Annual Sediment Retention and Hydraulic Residence Time Variability in a Riverine Wetland Receiving Unregulated Inflow from Agricultural Runoff

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by

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Thesis of Randall W. Holloway:

ANNUAL SEDIMENT RETENTION AND HYDRAULIC RESIDENCE TIME VARIABILITY IN A RIVERINE WETLAND RECEIVING UNREGULATED INFLOW FROM AGRICULTURAL RUNOFF

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DEDICATION

I dedicate this thesis to my grandparents, whose posthumous support allowed me to return to school and pursue this graduate degree.

ABSTRACT

Annual Sediment Retention and Hydraulic Residence Time
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Randall W. Holloway
Master of Science in Coastal and Watershed Science and Policy
California State University Monterey Bay, 2010

The ability of a wetland to treat nonpoint source pollution is in part a function of its hydraulic residence time and physical trapping of sediment. Natural hydrologic variability of unregulated inflows to natural and restored riverine wetlands causes variability in hydraulic residence time and associated sediment retention functions. This study quantified the sediment retention and hydraulic residence time variation in a self-restoring riverine wetland receiving unregulated stormwater inflow from an agricultural watershed. Elkhorn Slough, an estuary affected by nonpoint source pollution in Monterey County, California, lies immediately downstream of the 22 ha wetland site. Two stream gages monitored hydrology at the inlet and outlet. Sediment transport was determined using Helley-Smith and DH-48 sediment samplers. A Turner C3 fluorometer facilitated the residence time study. The residence times through the wetland varied inversely with discharge and ranged from three to 28 hours. The residence time distributions were bimodal likely due to channelization in the wetland. A strong inverse relationship occurred between average discharge at the outlet and centroid of the residence time distribution. The wetland retained 2681 tonnes of suspended and 139 tonnes of bedload sediment during a year with slightly above average rainfall, representing 71% of the total sediment load supplied. The site retained all supplied bedload, and it accounted for 5% of the total retained sediment. The amount of total sediment retention is on par with many constructed wetlands having more controlled hydrology.

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CHAPTER 1

INTRODUCTION

The conversion of undisturbed landscapes to agriculture and urban development reflect increasing demand for food and housing from a growing human population and economy. These conversions have led to both a dramatic loss of wetland area and decreased water quality in streams due to runoff containing nonpoint source (NPS) pollutants (Baker 1992; Charbonneau and Kondolf 1993; USEPA 1992). Common NPS pollutants are elevated levels of nutrients, sediments, pesticides, and pathogens. Although agricultural runoff is the largest supplier of NPS pollution to surface waters in the United States (Baker 1992; USEPA 1992), elevated sediment loads and concentrations are derived from urbanizing watersheds as well (Gellis 2009, Woodward and Foster 1997). Unintended soil loss from agricultural erosion is now the largest component of the increasing rate of anthropogenic earth movement (Hooke 2000).

The quantity and quality of sediment in a river influence the hydrology, morphology, and ecology of both the river and its receiving waters (Owens et al. 2005). Sediment physically affects ecological habitat by creating or covering structure and influencing levels of light penetration and water temperature. Sediment also chemically affects habitat by transport of adsorbed nutrients, pesticides, herbicides, and metals (Owens et al. 2005; Steiger et al. 2005). Sediment from agricultural runoff can change wetland plant communities and create favorable conditions for exotic species (Werner and Zedler 2002). Rates of sediment deposition above 0.3 cm/yr in riparian forests are common in watersheds with agricultural and urban land conversion (Bendix and Hupp 2000), which impair fine root production and biomass integral to water filtration (Cavalcanti and Lockaby 2005). However, the specific hydrology, soils, and flora of a wetland determine how much sediment deposition can occur without a decrease in biodiversity (Koning 2004).

Wetlands improve water quality impacted by NPS pollutants such as nutrients, pesticides, sediments, and pathogens (Fisher and Acreman 2004; Mitsch and Gosselink 2000; Raisin and Mitchell 1994). The hydraulic residence time through a wetland is closely tied to the ability to trap and retain sediment, which is a key process in general water quality improvement (Blahnik and Day 2000; Carleton et al. 2001; Woltemade 2000). Sediment retention and aggradation are the key geomorphic processes of wetlands since they are typically regions defined by local low elevation and low hydraulic gradient (Johnston 1991; Mitsch and Gosselink 2000; Phillips 1989). The Nonpoint Source Pollution Plan of the California State Water Resources Control Board (SWRCB 2000) identifies specific management measures for addressing NPS, including the conservation and restoration of wetland areas and addressing the effects of hydromodification of wetlands and other water bodies.

While constructed wetlands have proven to be effective at treating water for many NPS pollutants and have been the subject of most studies (Hammer 1989; Kadlec and Knight 1996), the efficacy of restored or native wetlands is not as well documented. Despite this lack of assessment, restored or natural wetlands are now used as a management tool to improve water quality and maintain habitat in agricultural areas (Whigham 1999, Zedler 2003). Restored and natural wetlands often differ from constructed ones by receiving highly variable or unregulated inflows, which create more complex flow paths, mixing, and residence time variability (Holland et al. 2004; Jordan et al. 2003; Knox 2008; Stern et al. 2001; Werner and Kadlec 1996; Wetzel 2001). Residence time variability due to unregulated flow may make sediment retention more difficult to predict. This aspect of wetland function is not yet clearly understood and scarcely studied (Stern et al. 2001).

The efficacy of restored and natural wetlands with unregulated inflow to treat agricultural runoff and particularly excess sediment is uncertain because of the variability in residence times, inflow rates, and amount of channelization (Hammer and Kadlec 1986; Holland et al. 2004; Knox 2008; Wetzel 2001). Sediment retention in constructed wastewater wetlands with regulated inflows averages 68% (Kadlec and Knight 1996). Wetlands with unregulated inflows ranged from 48-91% retention in a natural setting (Blahnik and Day 2000), and no net removal in a restored setting (Jordan et al. 2003). A review of 35 studies in 49 unregulated flow wetlands concluded similar treatment of NPS constituents to constructed ones but with high variability (Carleton et al. 2001). Knox (2008) compared sediment retention in two wetlands receiving irrigation runoff, with the unchannellized wetland removing 77% of its sediment supply and a channelized wetland acting as a net sediment source. Arp and Cooper (2004) found no substantial effect on sediment loads from relatively pristine riverine wetlands receiving unregulated runoff from snowmelt over pastureland.

This study assessed the residence time and sediment retention characteristics of a physically complex, unregulated wetland receiving NPS storm runoff from a chiefly agricultural watershed. The goal of this work was to improve understanding of the controls on sediment retention in such settings, and to compare this site's retention to sites with more regulated hydrology. I hypothesized that this large wetland would remove sediments but to a lesser extent than ones with regulated inflows. I also questioned whether a time lag for obtaining paired inflow/outflow water samples could be developed or predicted from stage or discharge. Paired samples that could account for lag time would most accurately reveal the water quality improvement afforded. The methods included stream gauging, sediment transport measurements, and dye fluorometry during and between storm runoff events through one year. The results quantify the effectiveness of unregulated flow wetlands at reducing sediment load and suspended sediment concentrations. Furthermore, they quantify variability of residence time at varied stage or discharge, which can be used in modeling removal rates of other NPS constituents and strategies for water quality sampling.

CHAPTER 2

METHODS

Study Area

The study wetlands are located two kilometers upstream of the mouth of the Carneros Creek watershed and 13 kilometers upstream of Monterey Bay through Elkhorn Slough in Monterey County, California (Fig.1). The 70-km^2 watershed above the study site has mixed land use, with rural residential, cultivated cropland, grazed grassland, maritime chaparral, and oak and riparian woodlands. The cultivated cropland covers six square kilometers and is primarily in strawberries, with some raspberries, flowers, and vegetables (Largay 2007). The Carneros watershed has a Mediterranean climate with cool winters, mild summer temperatures, and a high evaporation rate. It receives 45.7 cm yr^{-1} average precipitation (Laurel Marcus & Associates 2003) typically delivered in four to ten frontal storms between October and March. The stream also carries extreme floods during decadal-scale El Nino events. Carneros Creek is an annual stream with no summer flow. The groundwater basin in this area is under acute overdraft (Raines, Mellon and Carella Inc. 2002), but the study wetlands are separated from the aquifer by a thick confining clay layer located in the shallow subsurface.

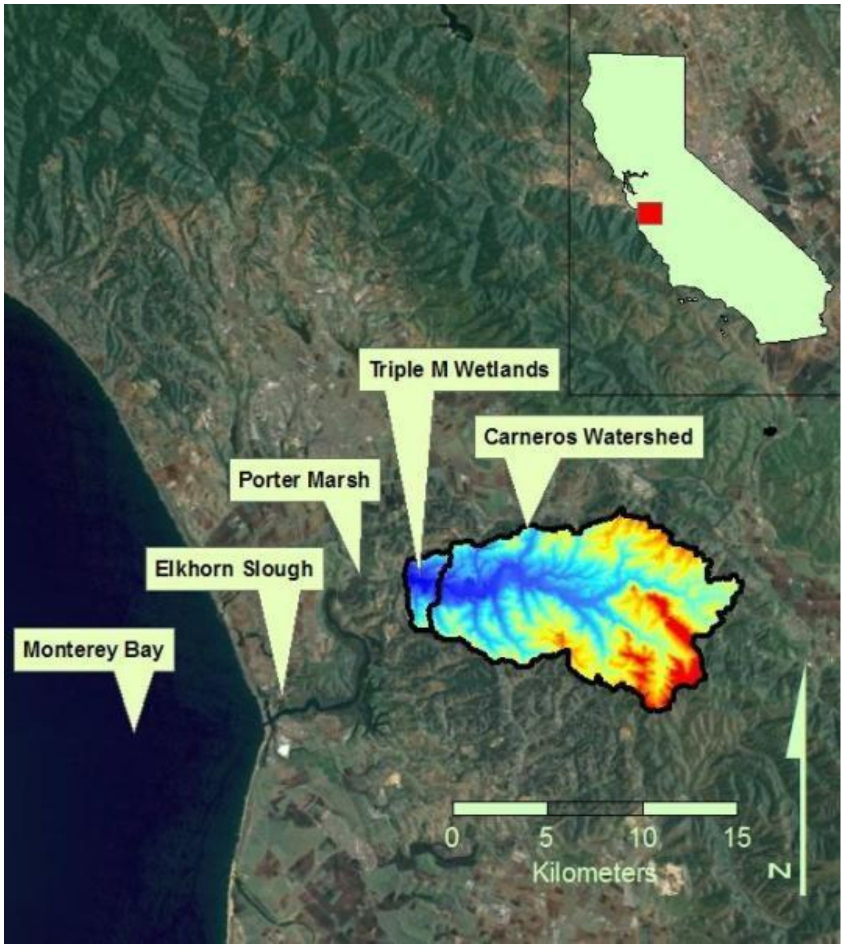

Figure 1. Study area of the Triple M Wetlands in a regional context and location in California. The outline of the Carneros Watershed is in two sections corresponding to the drainage above the Sill Road outlet and the Johnson Road inlet of the wetlands. Representations of watershed elevations from high to low are by bright red to dark blue color. The watershed discharges into Porter Marsh at the top of Elkhorn Slough, an estuary on Monterey Bay.

The study wetlands, "Triple M Wetlands", occur along a two-kilometer reach of Carneros Creek. They comprise the stream channel, riparian corridor floodplains, and marsh fields in the study site (Fig 2). The site includes riverine intermittent, palustrine forested, and palustrine emergent wetland types (NWI 2009). The Triple M Wetlands cover 22 ha, with 9 ha distributed in two marsh fields at the lower end of the site, 4 ha in the floodplains, and 9 ha in channel and riparian corridor. Flow enters under Johnson Bridge at an elevation of 5m (NGVD88) and exits the property across a ford at Sill Road at an elevation of 2 m (NGVD88) above sea level (Fig. 2). The soils are primarily Aquic Xerofluvents created by floodplain and channel processes, and Clear Lake Clay created by estuarine and still water wetland processes (Largay 2007, Los Huertos and Shennan 2002). These soils typify the transition from riverine/palustrine to estuarine environments. The riparian vegetation near the stream channel and floodplains is willow forest (*Salix* sp.), blackberry, and central coast scrub. The vegetation on the marsh fields is dominated by smartweed (*Polygonum amphibium),* with some cattails (*typha* sp.*)* also present. California Red-Legged Frog, California Tiger Salamander, and Santa Cruz Long-Toed Salamander are species of concern found at the site.

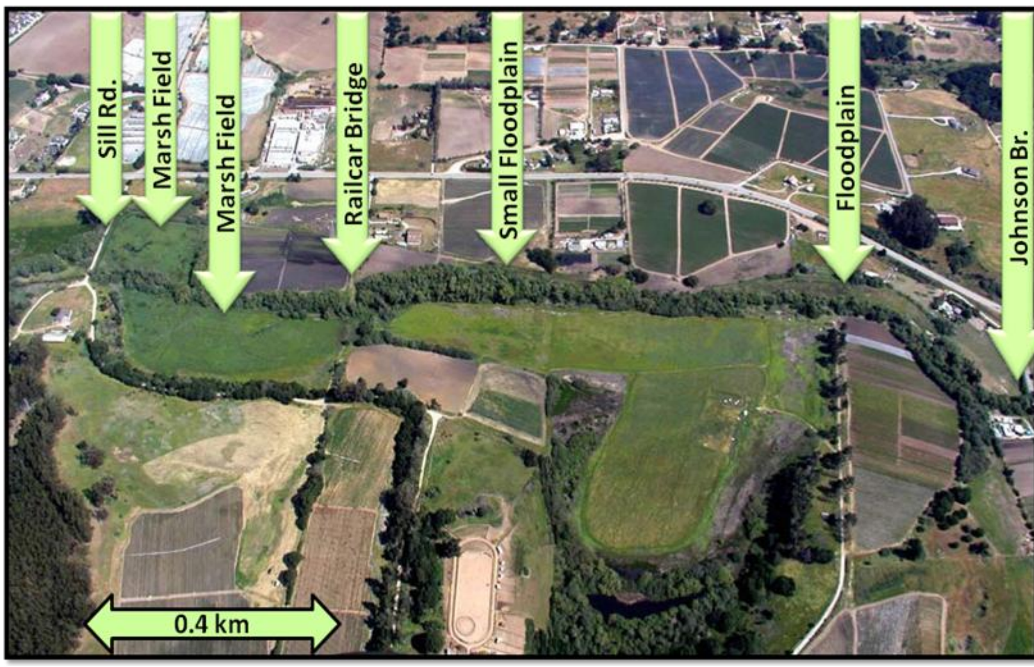

Figure 2. Study area of the Triple M Wetlands. Flow moves from right to left, entering under Johnson Bridge, passing two floodplains, an old railcar bridge, and two large marsh fields before exiting over Sill Road. Scale is not exact due to oblique angle of photo.

