrophytes seemed much more prevalent during October compared to August.

Similar to macrophytes, periphyton assemblages slow water velocities (Dobbs and Biggs 2002) and likely increase water depths for a given flow. Water attenuation by periphyton assemblages varies by growth form and architecture (Dobbs and Biggs 2002). Identification and quantification of algae on the Henrys Fork will contribute to an increased understanding of the interaction of algae and river processes.
Physical Stream Characteristics and Aquatic Macrophytes


Changes in productivity of aquatic macrophytes on the Henrys Fork have been attributed to low winter flows with associated build up of thick ice, high variation between winter and spring flows, influx of silt from draw downs at Island Park Reservoir, and winter foraging by trumpeter swans and other waterfowl (Shea et al. 1996).

Sediment

As described in the previous section, different species of aquatic macrophytes occupy different habitat niches according to substrate and/or a combination of substrate and other abiotic factors. Fine sediments contain important nutrients needed for growth and reproduction of aquatic macrophytes. Once established, aquatic macrophytes increase retention of fine sediments and coarse and fine particulate organic matter by trapping particles and reducing velocities to allow suspended particles to settle (Sand-Jensen 2008, Horvath 2004, Koetsier and McArthur 2000). This positive feedback loop allows macrophytes to create an abiotic environment favorable to their continued growth and expansion.

Patterns of sediment deposition are related to plant morphology and canopy structure. Fine sediment deposition markedly raised the surface bed within dense patches of Callitriche cophocarpa and Elodea canadensis (Sand-Jensen 2008). The increased occurrence of silt along transects with higher cover of E. canadensis at Harriman East is likely a result of sediment deposition associated with E. canadensis. Dense patches of Ranunculus peltatus showed variable patterns of sediment deposition with fine sediments in the upstream portion of the patches and coarse sediments in the downstream portion of the patches (Sand-Jensen 2008). Distinctive patterns of flow and fine sediment deposition were also created by different growth forms of Ranunculus between river reaches (Cotton et al. 2006).

Pebble counts at 25–30 plots along each transect likely did not account for all the variability within a transect (A. Henry and J. DeRito, personal observations). However, most of the fine sediments (sils and sands) deposited on the surface during the 1992 sediment release from Island Park Reservoir appear to have been transported out of the vicinity of Harriman State Park. Areas of fine sediments up to 1 m (3.3 ft) deep of were observed at Big Bend and Millionaire’s Pool during 1993, (Shea et al. 1996); the deep area observed during this study was approximately 0.3 m (1 ft) deep at Millionaire’s Pool and transect 2B at Big Bend. This trend was also apparent from the pebble counts. The frequency of sand on transects 2B and 3 was relatively high compared to transects further upstream. The composition of substrate below the surface was not sampled, so fine sediments may still be embedded within the river bed.
**Water Velocity**

Surface water velocity on the Henrys Fork generally decreased from upstream to downstream transects, consistent with decreasing river gradient over the same area (Gregory 2008). An exception to this was transect 2B, which had lowest surface water velocity compared to other transects. In addition to being the widest transect and therefore having greater cross-sectional area to pass the same amount of flow, wind speed had increased during sampling and may have decreased surface water velocities compared to other transects (J. DeRito, personal observation).

Water velocity also generally decreased from the surface to the bottom of the water column because measurements lower in the water column were typically within stands of aquatic macrophytes and/or algae. Lower current velocities within macrophyte stands have been reported for multiple rivers and experimental flume studies. Water velocities within three species of aquatic macrophytes (*Myriophyllum triphyllum*, *Potamogeton crispus*, and *Glyceria fluitans*) were lower than water velocities at the water surface (Dodds and Biggs 2002). Flow velocities in stands of *Ranunculus* were < 0.1 m/s (0.3 ft/s), compared to velocities up to 0.8 m/s (2.6 ft/s) outside of the macrophyte stands in the River Frome catchment, United Kingdom (Cotton et al. 2006). Flow velocities in patches of *Callitriche* were 11-fold lower than velocities measured upstream of the patches in Danish Streams (Sand-Jensen and Mebus 1996). Macrophyte beds in Whakapipi Stream, New Zealand were estimated to decrease water velocity by 41% during the growing season (Champion and Tanner 2000).

The inverse correlation of current velocity and percent cover of aquatic macrophytes at the Henrys Fork is similar to results from the lower River Spree in Germany (Schulz et al. 2003), the Bow River below Calgary (Chambers et al. 1991) and hypothesized for the Breitenbach in Germany based on variation in velocity among sites (Horvath 2004). The lack of strong correlations in this study compared to other studies may be due to differences in biomass or stem density for plots with the same percent cover, differences in plant morphology, and/or differences in the amount and type of algae among sampled plots. The location of velocity measurements during this study may have also contributed to a weak correlation with percent cover. Velocities were measured at the upstream edge of the sample plot and were likely influenced by vegetative characteristics outside of the plot.

