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Examination of the Relative Influence of Vegetation, Distance from Inflow, and Elevation on Sedimentation in a Coastal Californian Wetland

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EXAMINATION OF THE RELATIVE INFLUENCE OF VEGETATION,

DISTANCE FROM INFLOW, AND ELEVATION ON SEDIMENTATION

IN A COASTAL CALIFORNIAN WETLAND

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Master of Science

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Ryan M. Bassett

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Thesis of Ryan M. Bassett:

EXAMINATION OF THE RELATIVE INFLUENCE OF VEGETATION,

DISTANCE FROM INFLOW, AND ELEVATION ON SEDIMENTATION

IN A COASTAL CALIFORNIAN WETLAND

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ABSTRACT

Examination of the Relative Influence of Vegetation, Distance from Inflow, and Elevation on Sedimentation in a Coastal Californian Wetland

by
Ryan M. Bassett
Master of Science in Applied Marine and Watershed Science

California State University Monterey Bay, 2017

Wetlands and floodplains can act as areas of sediment deposition and storage. Therefore, they have the capability to improve downstream water quality and physical habitat. However, sedimentation rates may vary greatly within even a single wetland or floodplain. Much of the knowledge on wetland sedimentation rates is based on studies in controlled wetlands, where the setting and inflow may be carefully manipulated. While wetland systems receiving unregulated inflows are far more abundant, they are not as well studied. Determining which environmental factors drive deposition patterns may allow land managers to optimize sedimentation in managed wetlands. Additionally, quantified rates of sedimentation and land accretion have become important for managers considering the likelihood of habitat conversion, such as from freshwater wetlands to brackish or salt marsh, given climate change and subsequent sea level rise.

We evaluated the influence of vegetation type and density, elevation, and proximity to the point of inflow on sedimentation in a natural Californian wetland receiving unregulated inflows through model comparison and evidence ratios based on Akaike information criterion weights. In addition to generating an interpolated surface generated from 59 artificial grass mat sediment traps, we conducted a mass-balance sediment budget to act as an independent check of the total sedimentation in the wetland basin. Sedimentation values over the eight month study period ranged from 254.0 to 2875.2 g/m^2, with an average of 1054.6 g/m^2.We found strong evidence that distance from the point of inflow was the driving factor in depositional patterns, with vegetation also potentially playing a role. However, some of these postulated influences may have been confounded with each other; vegetation type and density were determined to be moderately correlated with distance from the point of inflow ($R = 0.273$ and $R = 0.325$, respectively). This limited our ability to conclude if vegetation was a driving influence on observed sedimentation patterns.

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CHAPTER 1

INTRODUCTION

Nonpoint source pollution has historically been the primary cause of water quality impairment in the United States, due primarily to the influx of excess nitrogen, phosphorus, and sediment from agricultural lands (Baker 1992, Puckett 1995), and remediation efforts have often proven to be difficult (Rissman and Carpenter 2015). Of these pollutants, excess sediment has been recognized as one of the most ubiquitous problems, where particles may increase turbidity, facilitate the transport of nutrients, pesticides, heavy metals, and pathogens, and settle in channels and waterbodies decreasing capacity (Brown and Froemke 2012). Wetlands and floodplains act as areas of sediment deposition and storage, and as such, have the capability to improve downstream water quality and physical habitat. Understanding how well, and at what rate, these environments are able to remove sediment is valuable for land managers who must decide how much wetland or floodplain acreage is required to maintain specific water quality standards.

Studies that quantify wetland functions have been conducted on artificial wetlands in many physical settings where variables such as flow, sediment load, and level of pollutants may be manipulated. Such studies have established that treatment wetlands remove excess nutrients and pollutants (Spieles and Mitsch 1999; Jordan et al. 2003; Howell et al. 2005; Keizer-Vlek et al. 2014; Miller 2014), sediment (Braskerud et al. 2000; Braskerud 2001; Harter and Mitsch 2003; Jordan et al. 2003; Ockenden et al. 2012; McAndrew et al. 2016), and provide habitat for wildlife (Knight 1997; Knight et al. 2001; Levy 2015). Although these studies have been able to quantify wetland functions in controlled conditions or in general terms, the results might not directly apply to natural, unregulated systems (Jordan et al. 2003). Much of the existing knowledge on sedimentation rates in uncontrolled systems comes from coastal marsh, delta, and riparian studies; sediment accumulation in freshwater wetlands is not as well understood. Sediment budgets are often useful tools in studying sinks, sources, and watershed dynamics. However, even when continually monitored, sediment budgets do not provide any information on the spatial variability of deposition, or if erosion also occurs within the study area. Knowledge about the spatial variability of sediment deposition is important for understanding sedimentation and storage rates, and for management decisions (Wasson 2002).