The Triple M Wetlands are present because of avulsion from a failed channelization project. Historically, landowners in the area deepened and straightened the stream channel, and the resulting dredge spoil piles isolated the adjacent floodplain. Reclamation ditches then ringed the floodplain to allow crop cultivation. During strong winter storms in the late 1990's a transverse sand fan 100 m downstream from the study site blocked the canal. The blocked flow caused channel aggradation and a complex pattern of avulsion that restored flow to four segments of the historic floodplain, These four segments comprise (from upstream to downstream) two small reaches of floodplain and two large marshes (Fig 2). In 2000, another avulsion immediately downstream of the Railcar Bridge further increased conditions for marsh habitat above Sill Road (Fig. 2) (Largay 2007).

Hydrology

Two continuous recording pressure transducers (Solinst Levelogger Gold) set to 15 minute intervals produced a time record of water stage at the inlet (Johnson Bridge) and outlet (Sill Road) of the study area. The transducer record was rated for discharge using instantaneous measurements at Johnson Road (n=17) and Sill Road (n=11) in water year 2010 (WY2010) across a large range of stage, where WY2010 is the period from October 2009 until October 2010. Discharge data were collected using a Pygmy (Gurley) current meter or an acoustic Doppler current profiler (SonTek) and velocity-area methods of Harrelson et al. (1994). Four discharge measurements were made using the surface float method when the stream was unsafe to wade at Johnson Road. Discharge-rating curves for the sites were developed by regressing instantaneous discharge measurements against associated stage data (Appendix D). The discharge-rating curves applied to the time record of stage data resulted in hydrographs for both sites. Flow duration curves were produced by ranking discharges in the hydrographs (Vogel and Fennessey 1995).

Sediment Budget

Sediment Budget Symbol definitions

Y = Yield of sediment (tonnes/yr)

S = Sediment storage at the Triple M Wetlands (tonnes/yr)

$\mathcal{E}$ = Uncertainty or error

L= Load, instantaneous (g s^{-1})

Q = Discharge of water (m^3 s^{-1})

t = 900 seconds (number of seconds in 15 minutes)

n = 35,040 (number of 15-minute periods in a year)

u = 10^{-6} (conversion from grams to tonnes)

(Subscript and superscript terms)

sus = Suspended sediment

bed = Bedload sediment

in = Johnson Bridge site

out = Sill Road site

side = Tributaries between Johnson Bridge and Sill Road

a,b = Regression coefficients from sediment rating curve

Sediment Budget Form

A sediment budget approach was used to determine storage of sediment at the Triple M Wetlands during the study period of water year 2010. The sediment budget was modeled as:

$$Y_{in} - Y_{out} = S + \epsilon$$

The full equation for the sediment budget with terms for both suspended load and bedload then became:

$$\left(Y_{sus,in} + Y_{side} + Y_{bed,in}\right) - \left(Y_{sus,out} + Y_{bed,out}\right) = S + \varepsilon$$

where error $\mathcal{E}$ was assumed to have a mean of zero. The yield terms in the sediment budget, with the exception of the side tributary adjustment Y_{side}, have the general form:

$$Y = u \times t \times \sum_{i=1}^{n} L_i$$

where

$$L_i = aQ_i^b$$

The yield is thus a summation of load over time and load is a function of discharge, which is in turn a function of stage.

The output of sediment (Y_{out}) is the annual yield at Sill Road while the input (Y_{in}) to the wetlands include the sediment entering above Johnson Bridge and from small side tributaries located between Johnson and Sill Roads. The tributary input was not directly measured, so the additional sediment input (Y_{side}) is assumed proportional to drainage area. As the tributaries were small and do not have sand bed channels it was also assumed that they carried only suspended loads. The drainage area difference between the Johnson and Sill sites equals 3.93 km^2, or 5.8% of the area upstream of Johnson Rd. Consequently, the equation correcting for the addition of suspended sediment from the side of the study area was:

$$Y_{side} = 0.058 \times Y_{sus,in}$$

The sediment budget equation was filled by 10,000 sets of bootstrapped sediment rating curves at each site applied to their respective discharge record to create a distribution for each variable from which the mean and confidence intervals could be calculated.

Sediment Rating Curves

Input and output of suspended and bedload sediments were measured at the Johnson and Sill sites in water year 2010. Sampling was event-based to capture the different sections of the hydrograph and periods between storms. Instantaneous bedload and suspended load measurements were regressed against associated discharge measurements to create sediment-rating curves for each site. The suspended sediment-rating curves resulted from a regression of suspended load measurements at Johnson (n=49) and at Sill (n=50) against corresponding discharge measurements. To avoid biasing the curve fits to fit the outcome of low discharges, which were far more common, I rarified the data by binning into 0.1 $m^3 s^{-1}$ sections from zero to one $m^3 s^{-1}$ and averaging the values in each bin. Likewise, each bedload-rating curve resulted from a regression of n=24 bedload measurements against corresponding discharge measurements at Johnson Bridge. Bedload data were rarified for values taken under 2.0 $m^3 s^{-1}$ by binning data into 0.5 $m^3 s^{-1}$ sections. No bedload transport events occurred at Sill Road during the course of this study, as measured bedload was zero for all surveys (n=24). As few as 12 samples per year can give good estimates of annual suspended sediment loads when collected on a hydrological and not a calendar basis, and the resulting sediment-rating curves are much less expensive to create than with automatic samplers or turbidometers (Horowitz 2003).

Suspended sediment concentration (SSC) samples were collected with a DH-48 depth-integrated sampler (Guy and Norman 1970; IAEA 2005). SSC samples were processed at CSUMB according to standard protocols (CCoWS 2004, Guy 1969). Bedload sediment samples were collected with a Helley-Smith sampler using standard methods (Guy and Norman 1970; IAEA 2005). Macroscopic pieces of organic matter were removed from dried samples before they were weighed.

The input (Y_{in}) and output (Y_{out}) was estimated by constructing sediment-rating curves through regression of instantaneous suspended sediment load and bedload in g s^{-1} against corresponding discharge. Instantaneous loads of suspended sediment were calculated as:

$$L = SSC \times Q$$

where *SSC* is suspended sediment concentration (g L^{-1}) and *Q* is discharge (L s^{-1}). Measured discharge from the time of the SSC samples was used when available and rated discharge was used when direct measurements were not made.

The equation for the sediment-rating curves was modeled as a power function (Asselman 2000; Cox et al. 2008; Walling 1978):

$$L = aQ^b$$

The sediment rating curve regressions used a generalized linear model with a log link function and gamma error, as this method eliminates back-transformation of logged values and subsequent bias corrections common to skewed data such as discharge, concentration, and loads (Cox et al. 2008).

Bootstrapped Sediment Rating Curves

In order to estimate the variation or uncertainty in the total storage term *(S)* from the sediment budget, a Monte Carlo bootstrap technique was used where the instantaneous load data were repeatedly re-sampled with substitution 10,000 times to create alternate rating curves for each variable (Efron 1982). The bootstrap technique effectively removes the influence of any one data point on the fit of the regression. The ‘cloud’ of 10,000 alternate lines from bootstrapping the data created a distribution of yield estimates for suspended and bed loads at Johnson Bridge and Sill Road after input to the sediment budget equation. The distribution of 10,000 storage estimates allowed calculation of uncertainty in the storage term through statistics of mean, standard deviation, and resulting confidence intervals. As the 10,000 sets of yield terms input to the sediment budget were paired randomly (resulting in high estimates at Johnson paired with low estimates at Sill and vice versa) the storage term assumes independence in the errors between the sites.

Residence Time

Hydraulic time of travel was measured through the downstream marsh fields by slug dye- tracer experiments (Holland et al. 2004; Kadlec and Knight 1996; Kilpatrick and Wilson 1989; Stern et al. 2001). Slug dye experiments were performed 6 times, once from Johnson Bridge to Railcar Bridge, once from Johnson to Sill, and five times from Railcar Bridge to Sill Rd. The dates of the experiments were 1/29/2010, 2/6/2010, 2/11/2010, 2/24/2010, 3/9/2010, and 3/23/2010 respectively. An in situ fluorometer (Model C3, Turner Corporation) measured fluorescence from passing dye concentrations every 30 seconds at Sill Road.

The fluorometer record resulted in a residence time distribution for each experiment. The residence time distribution (RTD) consists of the fluorescence weighted by discharge at the time of measurement graphed versus time from injection of the dye upstream. Weighting of the fluorescence record by discharge was through multiplying fluorescence in Relative Fluorescence Units by discharge in $m^3 s^{-1}$. Fluorescence could not be converted to concentration due to differences in the background levels present in laboratory calibration and actual background levels in the field. Starting dye dosage was initially calculated using the formula from Kilpatrick and Wilson (1989) and then adjusted to achieve desired signal strength. The times to leading edge, peak, centroid, and trailing edge were determined. Time to leading edge was calculated as the time to a concentration equal to 3% of the peak concentration (Holland et al. 2004), while the trailing edge was time to 10% of the peak concentration (Kilpatrick and Wilson 1989). The residence time was the time to centroid, calculated as the time to one-half of the area under the curve given by the residence time distribution (Holland et al. 2004).

Background fluorescence was measured for one or more hours immediately prior to dye introduction (Stern et al. 2001). To adjust for the background fluorescence, the average background value was then subtracted from all values in that RTD (Wilson et al. 1986). The dye was pre-mixed into 4 gallons of water taken at the site before input to the creek. The tracer dye used in this study was Rhodamine WT (20% solution, Keyacid) an organic compound that is safe for use in water supply systems and the most widely used tracer for RTD studies (Kilpatrick and Wilson 1989; Wilson et al. 1986).

CHAPTER 3

RESULTS

Hydrology

Hydrographs

The hydrographs resulting from the discharge rating curves (Appendix D) show the discharge for each site over time, with ten major storm events in the winter of 2010 (Figure 3). The total rainfall for WY2010 was 63.6 cm (KCAAROMA2 2010). The maximum discharge at Johnson Bridge was 9.2 $m^3 s^{-1}$ and 11.8 $m^3 s^{-1}$ at Sill Road. The total water yield for WY2010 was 5.8 X 10^6 m^3 at Johnson, and 5.0 X 10^6 m^3 or 14% less at Sill. Flow began on 10/13/2009 and ended at 6/8/2010.

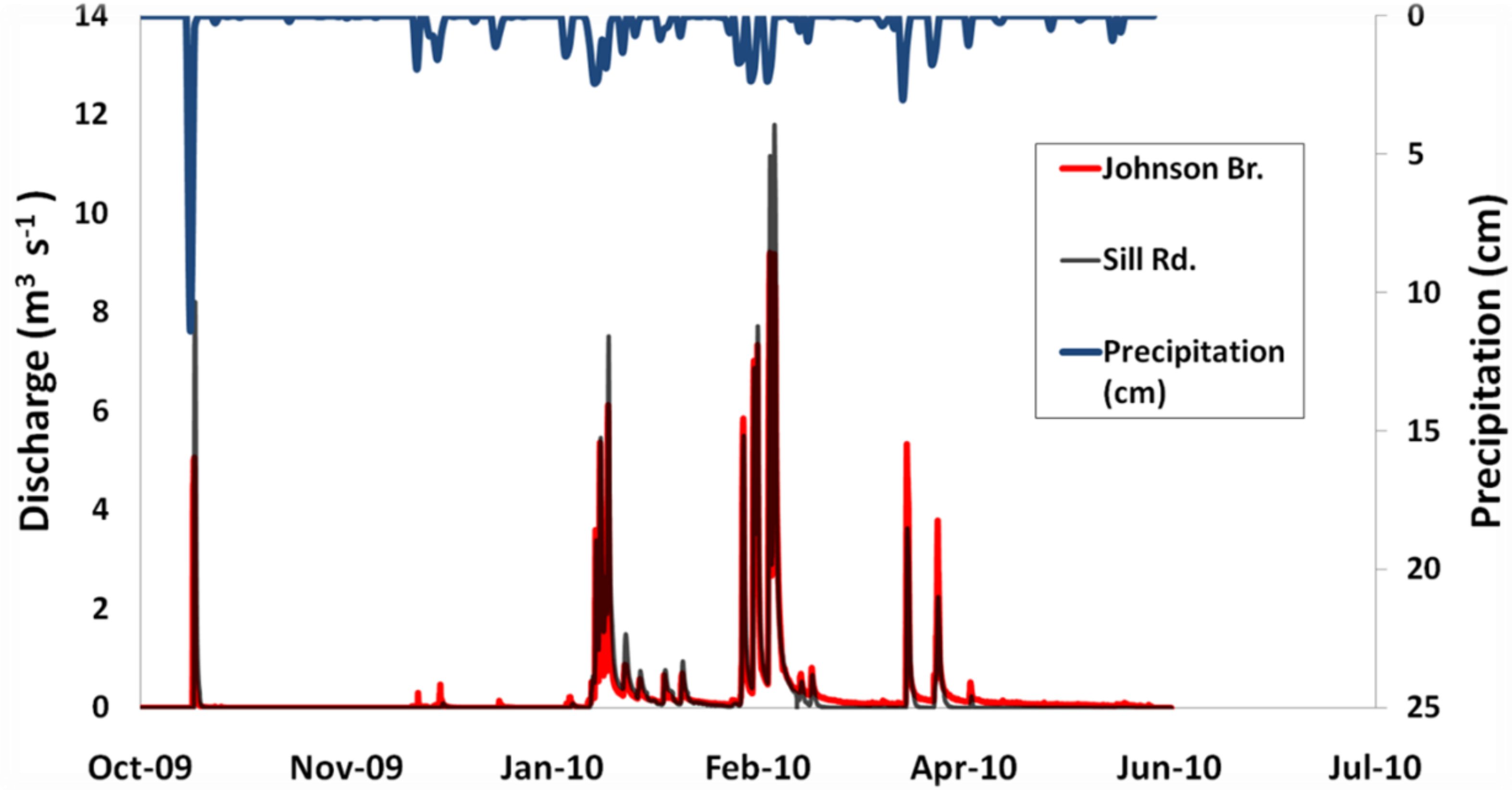

Figure 3. Hydrographs for the Johnson Bridge inlet and Sill Road outlet of the Triple M Wetlands from Oct-09 to Oct-10.