Water velocity attenuation by different species of aquatic macrophytes is highly variable and has been related to plant morphology (Dodds and Biggs 2002). Current velocities within patches of aquatic macrophytes with large leaf area on bushy shoots (e.g. *Elodea canadensis*) reduced velocities more than species with stream-lined leaves (Sand-Jensen and Mebus 1996). Because flow patterns were similar between individual macrophyte stands of the same species, macrophyte stands may be suitable functional units for analyzing the influence of macrophytes on flow and associated physical and biological processes (Sand-Jensen and Pedersen 1999).

Water velocity attenuation by assemblages of periphyton is also variable, and is dependent on growth form and architecture. Periphyton assemblages consistently reduced water velocities more than different species of macrophytes (Dodds and Biggs 2002) and therefore may contribute as much if not more to dynamic river processes.

Water velocity attenuation has important implications for nutrient transport and uptake, fine sediment deposition, and habitat heterogeneity (Dodds and Biggs 2002). Deflected flows around macrophyte stands contribute to forming a mosaic of highly variable substrate which has
important implications for spatial variability of sediment and biological communities that these areas support (Sand-Jensen and Mebus 1996). Slower water velocities within aquatic macrophytes increases deposition of sediment and organic matter, and increases nutrient retention by as much as 12% of the total nutrient load (Schulz et al. 2003).

**Nutrients**

Nutrients in aquatic systems (e.g., phosphorus and nitrogen) are influenced by a wide variety factors, including geology of parent materials, land uses adjacent to and upstream of the area of interest, septic waste water, and industrial runoff. Runoff with high nutrient concentrations will likely contribute to nutrient-rich sediments, providing favorable conditions for proliferation of aquatic macrophytes (Barko et al. 1991); aquatic macrophytes in turn accumulate more fine sediments rich in nutrients.

Aquatic macrophytes can obtain nutrients from the sediment and/or from the water column. The relative importance of sediments and water as sources of nutrients is complex and often inconclusive (Clarke and Wharton 2001). However, it is generally accepted that aquatic macrophytes fulfill their requirements for phosphorus, nitrogen, and micronutrients (e.g., calcium, potassium, etc) by direct uptake from the sediment because these nutrients are less available in aqueous forms (Chambers et al. 1989, Barko et al. 1986). Nutrient-specific differences in the sediment and water column and absorptive capacities of the roots and shoots may influence the site (roots vs. shoots) of uptake by aquatic macrophytes (Denny 1980).

Land use changes along the Henrys Fork that have likely affected water quality of the river within the study area include installation of a centralized sewer system (and discontinued use of individual septic tanks) and fencing of cattle off from the Henrys Fork. The centralized sewer system and treatment plant was installed at Mack’s Inn during 1982 and at Last Chance during 1986 (Gregory 2008). Cattle were fenced off of the river at Harriman East during 1986; Railroad Ranch during 1988; and Last Chance during 1989 (J. DeRito, personal communication).

Limited information on nutrients is available for the Henrys Fork. No data on nutrients in the sediment are available; however, sediments are likely rich in phosphorus due to the geological formations within the watershed. Phosphorus concentrations in the water may also be relatively high due to geomorphic factors. Water quality data were collected along the Henrys Fork during 1974 (Forsgren 1975), 1994 (Goodman 1994), 1995 (Goodman 1995), and 2009 (McMurray 2009). Concentrations of total inorganic nitrogen were similar at Last Chance, Osborne Bridge, and Pinehaven/Riverside Campground during all sampling periods (Fig. 25). The concentration of total phosphorus at Last Chance appears to have decreased from 1974 to the mid 1990s; total phosphorus was similar between 1994 and 2009 (Fig. 25).

Nutrient concentrations at the Henrys Fork prior to the late 1980s may have been artificially high due to anthropogenic inputs. Prior to installation of a centralized sewer system, high nutrients from septic tanks likely leached into the Henrys Fork. “Plumes of vegetation” were observed near summer homes along the Henrys Fork during the early–mid 1980s (R. Shea, personal communication).
Aquatic Macrophytes and Macroinvertebrates

The distribution and abundance of macroinvertebrate species in aquatic habitats is associated with many factors, including water chemistry (e.g., pH, phosphorus, and dissolved oxygen), river reach characteristics (e.g., erosion vs. deposition zone, riffle vs. pool), substrate, and aquatic macrophytes. Macroinvertebrates were not sampled during this or previous studies of aquatic macrophytes; however, their abundance and species richness at the Henrys Fork is likely associated with changes in aquatic macrophytes and associated variability in substrate and water velocity.