Flow Duration Curves

The flow duration curves at each site illustrate the intermittent nature of the system, with discharges below 1 $m^3 s^{-1}$ present over 90% of the time while occasionally reaching over 10 $m^3 s^{-1}$ (Fig. 4). The system is extremely 'spiky', often falling to very low discharges between storms.

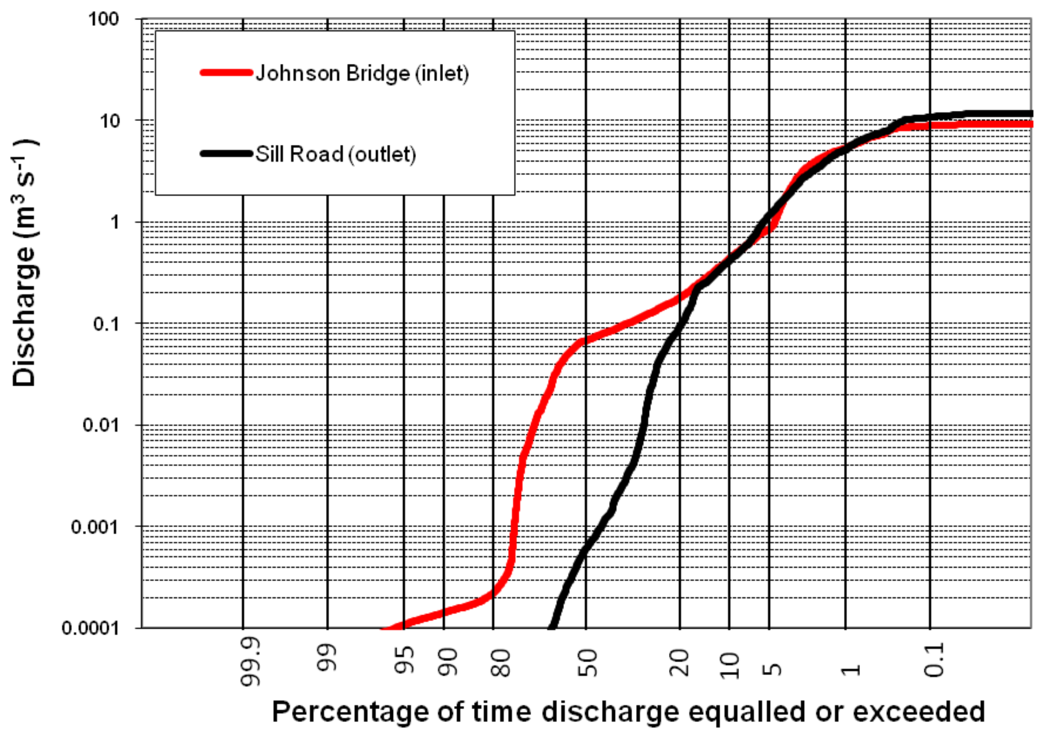

Figure 4. Lognormal plot of the flow duration curves at Johnson Bridge and Sill Road illustrating highly variable discharges and very little flow (<0.1 $m^3 s^{-1}$) approximately 75% of the time. Discharges less than 0.0001 $m^3 s^{-1}$ were omitted as they were far below any measured value.

Sediment Budget

Suspended load rating curves

The rating equation for original set of suspended load data at Johnson Road was:

$$L_{sus,in} = 457.8Q^{1.37}$$

The rating equation for original set of suspended load at Sill Road was:

$$L_{sus,out} = 148.9Q^{1.373}$$

A visual representation of these two suspended rating curves surrounded by 1,000 bootstrapped alternate curves show their relative loads across discharge, and how the bootstrapped curves have a central tendency around the original fit (Fig. 5).

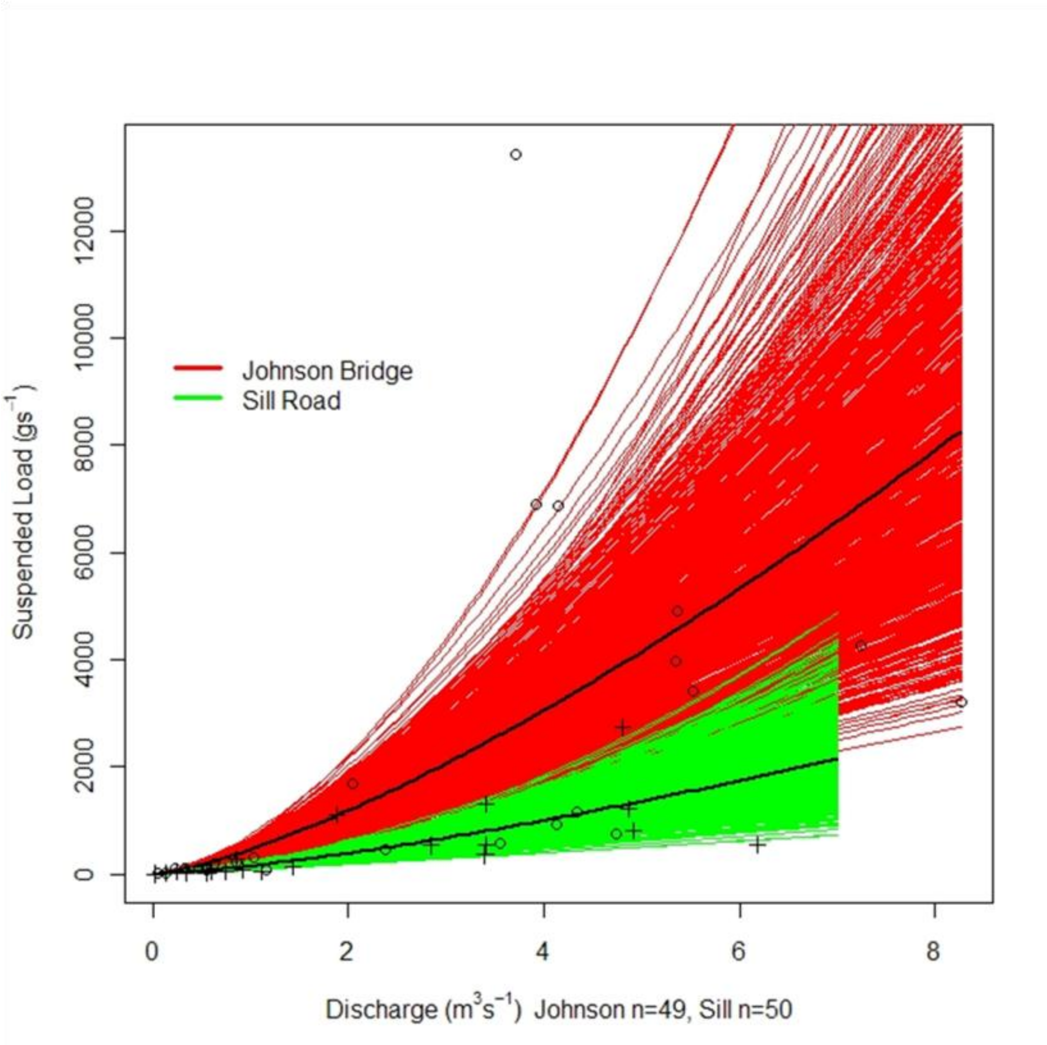

Figure 5. Visual representation of the suspended sediment rating curves for Johnson Bridge and Sill Road using original data set (black) surrounded by 1,000 alternate curves (red, green) using bootstrapped data sets.

Bedload rating curves

The rating equation for bedload at Johnson Road was:

$$L_{bed,in} = 19.62\,Q^{1.35}$$

No bedload transport occurred at Sill Road over all conditions present during the study period. The bedload size distribution at Johnson Bridge (Appendix E) returned a median grain size of 0.355 mm.

Sediment Budget Loads

The Triple M wetlands retained 70% of the suspended sediment and 100% of the bedload sediment supplied to them after accounting for side drainage area adjustment (Table 1). The total retention rate with suspended and bedload combined was 71%. Annual load was 2820 tonnes with a lower 95% confidence interval (1-sided, p=0.05) of 795 tonnes (Table 1).

Table 1. Sediment budget results in tonnes using the budget form:

$$(Y_{sus,in} + Y_{side} + Y_{bed,in}) - (Y_{sus,out} + Y_{bed,out}) = S + \varepsilon$$

Sediment Budget Variable	Mean	95% Upper Confidence Interval (1sided, p=0.05)	95% Lower Confidence Interval (1sided,p=0.05)
Johnson Br. Suspended *($Y_{sus,in}$)*	3618	5452	1784
Side Adjustment *(Y_{side})*	210	316	103
Johnson Br. Bedload *($Y_{bed,in}$)*	139	190	87
Sill Rd. Suspended *($Y_{sus,out}$)*	1146	1727	566
Sill Rd. Bedload *($Y_{bed,out}$)*	0	0	0
Storage *(S)*	**2820**	**4846**	**795**

A comparison of load duration curves for each site shows Sill Road having smaller loads across the whole range of high to low discharges, where high discharges are represented by a low percentage of exceedance (Fig. 6). The vertical difference between the curves indicates retention efficiency at a given exceedance, where the closer together the curves are equals less retention of sediment. Retention efficiency is the ratio of input at Johnson Bridge minus output at Sill Road divided by the input at Johnson Bridge. The efficiency was 54% at the lowest exceedance and highest discharge, and increased to near 100% at extremely small loads or discharge. These efficiency values are not exact due to different timing and amount of discharge at the sites, but an indication that the efficiency generally increases as discharge decreases.

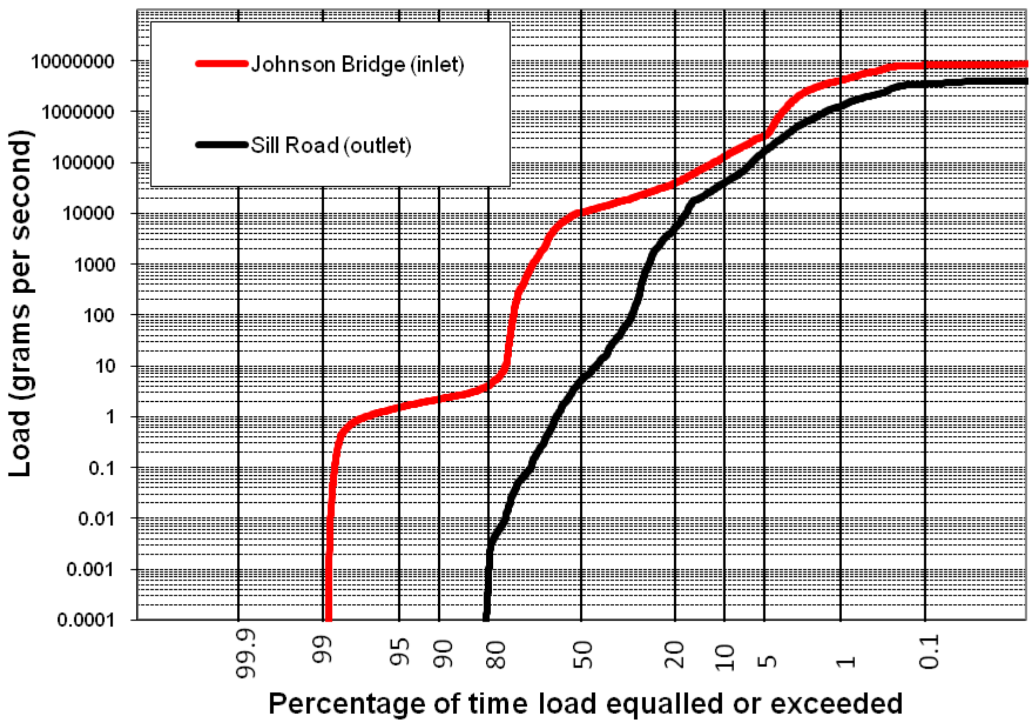

Figure 6. Lognormal plot of load duration curves at Johnson Bridge and Sill Road. A lower percentage of exceedance corresponds to larger discharges. Load values under 0.0001 g s^{-1} were omitted as they were far below any measured value.

Residence time

The residence time, or time to centroid of the residence time distribution varied inversely with stage and discharge (Table 2, Fig. 7), and the residence time distributions were bimodal for all dye experiments from Railcar Bridge to Sill Road (Fig. 7). Average and maximum stage and discharge varied little during three of those four experiments (Table 3). During the slug dye experiment at 0.1 m^3 s^{-1} average Sill Road discharge, interference and dropout occurred with the fluorometer readings, likely due to an observed oil slick (Fig. 7).

For calculations of centroid and other parameters on that experiment, the line was simply connected across the abnormal section. The experiment on 1/29/2010 from Johnson Bridge to Railcar Bridge, at an average Sill Road stage and discharge of 21.4 cm and 0.213 $m^3 s^{-1}$, showed little mixing and standard plug-flow. Time to centroid, start, peak, and end was 177, 132, 165, and 251 minutes, respectively (Fig. 8). The experiment on 2/6/2010 from Johnson Bridge to Sill Road used too little dye to register on the fluorometer.

Table 2. Residence time distribution parameters for the four slug dye experiments from Railcar Bridge to Sill Road in hours and minutes.

Sill Road Avg. Discharge ($m^3 s^{-1}$)	Start	Peak 1	Centroid (Residence Time)	Peak 2	End	Fluorometer Record Ends
0.001	5:48	7:49	27:50	30:16	41:32	119:27
0.1	13:00	17:03	19:42	25:37	28:46	89:35
0.26	1:41	2:11	12:23	9:08	30:11	91:31
1.24	1:19	1:58	2:51	57:01	58:27	73:39

Table 3. Maximum and average stage and maximum discharge during four slug dye experiments from Railcar Bridge to Sill Road.