Taxa richness, abundance, and biomass of macroinvertebrates were significantly higher on *Ranunculus*, *Berula*, and *Callitriche* than on gravel and silt substrates in an English chalk stream (Wright 1992). Higher abundances of macroinvertebrates are also found on dissected-leaf plants compared to broadleaf plants (Cheruvelil et al. 2000). For example, the density of invertebrates on *Myriophyllum sibiricum* was higher than the density on *Potamogeton richardsonii* (Gerrish and Bristow 1979). Densities of macroinvertebrates in shallow aquatic habitats in New York were more than 70% higher in patches *Stuckenia pectinata*, *Lemna trisulca*, and *Ceratophyllum demersum* compared to *Elodea canadensis* (Krull 1970).

Aquatic macroinvertebrates that are important food resources for rainbow trout (Tricoptera, Ephemeroptera, and Diptera; Angradi and Griffith 1990) have variable associations with aquatic macrophytes. Some species of mayflies (order Ephemeroptera) and true flies (order Diptera) are associated with aquatic macrophytes. Mayfly taxa in the family Baetidae were associated with aquatic macrophytes at the Serra de Cipó in southeastern Brazil (Goulart and Callisto 2005). The densities of two species in the family Chironomidae (order Diptera) were positively correlated to biomass of *Egeria densa* and *Potamogeton crispus* at a lowland stream in New Zealand (Collier et al. 1999). The densities of two genera from the Ephydridae and Culicidae families (order Diptera) were significantly correlated to the biomass of *S. pectinata* at the Coyote Hills Marsh in California (Bergey et al. 1992).

Caddisflies are often associated with unvegetated areas. Three genera of caddisflies and one genus of mayflies were associated with unvegetated areas at the Portneuf River, Idaho (Gregg and Rose 1985); however, only two species of aquatic macrophytes (*Ranunculus* and *Rorippa*) were used for comparison. Caddisfly larvae at the Henrys Fork were attached to leaves of fine-leaved pondweeds (e.g., *Stuckenia* spp.) but not other species of aquatic macrophytes during August 2009 (A. Henry, personal observation). This suggests that caddisfly larvae may show associations with different species of vegetation. The apparent decrease in *Stuckenia* spp. on the Henrys Fork may be contributing to less robust insect hatches.
Fig. 23. Estimates (mean ± SD) of *Stuckenia* spp., *Elodea canadensis*, and *Myriophyllum* spp., sediment releases from Island Park Reservoir, and annual peak discharge from Island Park Dam from 1958 to 2009 at the Henrys Fork of the Snake River, Idaho. Historical data compiled from Shea et al. (1996), Shea (1997), Shea (1999), and Shea (2001).
Fig. 24. Estimates (mean ± SD) of *Ranunculus aquatilis*, *Zannichellia palustris*, and *Callitriche* spp., sediment releases from Island Park Reservoir, and annual peak discharge from Island Park Dam from 1958 to 2009 at the Henrys Fork of the Snake River, Idaho. Historical data compiled from Shea et al. (1996), Shea (1997), Shea (1999), and Shea (2001).
Fig. 25. Mean ± SD of total phosphorus and total inorganic nitrogen during 1974, 1994, 1995, and 2009 at the Henrys Fork of the Snake River, Idaho. Data were compiled from Forsgren (1975), Goodman (1994), Goodman (1995), and McMurray (2009).
RECOMMENDATIONS

1. Monitoring of aquatic macrophytes on transects sampled during 1993–2009 should continue annually or every other year for 5 years to: 1) assess trends in vegetation over time; and 2) determine if the lower percent cover of aquatic macrophytes during 2009 compared to 1993 and 1999 is a long-term decreasing trend in cover of aquatic macrophytes or if it is part of the natural variability of aquatic macrophytes life cycles.

2. If inferences about trends in aquatic macrophytes need to be made about the target population of interest (Henrys Fork from Last Chance to Harriman East), random samples should be established for future assessments. This will meet the statistical analysis assumption that samples are randomly drawn from the population (Elzinga et al. 2001).

3. Aquatic macrophytes should be sampled throughout the growing season or during the peak growing season to assess maximum habitat available to adult rainbow trout and macroinvertebrates. The time of peak growth varies between years and species; however, sampling vegetation when flowering stalks and seeds are present likely represents a period of peak growth and will aid in the identification of species. Sampling during this time period will also provide information on peak available habitat during the angling season.

4. To quantitatively assess the physical and biological factors affecting aquatic macrophyte abundance, multivariate analysis or a priori modeling should be conducted. This will assess which factors and/or interaction of factors (e.g., substrate, nutrients, temperature, etc.) account for the most variation of aquatic macrophyte abundance and distribution.

5. Quantitative assessments should focus on the factors affecting Stuckenia spp. on the Henrys Fork given its apparent decline of since 1958 and its importance to foraging trumpeter swans and macroinvertebrates.
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