Sill Road Avg. Discharge ($m^3 s^{-1}$)	Sill Road Max. Stage (cm)	Sill Road Avg. Stage (cm)	Sill Road Max. Discharge ($m^3 s^{-1}$)
0.001	6.4	5.4	0.001
0.1	19.8	19	0.130
0.26	26.8	22.9	0.406
1.24	51.4	33.4	5.179

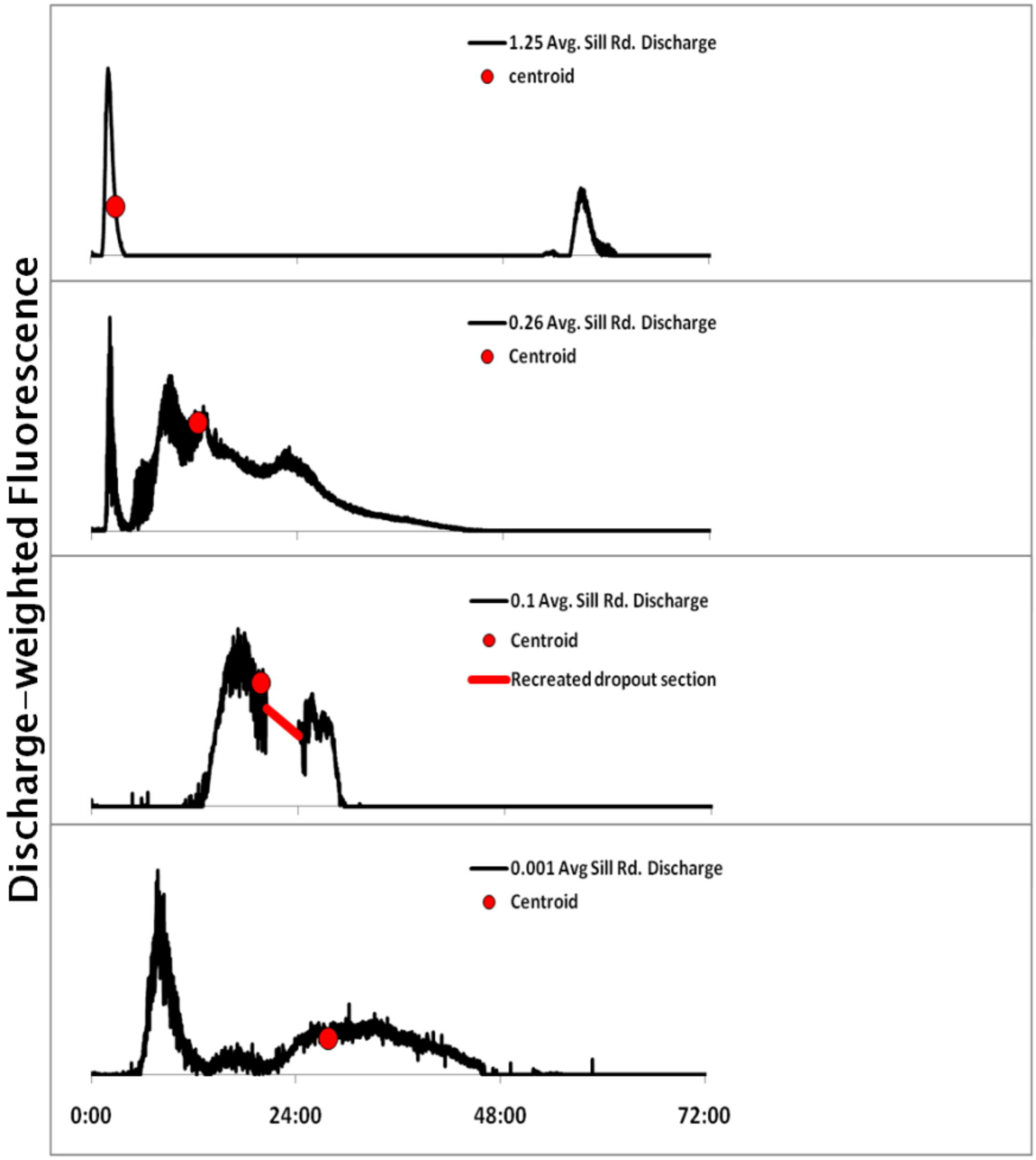

Figure 7. Residence time distributions from Railcar Bridge to Sill Road sorted by high to low average discharge (m^3s^{-1}) at Sill Road during experiment. Time in hours to centroid (red circles) varies inversely with discharge at Sill Road during experiments.

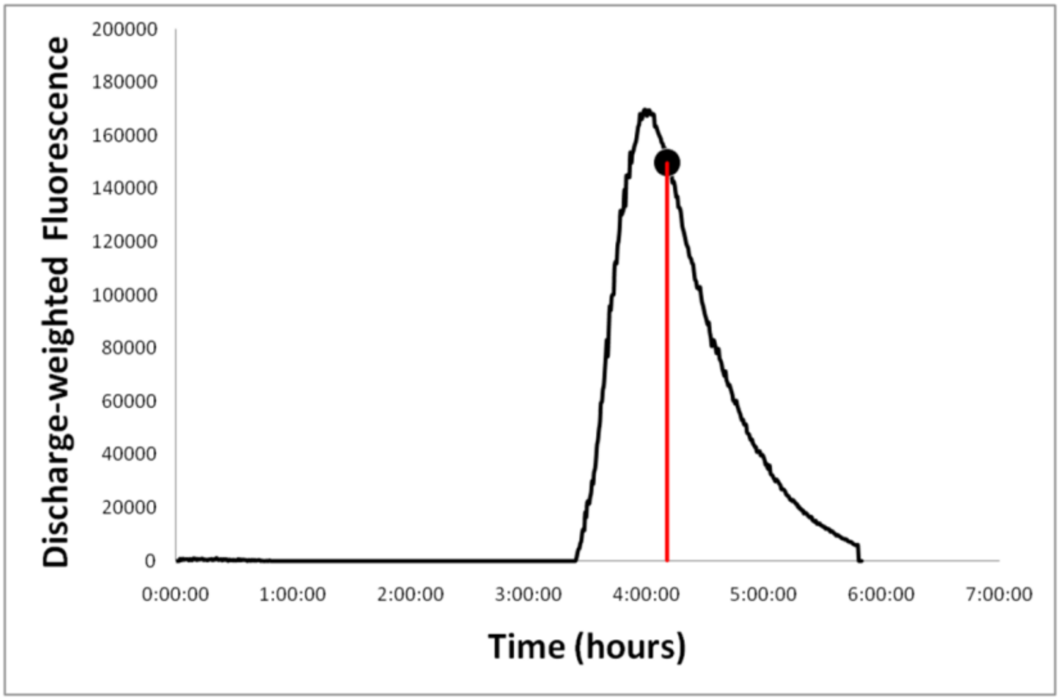

Figure 8. Residence time distribution from Johnson Bridge to Railcar Bridge during an average discharge of 0.18 $m^3 s^{-1}$ at Sill Road.

Prediction of residence time

The residence time as a function of discharge was inversely related and showed longer residence times occurred from Railcar Bridge to Sill Road at lower stages (Fig. 9). The exponential equation for this relation (n=4, p=0.015) was:

$$RT = 23.7e^{-1.745Q}$$

where *RT* is residence time (hours) and *Q* is average discharge at Sill Road ($m^3 s^{-1}$) during experiment.

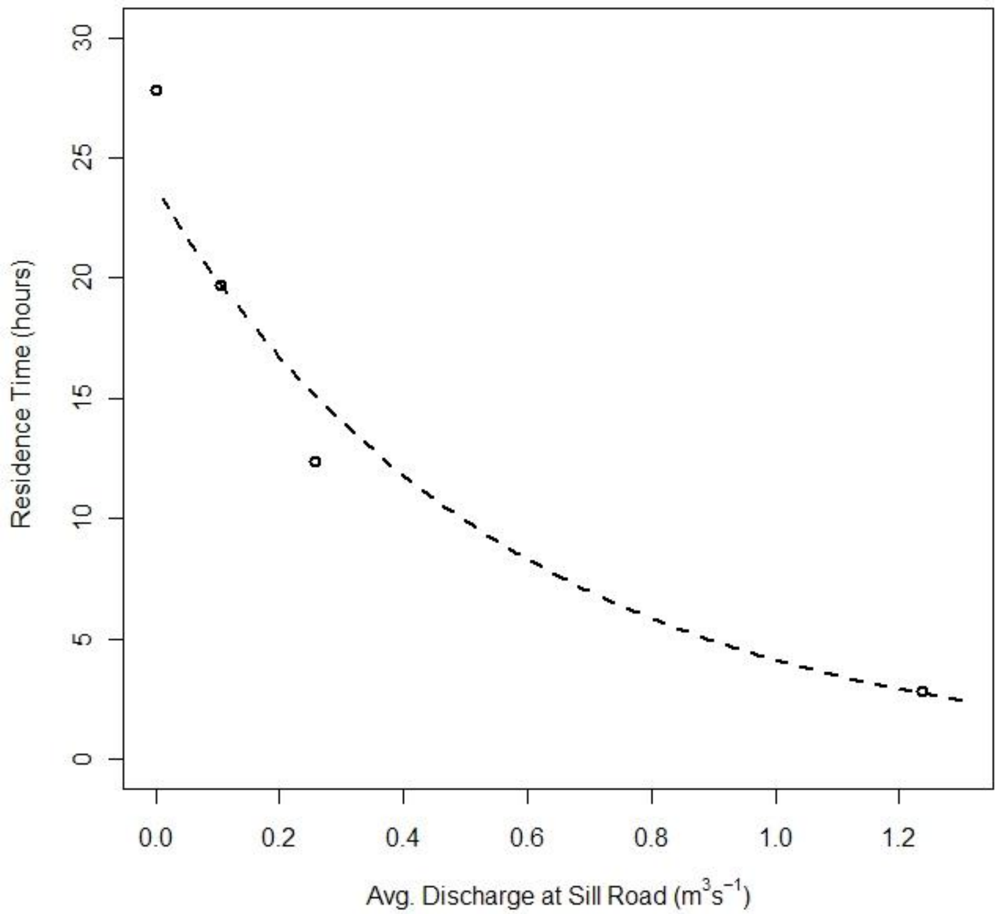

Figure 9. Inverse relation of residence time and average discharge at Sill Road during dye experiments from Railcar Bridge to Sill Road with an exponential model fit (n=4, p=0.015).

CHAPTER 4

CONCLUSIONS

The Triple M Wetlands retained 71% of sediment supplied to them during the study period of WY2010. Retention rates were 70% for suspended material and 100% for bedload. This amount of sediment retention is comparable to constructed wastewater wetlands averaging 68% (Kadlec and Knight 1996). The wetland retained 2681 tonnes of suspended and 139 tonnes of bedload in a year with slightly above average rainfall for a total of 2820 tonnes. All bedload delivered was retained, and accounted for 5% of the total. The distribution of 10,000 storage terms returned a lower 95% confidence interval (1-sided, $p=0.05$) of 795 tonnes. Retention efficiency increases as loads and discharges grow smaller when analyzed through differences between the load duration curves.

The 2820 tonnes of stored sediment spread over the 22 hectares of wetlands equals 7.5 mm/yr average deposition given a standard sandy soil volume/weight ratio of $1.7 g/cm^3$. This deposition rate exceeds the 1.33 mm/yr linear mean seal level rise trend observed in nearby Monterey Bay (NOAA 2010), and contrasts to areas of marshplain downstream in Elkhorn Slough, where low sediment accretion rates combine with high marshplain subsidence rates to put the marsh habitat at risk of drowning as sea level rises (Spear 2009). Similarly, the load of 795 tonnes of stored sediment from the lower 95% confidence interval equals 2.1 mm/yr, which also exceeds projected sea level rise. Tide gates 2 km downstream currently protect the lower end of the wetland site from inundation on highest tides. The deposition rate of 7.5 mm/yr is well over the 3 mm/yr that can impair fine root production and biomass integral to water filtration (Cavalcanti and Lockaby 2005); however, observation of dense wetland vegetation at the site over the last three years indicates no major impairments due to sedimentation rate.

Residual terms in sediment budgets such as storage derived from subtracting the output from the input have been criticized for containing the sum of errors from quantified sections (Kondolf and Matthews 1991). Qualitative and quantitative validation of the storage term is underway for this sediment budget by several methods. Concurrent and ongoing research at the site converts geomorphic change to the mass of accumulated sediments through detailed surveying and use of sediment traps. Initial results from the 1.3 ha floodplain located immediately downstream of Johnson Bridge (Fig. 2) show net deposition of 650 tonnes of mostly bedload sediment during the study period (Bassett 2010). While this floodplain contains only 6% of the wetland area, it retained approximately 23% of the total mass determined by this study, so sediment is not deposited evenly within the site. A series of scour chains placed in the channel for the study period also qualitatively validate deposition occurring on the bed.

The residence times through the marsh fields on the lower part of the wetland varied inversely with discharge and stage, and ranged from three to 28 hours. The residence time distributions were bimodal because the wetland is partially channelized, with some water moving quickly through the channel while some flows slowly over the marsh fields. In contrast, the dye experiment above the marsh fields from Johnson Bridge to Railcar Bridge was not bimodal and exhibited standard plug flow moving near average water velocity. While a strong exponential relationship was present between average stage at the outlet and residence times, using that relation or a more finely modeled one to take paired water quality samples may not be effective where the centroids fell between the two peaks of water exiting the site. Under that condition, a sample taken at the wetland exit would contain a mixture of water parcels and might not completely represent the water parcel sampled at the entrance to the wetland. This problem is not important when sediment or pollutant loads are constant through time, but could produce spurious estimates of water quality improvement when loads are strongly variable through time.

The difference in Johnson and Sill total water yields (788,775 m^3) was greater than expected given that Sill has 5.8% more drainage area than Johnson and may imply loss to groundwater interaction. However, factors such as evapotranspiration, farming use, and initial wetting of the soil and seasonal filling of the wetland volume account for much of the difference. Farming on the ALBA ranch takes 91,000 m^3 per year from the stream (Largay 2007). Evapotranspiration accounts for another 217,800 m^3 using an annual rate of 0.99 m/yr (Zone 2, CIMIS) spread over 22 ha.

CHAPTER 5

DISCUSSION

This study investigated sediment and hydraulic dynamics at the Triple M Ranch in order to inform management decisions in the area and further knowledge of riverine wetlands with unregulated inflows. Sediment directly affects habitat by creating and burying habitat, abrading soft tissues, light attenuation, and by transport of adsorbed nutrients, pesticides, other organic compounds, and metals. As treatment of NPS pollution by wetland restoration in large agricultural watersheds becomes more common, their performance under unregulated inflow is useful information for project design and subsequent land management. The results from ALBA's Triple M Wetlands indicate efficient sediment retention under highly variable residence times caused by the unregulated inflows.

ALBA's Triple M Project

Suspended sediment concentrations through the site were reduced by 71% as inferred from the reduction in suspended load, and this is above the 65% goal set by the management plan at the site, even before active restoration (ALBA 2008; Largay 2007). The high retention rate and resulting high vertical accretion rate indicates that some areas, especially where bedload is deposited, could prematurely fill in and convert to riparian forest. It is important to note that this study occurred during a relatively normal year, and that decadal-scale El Nino events will move much larger pulses of sediment. To avoid damage to the marsh fields at the downstream end of the property during such events, the upstream floodplain could be maintained at a low grade so it is able to trap high loads. Areas with observed high deposition rates, such as the primary floodplain, should be monitored, and sediment accommodation space should be re-excavated following extreme depositional events, as management did in 2001 and 2005. This episodic maintenance will protect the large marsh fields downstream from losing their wetland function through rapid infilling. Similarly, the planned ponds, designed to enhance amphibian habitat, will likely fill in at a rate higher than their surroundings and could need periodic sediment removal over the long term as well.

Without tide gates located two kilometers downstream of Sill Road, the Sill Road ford would occasionally be inundated on high spring tides (Largay 2007). The average rate of deposition, 7.5 mm/yr, calculated by dividing the retained volume by the wetland area indicates that the 1.3 mm/yr rate of sea level rise for the area (NOAA 2010) will not affect this site. Furthermore, this site should be resilient to even greater rates of sea level change should that occur. However, the average deposition rates do not account for spatial variation as was found in the floodplain (Bassett 2010), which may cause the marsh fields above Sill Road to have much lower than average deposition rates and less ability to balance out sea level rise.

The relation between discharge at Sill Road and residence time for the site can be used to design future water quality sampling strategies. However, the presence of dye concentration spikes before and after the centroid weakens the conclusions to be drawn from comparing such paired samples. Further modeling of the timing of the two spikes at a range of discharges could allow collection such paired samples Future modeling of rainfall-runoff can utilize the initial discharge rating curves in this study. Additionally, the residence times for this specific site can be primary inputs to modeling of wetland geochemical functions.

To increase residence time and corresponding water quality benefits, ALBA should consider filling in the old channel and drainage ditches around the marsh fields at the lower end of the wetlands. This would diminish the initial spike seen on the residence time distributions (Fig. 7) by blocking rapid channelized flow from outpacing the slower, unchannelized flow, thereby increasing the time in the wetland for a larger proportion of the runoff. Extremely high flows would still experience the initial spike though, as the lower fields become so full that a zone of unconfined rapid flow develops through the flat middle of the fields even with the channels still present on the edges. Another option to increase residence time through the lower fields could be to create a portion of long and winding channel similar to the nearby Molera experimental wetland. While this would not be feasible in the main section where large flows would destroy it, a small side tributary carrying runoff from a nearby mushroom farm could especially benefit from this approach.

Elkhorn Slough

The sediment rating curves from this study can be used to estimate the loads passing into Elkhorn Slough from the Carneros Creek watershed, and to quantify the larger sediment budget for the Slough. Concerning the sediment trapped in the Triple M Wetlands, managers may eventually need to weigh the benefit of allowing sediment passage to the Slough, with the benefits of sediment retention such as removal of nitrate or phosphate.

The bedload particle size distribution and other aspects of hydrology from this study can aid in the design of a channel through the wetlands to convey sediment to the Slough. Elkhorn Slough is losing approximately 120,000 m^3 of sediment in its main channel, and the 150 tonnes of bedload that could possibly be transported would only amount to 88 m^3. The amount of suspended sediment passed through the Triple M Wetlands in WY2010 of 1146 tonnes amounts only to 663 m^3. However, these small contributions to the Slough might be beneficial to marshplains located near the mouth of Carneros Creek where tidal scour has not been as destructive.

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APPENDICES

APPENDIX A

INTRODUCTION TO THE SCIENCE AND POLICY OF WETLANDS AS A BEST MANAGEMENT PRACTICE FOR TREATING AGRICULTURAL RUNOFF IN CARNEROS WATERSHED, CA

Introduction

Wetland Loss

Wetlands loss over the last two centuries in the United States has been extensive because they were viewed as a breeding ground for mosquitoes and disease (Vileisis 1997). These views caused wetlands to be drained and filled in order to make them more productive to society through conversion to agriculture, as they contain highly productive soils, and to urban development as they are flat and often located near waterways. The underlying reason for loss of wetland areas was the need for food and housing by the rapidly growing human population. As recently as the late 1960s the federal Agricultural Stabilization and Conservation Service promoted drainage of wetlands through cost-sharing programs with farmers, and much earlier programs such as the Swamp Land Acts of the 1800's actively encouraged draining and filling in of wetlands. The United States Forest Service estimates that in the 1750's there were over 344,000 square miles of wetlands in what is now the contiguous United States (Dahl 1990). Well over half of that acreage was gone by the year 2000, and agricultural usage had accounted for about 80% of that loss (Dahl and Allord 1996).

While wetland loss has decreased markedly over the last few decades, it continues, with 23,000 hectares lost in the 1980's (Dahl and Allord 1996). California has lost 90% of its wetlands representing the greatest loss in the nation (Mitsch and Gosselink 1993, National Wetlands Inventory). Urban development and agriculture caused especially large loss of the State's coastal wetlands, where only 5% of original wetlands remain (CERES 2008; NRASC 2009; Whigham 1999). Late 20th century research has highlighted wetland services and the magnitude of wetland loss. The growing recognition of impaired water quality in the nation's rivers and the loss of wetlands' beneficial functions resulted in the 1972 Clean Water Act (CWA).

Nonpoint Source Pollution

The land conversion to agricultural and housing uses has also resulted in impacts to water quality and ecological function in the form of nonpoint source (NPS) pollution. NPS pollution occurs when water runs over land, picking up and carrying pollutants into surface water resources and groundwater systems. Types of NPS pollutants include high concentrations of nutrients, sediment, pesticides, pathogens, or heavy metals. The NPS pollutants are different depending on land use, carrying high loads of oils and heavy metals from urban areas, nutrients, pesticides and sediments from agricultural lands (Woodward and Foster 1997; Viessman and Hammer 1993; Gellis 2009), and bacteria and sediment from livestock.

NPS is now the greatest source of surface and groundwater quality degradation in the US (Baker 1992). NPS pollution resulting from agricultural, livestock, and urban land use or development has caused widespread decreases in water quality (Howarth et al. 1996). Agriculture is the single largest contributor of NPS pollution nationwide and in California, with sediment, pesticides, and nutrients most responsible for water quality degradation (Baker 1992; EPA 1992; SWRCB 2000). The global increase in agricultural and urban land-use conversion has made humans the primary agent of geomorphologic change on earth, and the largest component of the exponentially increasing rate of anthropogenic earth movement is unintended land loss from agricultural erosion (Hooke 2000).

Congress amended the CWA in 1987 to include section 319 aimed at controlling and reducing sources of NPS pollution. Section 319 required states to assess and manage NPS pollution problems, and established an EPA grant program to fund state management programs. Current EPA guidance on Section 319 suggests that grants focus on impaired water bodies on the 303(d) list, and also establish management plans and caps on pollutant loads over a given time, both of which are commonly called TMDL’s or total maximum daily loads.

Policy: The CWA and Wetlands

The Clean Water Act of 1972 (CWA) signaled a new direction for federal policy and became the main law for managing wetland issues in the United States.

The Environmental Protection Agency administers the act, which contains two sections that commonly affect wetland management. Section 401 regulates water quality and state rights in the certification of water quality, while Section 404 regulates discharge of dredge and fills material into waterways, including wetlands. The CWA requires avoiding or minimizing wetland impacts when possible and mitigation when not possible. Mitigation is the replacement of destroyed areas by creation or restoration of other equal areas.

In the decade after the CWA, continued wetland loss from agricultural conversion prompted new federal legislation. In the 1980's the Swampbuster and Wetland Reserve Program decreased loss from agricultural conversion, which currently amounts to 30% of the total (Dahl and Allord 1996). The Swampbuster provision of the Food Security Act protects wetlands by withholding Federal benefits to farmers who degrade their wetland acreage. Other relevant federal laws are the Endangered Species Act (ESA), the 'Farm Bill', and the Coastal Zone Management Act. The California Environmental Quality Act (CEQA) serves as the state's basic environmental protection law. In California numerous departments such as Fish and Game, Water Resources, Coastal Commission and Conservancy, Parks and Recreation, as well as the California Environmental Protection Agency (CEPA) are responsible for wetland management.

The problems of wetland loss and mismanagement have improved under modern laws but many problems persist. Monitoring and enforcement usually lag far behind policy enactment. For example, standard evaluation of water quality is only available for 8% of remaining wetlands (NWI 2009). In addition, many administrations have underfunded agencies responsible for wetland management such as the EPA (Rosenbaum 2003), and budget problems in California compound the fiscal problems. Modern federal wetland initiatives aim to restore large areas of wetlands and streams (EPA 2000), and are often funded through mitigation projects.

The rise of mitigation as a primary policy response to wetland degradation is an area of concern to many involved in wetland science. Federal and state adoption of 'no net loss' policies led to less preservation of natural wetlands and more mitigation. While preservation and restoration of any wetland acreage regardless of its quality helps to stem the historic losses (Hey and Phillipi 1999), the assumption that mitigated or restored wetlands function equally to natural ones is not always defensible, especially in the short term (Whigham 1999). Mitigation work by the primary state and federal wetland protection programs has resulted in a net reduction in wetland quality, as it commonly creates lower quality wetlands to replace natural wetlands of higher quality (Ambrose et al. 2006). Many degraded wetlands are difficult to restore, and after restoration can take decades or more to match natural function levels (Zedler 2000). Another problem in policy has been agreement on the legal definition of a wetland. Scientists define wetlands primarily by associated types of vegetation, soil, and hydrology. There is general agreement that scientists can determine the presence of a wetland by these factors but the many possible combinations have made a universal definition problematic.

Wetland Types

People refer to wetlands as swamps, fens, potholes, playa lakes, mangroves, marshes (salt, brackish, intermediate, and fresh), forested wetlands, bogs, wet prairies, prairie potholes, and vernal pools. Although these places can differ greatly, they all have distinctive plant and animal communities because of the wetness of the soil. Water continuously floods some wetland areas (permanent) while other areas only flood for a short time of the year (seasonal). Scientific classification of wetlands divides them into five major categories: marine, estuarine, lacustrine, riverine, and palustrine (Cowardin 1979). Marine and estuarine wetlands are associated with the ocean and include coastal wetlands, such as tidal marshes. Lacustrine wetlands are associated with lakes, while riverine wetlands form along rivers and streams. Palustrine wetlands may be isolated or connected wet areas and include marshes, swamps, and bogs.

Natural wetlands are areas that have historically been undisturbed, created ones are established where none existed before, and restored ones take what was once a natural area that was modified or degraded and reestablishes wetland function (Hammer 1997). Most created wetlands are constructed specifically to improve or treat impaired water and are called treatment wetlands. Treatment wetlands are projects to improve water quality by reducing constituents including nitrogen, phosphorus, heavy metals, pesticides, and sediment (Kadlec and Knight 1996; Mitsch and Gosselink 2007).

Wetland Benefits

Wetlands provide a variety of benefits to society, and recognition of these benefits prompted changes in policy and efforts to preserve, restore, and construct them. Modern studies indicate that shallow waters such as wetlands, which only cover 1.5% of earth’s surface, provide 40% of renewable ecosystem services and have a yearly dollar value of 40 trillion (Zedler 2000). Wetlands are highly productive and diverse habitats on par with coral reefs and tropical forests.

Some major benefits provided by wetlands are:

Habitat for aquatic birds and other animals and plants, many of them endangered
Production and nursery of fish and shellfish
Flood and drought relief
Water storage
Soil retention
Water quality improvement
Recreational, open space and aesthetic values
Food and timber production
Carbon sequestration

One of the most important benefits wetlands provide is the improvement of water quality. Wetlands improve water quality by reducing the concentration of pollutants such as excess nitrates, phosphorus, sediments, pesticides, and pathogens (Blahnik and Day 2000; Fisher and Acreman 2004; Jordan et al. 2003; Mitsch and Gosselink 2007; Raisin and Mitchell 1994; Richardson, 1989). Research on treatment wetlands has found the ratio of wetland surface area to contributing watershed area predicts effectiveness of treatment, and pollutant removal correlates with retention time and hydraulic loading rates (Carleton 2001, Kadlec and Knight 1996). Plant diversity and extent also affect water quality treatment up to a point, and then storage capacity and residence time become most important (Line et al. 2008). The hydraulic residence time through a wetland is closely tied to the ability to trap and retain sediment, which is a key process in general water quality improvement (Blahnik and Day 2000; Carleton et al. 2001; Woltemade 2000). Sediment retention and aggradation are the key geomorphic processes of wetlands since they are typically regions defined by local low elevation and low hydraulic gradient (Johnston 1991; Mitsch and Gosselink 2000; Phillips 1989).

While constructed wetlands have proven to be effective at treating water for many NPS pollutants and have been the subject of most studies (Hammer 1989; Kadlec and Knight 1996), the efficacy of restored or native wetlands is not as well documented. Despite this lack of assessment, land managers now use restored or natural wetlands as a tool to improve water quality and maintain habitat in agricultural areas (Whigham 1999, Zedler 2003). Restored and natural wetlands often differ from constructed ones by receiving highly variable, or unregulated, inflows, which create more complex flow paths, mixing, and residence time variability (Holland et al. 2004; Jordan et al. 2003; Knox 2008; Stern et al. 2001; Werner and Kadlec 1996; Wetzel 2001). Residence time variability due to unregulated flow may make sediment retention more difficult to predict. This aspect of wetland function has not been rigorously studied and remains poorly understood (Stern et al. 2001). The efficacy of restored and natural wetlands with unregulated inflow to treat agricultural runoff and particularly excess sediment is uncertain because of the variability in residence times, inflow rates, and amount of channelization (Hammer and Kadlec 1986; Holland et al. 2004; Knox 2008; Wetzel 2001).

NPS Pollution in Elkhorn and Carneros Watersheds

The water quality in the Carneros and Elkhorn watersheds in Monterey County on California's central coast are impacted by NPS pollution and are areas where wetland restoration and watershed management is pursued (Figure A1). Agricultural runoff is the primary stressor to ecosystem function in these watersheds, causing high concentrations of sediment, nutrients, and pesticides (ESTWPT 2007). These high concentrations of NPS pollutants especially affect the freshwater reaches of upper Elkhorn Slough and lower Carneros Creek, which is habitat for federally listed amphibians (ESTWPT 2007). The Clean Water Act's regional 303(d) listings currently include Carneros Creek for ammonia and Elkhorn Slough for pathogens, pesticides, and sedimentation/siltation (CCRWQCB 2009). Water quality studies show both areas exhibit effects of NPS pollution such as eutrophication, with oxygen levels below 5.0 mg/l in the summer. Carneros Creek has also had chlorophyll *a* levels of 1000 mg/l, and extremely high orthophosphate concentrations (CCAMP 2000). Orthophosphate water pollution is derived from agricultural land application, and along with nitrogen, phosphorus input encourages eutrophication in wetlands (Elser 2007).

Sediment in Elkhorn Slough

The lateral erosion of marsh and mudflat habitat and NPS-derived water quality issues are two of Elkhorn Slough's major environmental problems, leading to a dual role for sediment in this system (ABA 1989; CCRWQCB 2009; ESNERR 2008). The Slough is losing ecologically valuable marsh habitat in part because the input of fine sediment from Carneros Creek is insufficient to replace the sediment lost from erosion. However, the sediment and water inputs from Carneros Creek are contaminated and negatively affect habitat quality in the Slough. The major historic sources of sediment to the Slough used to be the Salinas and Pajaro rivers, but harbor construction in 1947 resulted in most of their sediment being output to Monterey Bay (PWA 2008).

The Elkhorn Slough Tidal Wetland Project has a goal of quantifying the sediment budget for Elkhorn Slough to determine if marsh restoration projects would be sustainable (ESTWP 2007), but limited data exist to quantify the sediment budget (PWA 2008). Given the loss of sediment from the historic sources, Carneros Creek was identified as one of the few remaining potentially significant sediment contributors to Elkhorn Slough (PWA 2008). Agencies including the Agriculture and Land Based training Association, Elkhorn Slough Foundation, Resource Conservation District of Monterey County, and Natural Resources Conservation Service have initiated many agricultural best management procedures in cooperation with growers and private landowners, land acquisitions, and wetland restoration projects in the area to improve water quality and restore or retain diverse habitat around Elkhorn Slough (ESNERR 2008). These management activities have also reduced the sediment load entering the Slough as a byproduct of improving non-point source pollution. Thus, watershed management activities focused on water quality may have the unintended consequence of exacerbating marsh erosion in Elkhorn Slough.

ALBA's Triple M Wetland Restoration Project

As an example of regional best management practices, the Agriculture and Land-Based Training Association, ALBA, is engaged in management of a wetland restoration project at Triple M Ranch with goals of improving water quality while creating and conserving wetland habitat. The Triple M Ranch is situated at the lower end of the Carneros Watershed (Fig. A1), so it has the potential to trap a large proportion of sediment before it reaches Elkhorn Slough. ALBA has managed the ranch since 2001 after the Elkhorn Slough Foundation acquired an easement in 2000 barring development of the property for a housing complex. The wetlands of the Triple M Ranch were channelized and drained for agriculture with the assistance of the Agricultural Stabilization and Conservation Service in the mid-twentieth century and the drainage ditches were maintained through the 1980's by the local mosquito abatement district (Largay pers. comm. 2010).

The site is currently in a state of "self-restoration" (sensu Smith et al., 2009) as management has allowed the levees to degrade and reestablish wetland hydrology in preparation for more extensive modifications that are part of the restoration plan. Two specific objectives of the restoration plan are reducing turbidity and suspended solids exiting the property in Carneros Creek by 65 percent (ALBA 2008; Largay 2007; Reis 2007). ALBA's approach at Triple M is adaptive management, which involves applying ideas from research, monitoring data, and then appraising the results and optimizing parts of the system to achieve desired goals (Christensen et al. 1996). In this case, the goals are retaining habitats while reducing excess sediments and water quality constituents such as nutrients, pesticides, and pathogens (Largay 2007, ALBA 2008). The reductions are expected to occur as water flows through the wetland (Kadlec and Knight 1996; Mitsch and Gosselink 2007). Parameters affecting water quality improvement include surface water entry and exit rates, initial load of constituents, vegetation type, land use, and residence time through the site. To some extent, ALBA can vary these parameters, except entry rate and initial loads, and then monitor the changes in water quality exiting the property.

The Triple M site has an important role in the area and lies at the transition from freshwater to marine influence. In this system, precipitation falls on the Carneros watershed and conditions such as land use, antecedent moisture, climate, geology, and soil type affect the timing and amount of water and sediment discharge entering the Triple M Ranch at Johnson Bridge (Fig. A2). The Triple M wetlands then affect the timing and amount of water and sediment discharges being output from the site at Sill Rd (Fig. A2). After exiting the site, discharges then move downstream through Porter Marsh to Elkhorn Slough and Monterey Bay (Fig. A1 and A2).

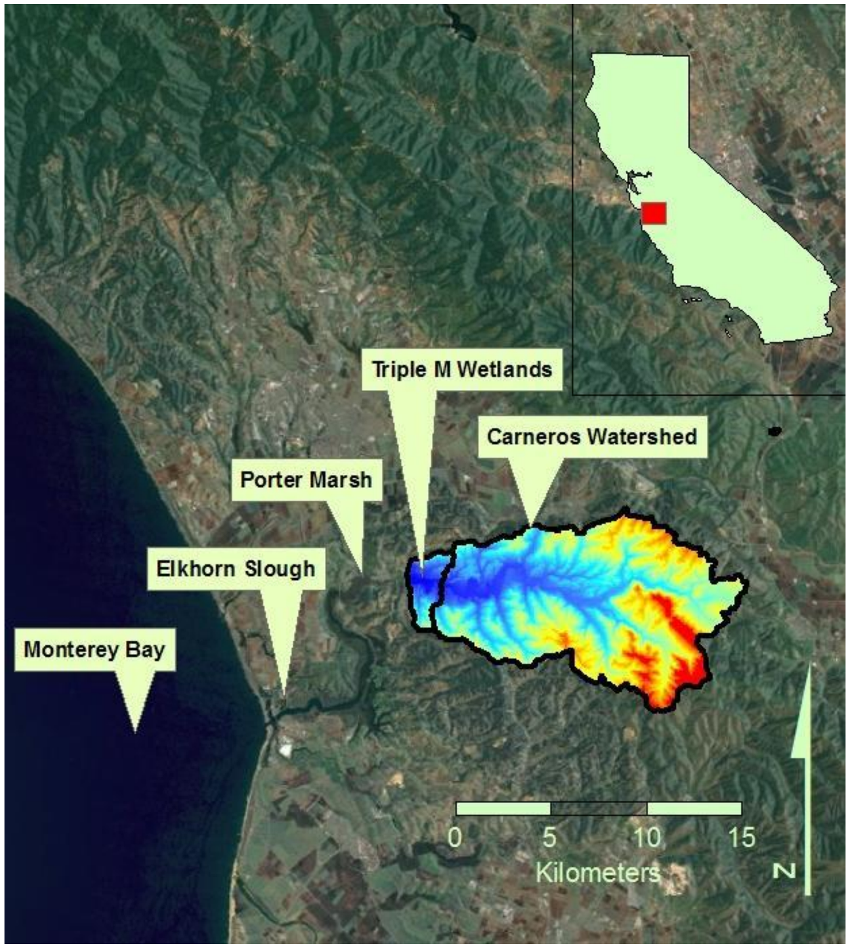

Figure A1. Study area of the Triple M Wetlands in a regional context and location in California. The outline of the Carneros Watershed is in two sections corresponding to the drainage above the Sill Road outlet and the Johnson Road inlet of the wetlands. Representations of watershed elevations from high to low are by bright red to dark blue color. The watershed discharges into Porter Marsh at the top of Elkhorn Slough, an estuary on Monterey Bay.

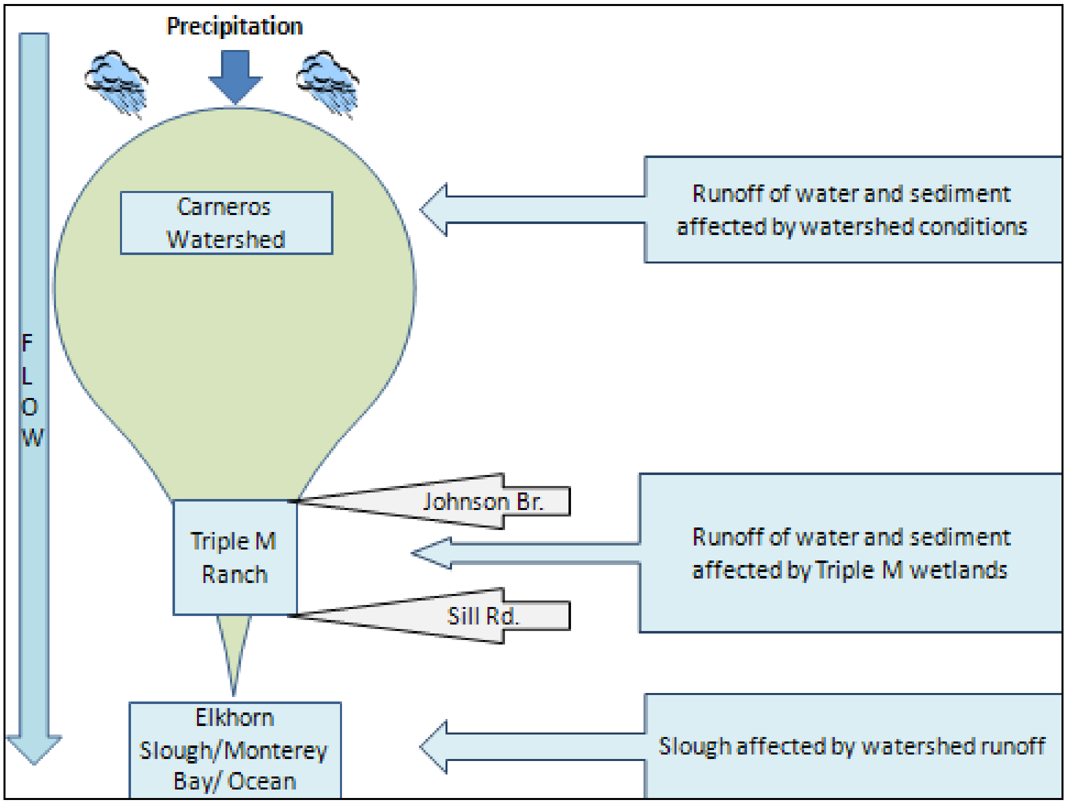

Figure A2. Context of the Triple M Ranch and wetlands in the hydrologic system. The Johnson Bridge and Sill Road sites are at the inlet and outlet of Carneros Creek on the property.

Research Objectives

The Triple M Ranch in the Carneros Creek watershed is the site of a riverine wetland where no quantitative estimate of sediment transport or deposition has been developed (Largay 2007). The primary objective of this study was to investigate sediment and hydraulic dynamics at the Triple M Ranch in order to inform management decisions in the area and further knowledge of riverine wetlands with unregulated inflows. The specific objectives were to:

1. Quantify a sediment budget with input, output, and storage at the site (Fig. A3)
2. Determine hydraulic residence time distributions and associated parameters of time to leading edge, peak, centroid, and trailing edge through the site at a range of values of stage and discharge.

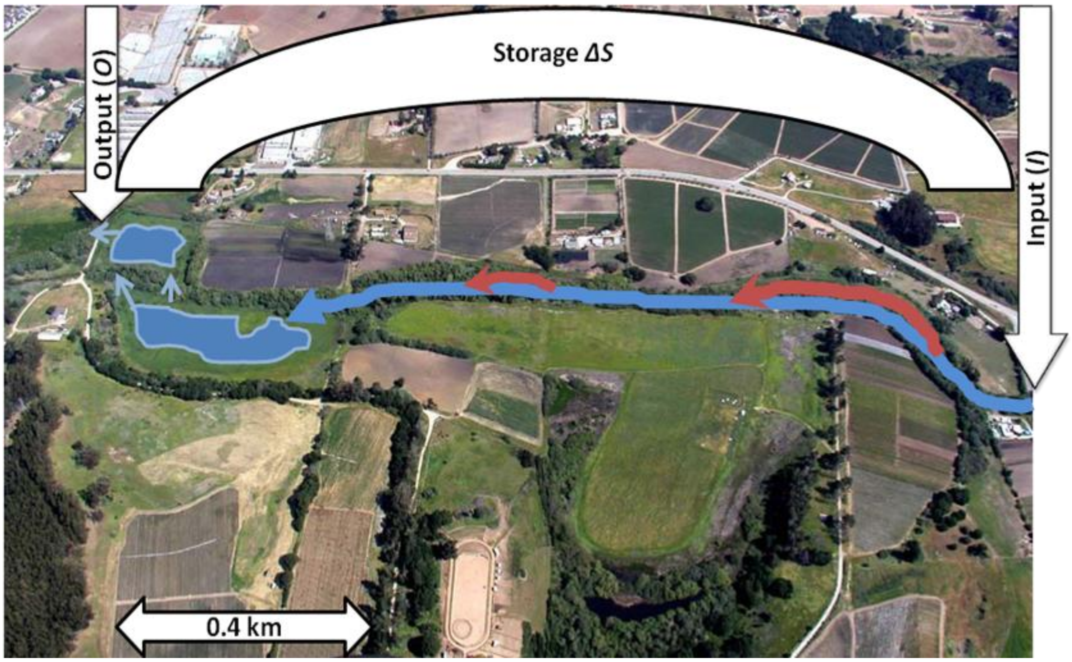

Figure A3. Illustration of the primary sediment budget components at the Triple M Wetlands. Normal flow path is shown in blue with flood flow paths in red. A new road crossing with adjustable culvert gates is planned at the output, and ponds and islands

Policy Applications

The information from this study can directly support the adaptive management decisions of ALBA and ESNERR by quantifying the amount of sediment and water inputs and output at the site. The sediment dynamics are important for determining if habitat will be lost through the site filling in, maintaining site elevation against sea level rise and as a possible supply source for the Elkhorn Slough. The residence times are important for accurately sampling suspended sediment concentration reduction through the site, and modeling of geochemical functions such as reductions of nitrates and agricultural pesticides. The results provide data to monitor changes upstream of the site in the form of a discharge-rating curve, sediment-rating curves, and bedload particle size at the input (Fig. A3). The results provide similar data at the output to monitor changes within the site in advance of planned modifications, such as installation of adjustable flow gates at the outlet, and construction of vegetated ponds and islands in the marsh fields (Fig. A3).

One of the goals of ALBA's restoration plan is to reduce suspended load and bedload exiting the Triple M wetlands (ALBA 2008; Largay 2007). This research determined the reduction in load through quantifying the mass of sediment retained during the study period. Sediment budget results from this study can predict the longevity of the Triple M Wetlands under different action plans and in response to environmental change. For example, if the volume of this mass is calculated and spread over available deposition areas it would reveal elevation change in those areas. The elevation change influences the response of the site to sea level rise and anticipates the timeframe for conversion of marshes to riparian forest due to sedimentation. Sedimentation is currently a threat to endangered species at the site (Reis 2007). The timeframe and amounts of sediment storage could trigger management actions to collect and move sediment accumulations with heavy equipment as done in 2001 and 2005 (Laurel Marcus & Assoc. 2003).

The results of the residence time analysis will improve monitoring strategies for the Triple M wetlands by reducing error and variation caused by sampling different parcels of water. Samples from the same parcel of water evaluate the site's effectiveness at improving water quality and the changes induced by adaptive management. Models of wetland geochemical function also rely upon estimates of residence time to determine reductions of nitrate and agricultural pesticides, which are both present in the Carneros watershed (Mitsch and Gosselink 2007, Stern et al 2001200).

This study provides baseline data in advance of management activities that might change the sediment dynamics or residence time properties at the site or in the watershed. For example, management can decide to modify pathways of flow, rates of flow in and out of different wetland sections, composition of vegetation, or the distribution of land uses. Two such planned modifications for which ALBA has received EPA funding are constructing vegetated ponds and islands in the marsh fields and replacing the Sill Road stream crossing with a water control structure. The ponds and islands are to act as breeding habitat for federally listed California red-legged frogs, California tiger salamander, and Santa Cruz long-toed salamanders (ALBA 2008; Denise Duffy & Assoc. 2008). The water control structure will allow control of residence time and wetland hydrology by adjustable gates placed in the culverts, and is meant to protect habitats in the case of changing climatic or watershed conditions (ALBA 2008). The gates will initially be set to the same elevation as the current crossing but can be adjusted, for example, to change residence times at the site. The sediment-rating curve for Sill Road and the residence time data from this study can determine the effects of adjusting the gates or installing the ponds and islands by comparing them to future data. Changes in the upstream watershed characteristics can also be determined through the results of this study. Comparison of the load and sediment-rating curves to future ones at Johnson Road can indicate a change in the watershed properties upstream (Watson et al. 2003). Likewise, the discharge records can be compared to future results for use in modeling of rainfall to runoff.

This study has implications for management of Elkhorn Slough. Elkhorn Slough is currently experiencing loss of marsh and mudflat habitat due to tidal scour and diminished sediment input. Carneros Creek has been identified as a possible significant sediment source to alleviate that loss of habitat (ABA 1989; ESTWP 2009). However, improving agricultural practices to reduce erosion and establishment of the Triple M wetlands to improve water quality has limited the transport of Carneros Creek's sediment to the Slough (PWA 2008). Tide gates at Parson's Slough near Porter Marsh also disconnect water and sediment delivery from Carneros Creek to the main Slough (Fig. A3) (Caffrey et al. 2007). This sediment budget study quantifies the feasibility of reducing marsh and mudflat habitat loss by conveying more loads, especially bedload, through Triple M Ranch. If ALBA and ESNERR decide to convey more sediment through the Triple M wetlands, this study will provide sediment rating curves and yields, and mean bedload particle size to design a new stream channel. The Elkhorn Slough Tidal Wetland Plan intends to create a sediment budget for the Slough to quantify existing and historic sediment sources (ESTWP 2009). This large-scale budget will determine if marsh restoration projects are sustainable. The results of this study quantify Carneros Creek's current and possible inputs to that large-scale budget.

On a broad scale, the results will not only inform adaptive management in the region, but also increase the limited scientific knowledge of how effective unregulated flow wetlands are at treating sediment load and concentrations. The scientific knowledge can then affect decisions on how to proceed with restoration projects and manage regulatory problems of NPS pollution.

APPENDIX B

DATA SUMMARY TABLES FOR SEDIMENT RATING CURVES

Table A1. Suspended sediment data for Johnson Bridge.

Johnson Br. Sample Date and Time	Johnson Br. Suspended Load (g s^{-1})	Johnson Br. Discharge ($m^3 s^{-1}$)
10/13/2009 15:55	13436.8	3.71
10/15/2009 10:36	4.1	0.04
12/13/2009 9:22	12.5	0.09
12/16/2009 10:00	7.1	0.01
12/27/2009 8:15	34.7	0.06
1/18/2010 10:40	314.1	0.23
1/19/2010 13:00	6874.6	4.14
1/21/2010 13:30	311.3	1.03
1/22/2010 11:00	3965.4	5.36
1/26/2010 13:00	232.4	0.86
1/27/2010 14:05	13.5	0.35
1/29/2010 11:20	4.6	0.18
2/5/2010 10:03	57.7	0.46
2/6/2010 10:00	13.4	0.22
2/6/2010 12:00	7.4	0.21
2/9/2010 9:35	273.4	0.61
2/9/2010 10:50	203.3	0.63
2/9/2010 15:50	91.5	0.64
2/10/2010 10:00	10.6	0.21
2/12/2010 14:30	6.1	0.13
2/15/2010 10:00	2.2	0.10
2/19/2010 13:30	3.5	0.08
2/21/2010 13:00	6.9	0.09
2/23/2010 14:30	8.9	0.11
2/24/2010 10:30	3406.8	5.53
2/24/2010 16:00	468.5	2.38
2/25/2010 10:50	28.3	0.50
2/25/2010 15:00	58.2	0.45
2/26/2010 15:15	115.5	0.30
2/26/2010 16:15	363.9	0.50
2/27/2010 8:45	911.1	4.13
2/27/2010 10:30	1158.1	4.34
2/27/2010 13:00	579.0	3.55
3/2/2010 11:15	6888.6	3.92
3/2/2010 15:00	4263.3	7.25
3/2/2010 17:15	3196.3	8.26
3/3/2010 11:15	4892.3	5.36
3/4/2010 10:45	740.7	4.74
3/5/2010 10:45	87.8	1.16

3/7/2010 10:40	12.4	0.54
3/9/2010 13:35	12.3	0.31
3/10/2010 12:45	117.3	0.56
3/13/2010 11:00	16.5	0.42
3/23/2010 15:30	1.0	0.10
3/28/2010 9:15	2.8	0.09
3/30/2010 12:20	0.8	0.12
4/1/2010 15:10	0.5	0.09
4/12/2010 10:30	1693.9	2.05
4/14/2010 12:30	19.2	0.31

Table A2. Suspended sediment data for Sill Road.

Sill Road Sample Date and Time	Sill Road Suspended Load (g s^{-1})	Sill Road Discharge (m^3 s^{-1})
10/15/2009 10:54	22.9	0.12
12/13/2009 11:31	9.4	0.08
12/16/2009 12:30	0.2	0.00
12/27/2009 9:05	0.0	0.00
1/18/2010 11:40	16.8	0.24
1/19/2010 13:41	1109.2	1.89
1/21/2010 14:25	132.9	1.43
1/22/2010 11:45	1311.7	3.41
1/26/2010 14:15	135.0	0.86
1/27/2010 14:10	19.5	0.54
1/29/2010 12:32	4.1	0.27
2/5/2010 10:55	44.3	0.60
2/6/2010 10:30	13.9	0.26
2/6/2010 13:10	8.8	0.22
2/7/2010 10:25	3.3	0.15
2/7/2010 11:10	3.1	0.18
2/9/2010 9:18	7.3	0.24
2/9/2010 10:30	3.4	0.31
2/9/2010 16:15	39.8	0.72
2/10/2010 10:15	18.5	0.30
2/11/2010 13:00	3.7	0.13
2/11/2010 15:00	2.9	0.13
2/12/2010 15:00	4.0	0.11
2/15/2010 10:15	1.9	0.07
2/19/2010 14:00	1.0	0.04
2/21/2010 13:15	2.0	0.03
2/23/2010 15:00	0.2	0.05
2/24/2010 11:30	1208.1	4.86
2/24/2010 14:05	796.7	4.92
2/25/2010 10:15	40.7	0.76
2/25/2010 15:45	27.7	0.58
2/26/2010 14:45	6.8	0.32
2/26/2010 16:00	9.0	0.35
2/27/2010 9:45	533.3	3.41
2/27/2010 13:15	354.5	3.40
3/2/2010 11:45	384.4	0.88
3/2/2010 16:30	2748.8	4.80
3/3/2010 12:00	531.2	2.85
3/4/2010 10:30	537.5	6.18

3/5/2010 11:45	37.2	1.11
3/7/2010 10:25	3.7	0.40
3/9/2010 12:30	4.3	0.20
3/10/2010 13:00	33.2	0.39
3/13/2010 11:30	8.5	0.37
3/23/2010 15:00	0.0	0.00
3/28/2010 14:30	0.0	0.00
3/30/2010 12:10	0.0	0.00
4/1/2010 15:00	0.0	0.00
4/12/2010 11:30	301.0	0.83
4/14/2010 12:45	5.7	0.14

Table A3. Bed load data for Johnson Bridge.

Johnson Br. Sample Date and Time	Johnson Br. Bed Load ($g\ s^{-1}$)	Johnson Br. Discharge ($m^3\ s^{-1}$)
12/13/2009 9:22	0.9	0.09
12/27/2009 8:15	0.0	0.06
1/18/2010 10:40	3.2	0.23
1/21/2010 13:30	2.8	1.03
1/26/2010 13:00	0.8	0.86
1/29/2010 11:20	0.5	0.18
2/5/2010 10:03	2.1	0.46
2/6/2010 12:00	5.1	0.21
2/9/2010 9:35	5.9	0.61
2/9/2010 15:50	2.8	0.64
2/23/2010 14:30	1.2	0.11
2/25/2010 15:00	13.2	0.45
2/26/2010 15:15	2.0	0.30
3/2/2010 15:00	347.6	7.25
3/2/2010 15:15	337.9	7.04
3/2/2010 17:15	358.0	8.26
3/3/2010 11:15	127.9	5.36
3/5/2010 10:45	49.4	1.16
3/9/2010 13:35	17.9	0.31
3/10/2010 12:45	10.4	0.56
3/13/2010 11:00	13.4	0.42

Table A4. Bed load data for Sill Road.

Sill Road. Sample Date and Time	Sill Road. Bed Load (g s^{-1})	Sill Road Discharge (m^3 s^{-1})
10/15/2009 10:54	0	0.12
12/13/2009 11:31	0	0.08
12/27/2009 9:05	0	0.00
1/18/2010 11:40	0	0.24
1/21/2010 14:25	0	1.43
1/26/2010 14:15	0	0.86
1/29/2010 12:32	0	0.27
2/5/2010 10:55	0	0.60
2/6/2010 13:10	0	0.22
2/9/2010 9:18	0	0.24
2/9/2010 16:15	0	0.72
2/23/2010 15:00	0	0.05
2/25/2010 15:45	0	0.58
2/26/2010 14:45	0	0.32
3/2/2010 16:30	0	4.80
3/2/2010 0:00	0	0.45
3/2/2010 0:00	0	0.45
3/3/2010 12:00	0	2.85
3/5/2010 11:45	0	1.11
3/9/2010 12:30	0	0.20
3/10/2010 13:00	0	0.39
3/13/2010 11:30	0	0.37

APPENDIX C

EXAMPLE OF R STATISTICAL PROGRAM CODE FOR DISCHARGE-RATING CURVE COEFFICIENTS

```
# (LSillDRC is load data in g s-1 and QSillDRC is discharge in m3s-1)
LSillDRC <- SSSCgps
QSillDRC <- SillQeither
maxQSillDRC <-max(QSillDRC,na.rm=T)
QvecSillDRC = seq(0,maxQSillDRC,length.out=100)
# Fit a power-function with gamma error:
logQSillDRC = log(QSillDRC)
mSillDRC = glm( LSillDRC ~ logQSillDRC, Gamma(log) )
beta0SillDRC = mSillDRC$coef[1]
beta1SillDRC = mSillDRC$coef[2]
beta0SillDRC
beta1SillDRC
summary(mSillDRC)
# Bootstrap m alternative model fits, to obtain a distribution for the variable:
n=length(LSillDRC)
m=10000
bootcoeff0SillDRC <- rep(NA,m)
bootcoeff1SillDRC <-  rep(NA,m)
for(j in 1:m) {
ivec=sample(1:n,replace=TRUE)
Qj = rep(NA,n)
Lj = rep(NA,n)
for(i in 1:m) {
Qj[i]=QSillDRC[ivec[i]]
```

```
Lj[i]=LSillDRC[ivec[i]]
}
logQj=log(Qj)
mj = glm( Lj ~ logQj, Gamma(log) )
#get bootstrap coeffs
#backtransform coefficient 1 with e to the x
bootcoeff0SillDRC[j] = 2.718281828^mj$coef[1]
bootcoeff1SillDRC[j] = mj$coef[2]
}
# obtain coefficients for bootstrapped models
bootcoeff0SillDRC
bootcoeff1SillDRC
```

APPENDIX D

DISCHARGE RATING CURVES

Discharge-rating curves for the Johnson and Sill Road sites were developed by regressing instantaneous discharge measurements against associated stage data from WY2010. The goal was to find the best fit regardless of model form, especially for high discharge as that is where more load is moved. Linear, two-parameter power, and third order polynomial trend lines were compared and r-squared values used to pick best fit. The fit of a single model to all discharge data at both Johnson and Sill roads was deemed unacceptable due to bad fit on the middle of the Johnson curve and high end of the Sill curve (Fig. A4, A5).

Data at each site were split into low and high segments to obtain closer fit to data. For Johnson Bridge data the segments were separated based on geomorphology of the channel, using the stage (Z) at which water reached the concrete bridge structure on the left bank (Fig. A6, A7). Stage is zero at the location of the pressure transducers installed at each site. The best-fit model for Johnson Bridge was the power for the low segment ($Z<2.861$, $R^2= 0.987$):

$$Q = 0.1174 \times Z^{1.9194}$$

The third order polynomial was the best-fit for the high segment ($Z>2.861$, $R^2=0.981$):

$$Q = (-0.1884 \times Z^3) + (3.1569 \times Z^2) - (12.741 \times Z) + 15.79$$

where Z is stage (psi) and Q is discharge (m^3s^{-1}). For Sill Road data the segments were separated based on visual change of slope in the stage to discharge graph combined with estimation of stage at which flow becomes widespread over the Sill Road crossing (Fig A8, A9). The best-fit models for Sill Road were the power for the low segment ($Z<2.6105$, $R^2= 0.997$):

$$Q = 2.08E11 \times Z^{24.203}$$

The power was also the best fit for the high segment (Z>2.6105, R2=0.991):

$$Q = 0.00001724 \times Z^{9.8854}$$

where Z is stage (psi) and Q is discharge (m^3s^{-1}).

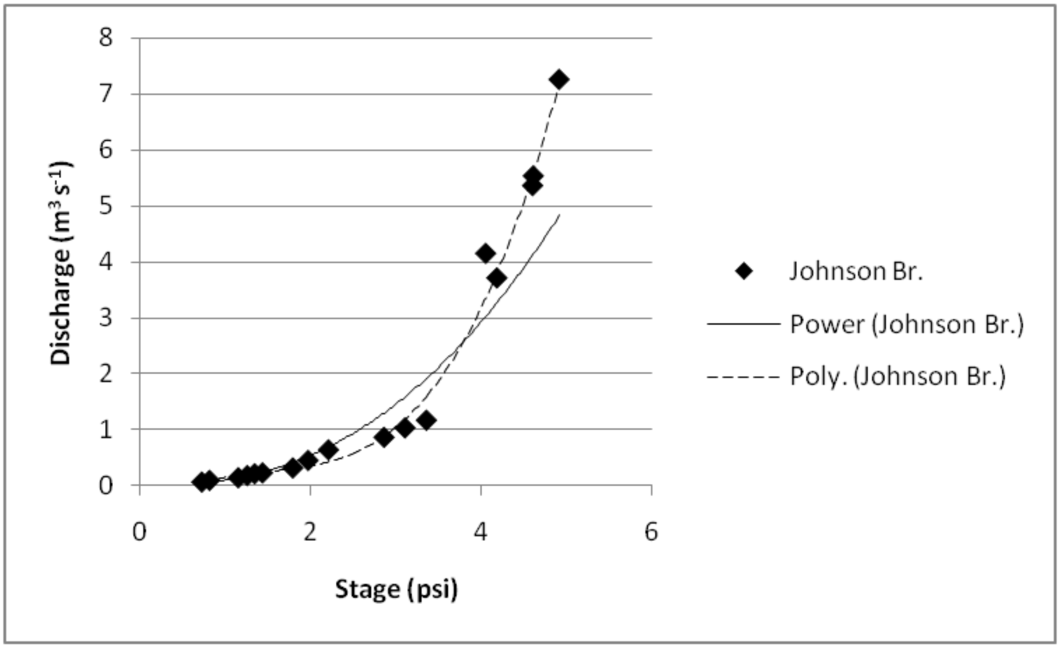

Figure A4. Fit of discharge rating curves in power and polynomial forms to all data at Johnson Bridge.

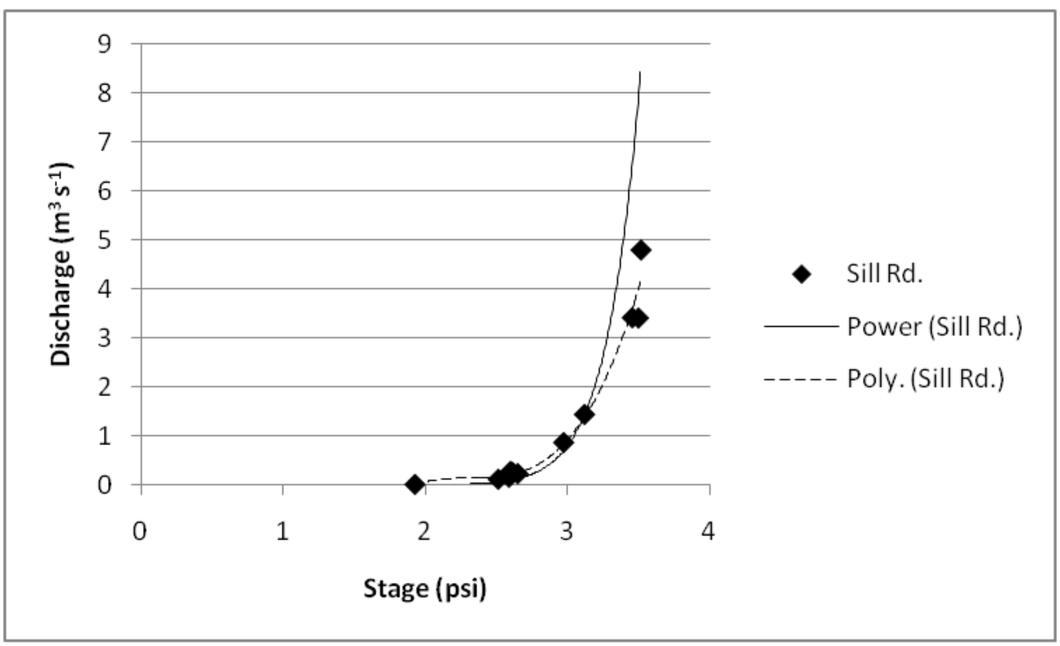

Figure A5. Fit of discharge rating curves in power and polynomial forms to all data at Sill Road.

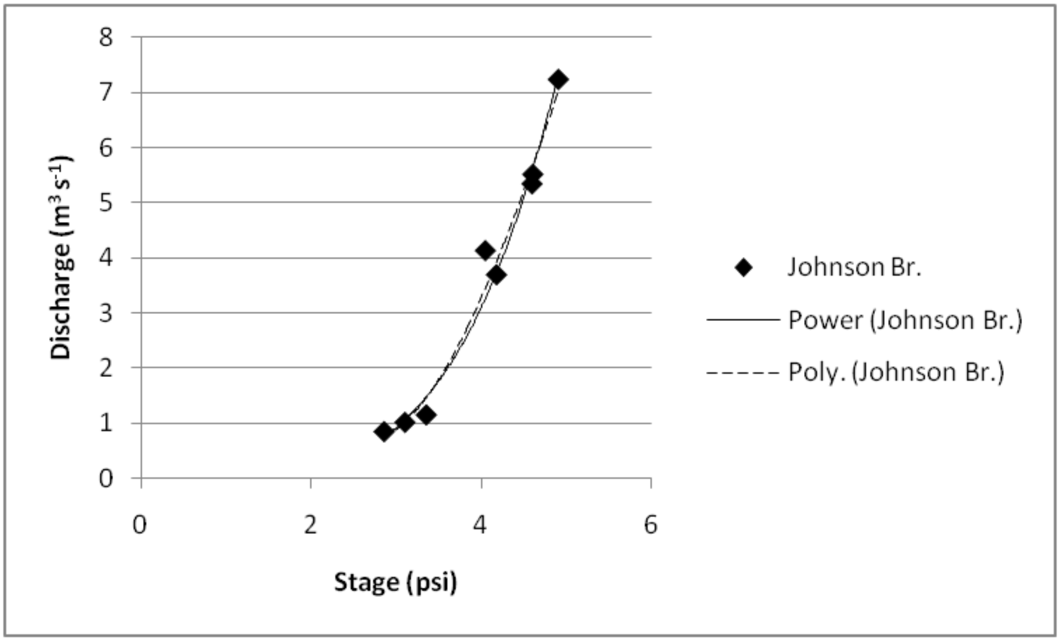

Figure A6. Fit of discharge rating curves in power and polynomial forms to data at Johnson Bridge, Z ≥ 2.861 psi.

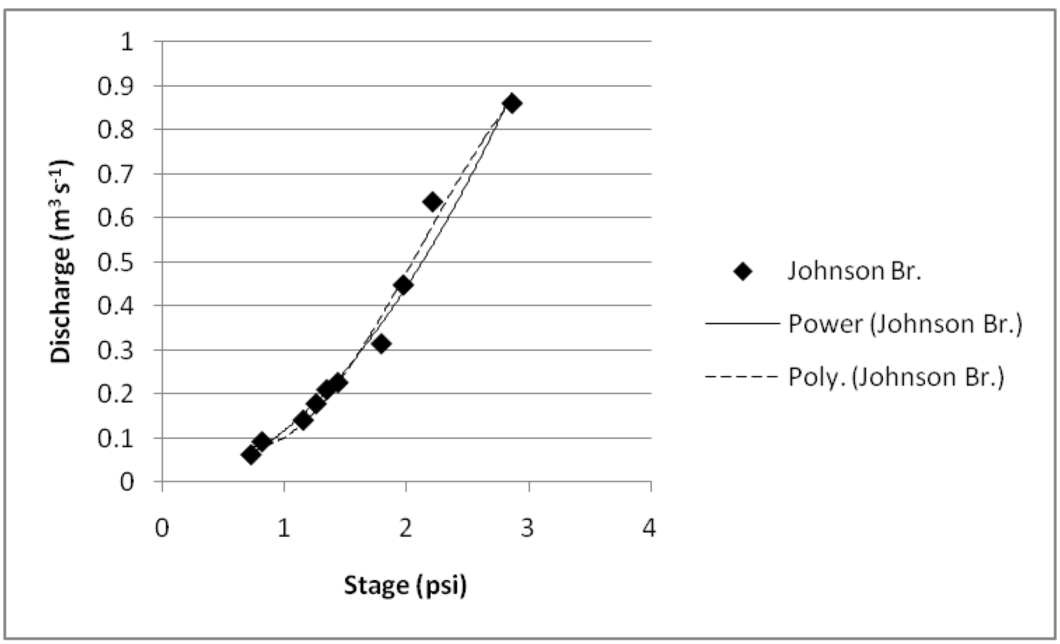

Figure A7. Fit of discharge rating curves in power and polynomial forms to data at Johnson Bridge Z ≤ 2.861 psi.

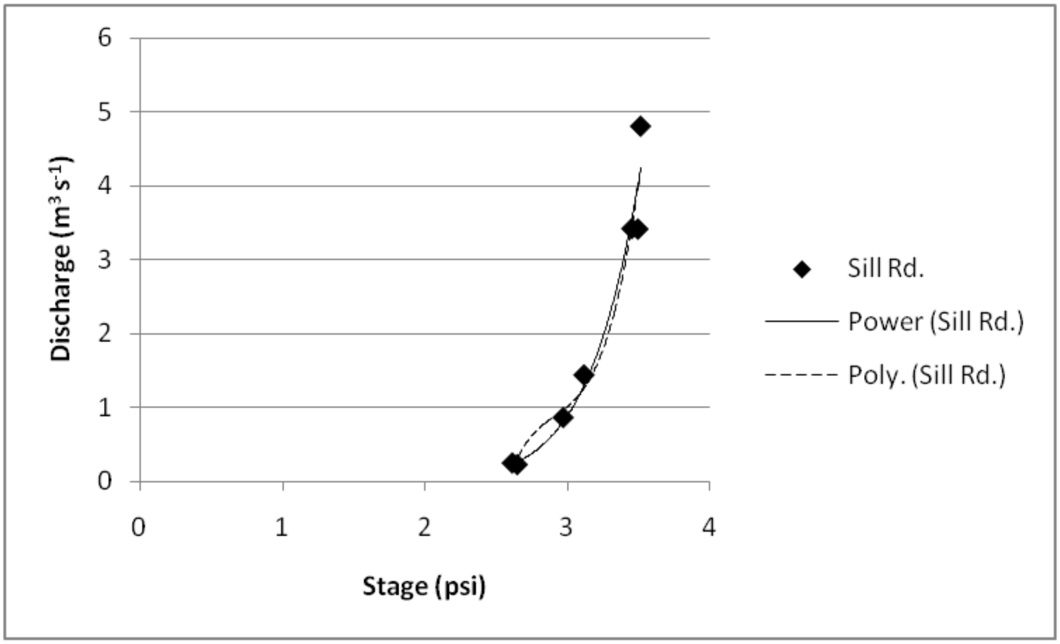

Figure A8. Fit of discharge rating curves in power and polynomial forms to data at Sill Road Z ≥ 2.6105 psi.

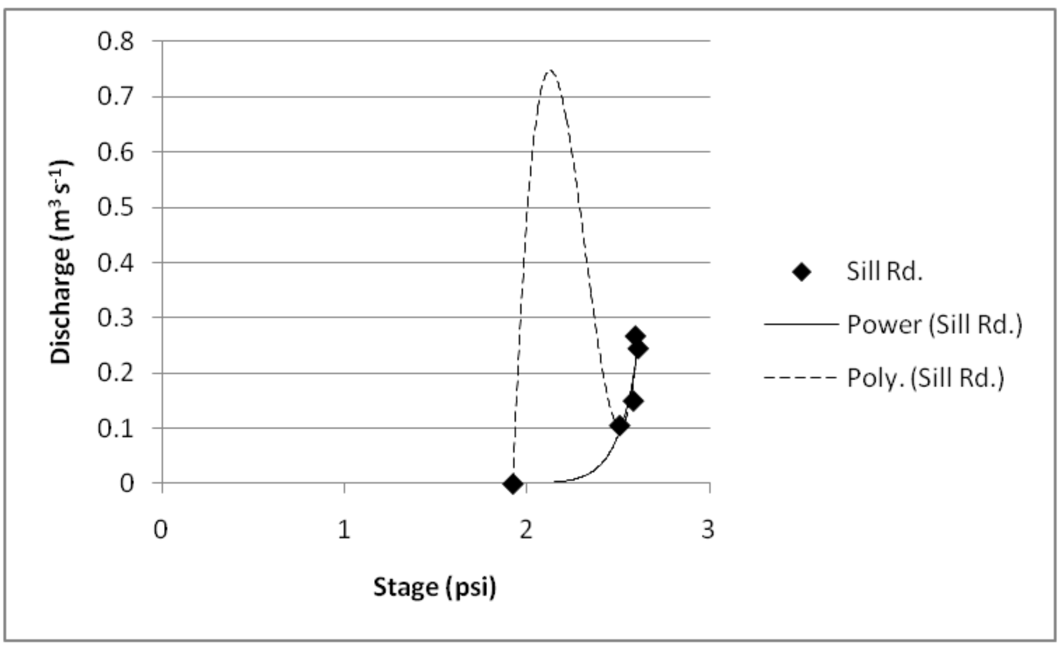

Figure A9. Fit of discharge rating curves in power and polynomial forms to data at Sill Road Z ≤ 2.6105 psi.

APPENDIX E

BEDLOAD PARTICLE SIZE DISTRIBUTION AT JOHNSON BRIDGE

Table A5. Bedload size distribution at Johnson Bridge for all samples.

Particle Size (mm)	Mass (g)	% of all samples	Cumulative %
0	52.4	0.58%	0.58%
0.125	274.127	3.04%	3.63%
0.25	2145.176	23.82%	27.44%
0.355	3959.163	43.96%	71.40%
0.5	1956.44	21.72%	93.13%
1	433.172	4.81%	97.94%
2	150.231	1.67%	99.61%
4	25.096	0.28%	99.88%
6.3	5.73	0.06%	99.94%
8	4.72	0.05%	99.95%
25	0	0.00%	100.00%