Sedimentation rates in natural wetland and floodplain systems have been well documented (Trimble 1999; Craft and Casey 2000; Steiger and Gurnell 2001; Callaway et al. 2012), as have the patterns of deposition (Lambert and Walling 1987; Asselman and Middlekoop 1995; Middelkoop and Asselman 1998; Dezzeo et al. 2000; Steiger et al. 2003; Krovang et al. 2007; Hupp et al. 2015). Similar studies have also been conducted on artificially constructed or treatment wetlands (Fennessy et al. 1994; Harter and Mitsch 2003; Jordan et al. 2003; Nahlik and Mitsch 2008; Mitsch et al. 2014).

In addition to quantifying the overall wetland or floodplain sediment trapping ability, some studies were able to gain some qualitative inference on the effect certain environmental factors have on sedimentation. Sedimentation rates tended to be highest near the point of inflow due to the proximity of the sediment source (Braskerud 2001; Harter and Mitsch 2003; Bannister 2015). Aquatic vegetation has been shown to increase sedimentation rates by reducing water velocity, thereby allowing particles to settle out of suspension, or by reducing the amount of sediment resuspended after settlement. However, Harter and Mitsch (2003) observed that open, deep water areas trapped more sediment than the shallower vegetation areas, indicating that other factors, or a complex interaction of factors, may be driving the observed results. It is apparent that sedimentation patterns, and potentially sedimentation rates, are driven in part by the environmental factors influencing water velocity at the wetland or floodplain site. Since the inferred influence of the environmental factors on sedimentation have been largely qualitative in previous studies, additional work quantitatively examining multiple factors would prove useful. An understanding of which conditions best facilitate sedimentation would prove valuable for land managers tasked with improving downstream water quality.

OBJECTIVES

We examined spatial patterns of sediment retention in an unregulated wetland to better understand the relative importance of local environmental factors on deposition. Our objective was to measure the impact of several environmental factors on the rate of sediment storage on an ephemeral wetland receiving unregulated inflows. The controlled environmental factors included: vegetation type, vegetation density, distance from inflow, and elevation. We measured the spatial variability of sediment retention during a single winter. The relative importance of the environmental factors (and combinations of factors) was estimated using Akaike's Information Criterion (AIC) model selection on a set of sixteen generalized linear models.

CHAPTER 2

METHODS

STUDY SITE

Carneros Creek is a major freshwater tributary flowing into the Elkhorn Slough - a National Estuarine Research Reserve, located north of Monterey, CA (Fig.1). Although there is evidence that the creek was once a meandering coastal stream, Carneros Creek was straightened and channelized prior to 1917 to maximize available agricultural space and provide some degree of flood protection (Largay 2007). The Agriculture and Land Based Training Association (ALBA) owns and operates Triple M Ranch, which is located on Carneros Creek. This ranch consists of 78 hectares of farmland with 54 hectares left uncultivated as natural and ruderal land. Soils on site are predominantly Aquic Xerofluvents created through channel and floodplain processes and Clear Lake Clay associated with still water wetlands (Los Huertos and Shennan 2002; Largay 2007). Of the 83.5 km^2 Carneros Creek watershed, 71.3 km^2 drains into the study location. During a high flow event in 1998, sediment blocked the canal and forced the channel to avulse into the adjacent agricultural fields. Instead of repairing the dredge-spoil levees and rechannelizing the stream, land managers allowed the flooded fields to naturally revert back to seasonal wetlands.

This study focused on two wetlands located downstream of the channel avulsion within Triple M Ranch (Fig. 2). The northern of the two wetlands is approximately 2.01 hectares, whereas the southern wetland is approximately 3.34 hectares. During our study period, flow first entered the southern wetland, after which it immediately entered the northern wetland. An unmeasured fraction of streamflow bypassed the southern wetland and directly entered the northern wetland. No flow exited the study site without first passing through at least one of the wetlands.

We gaged streamflow both upstream and downstream of the study wetlands using Levelogger pressure transducers (Solinst Canada, Ltd.). Physical limitations related to channel morphology made it impractical to rate a stream gage immediately upstream of the studied wetlands. Therefore, the flow entering the wetlands was adjusted to account for the additional watershed area between the upstream and downstream gaging sites. The downstream gaging site was immediately downstream of the northern wetland. The elevations of the upstream and downstream gaging locations were 5 m and 2 m (NGVD88), respectively.

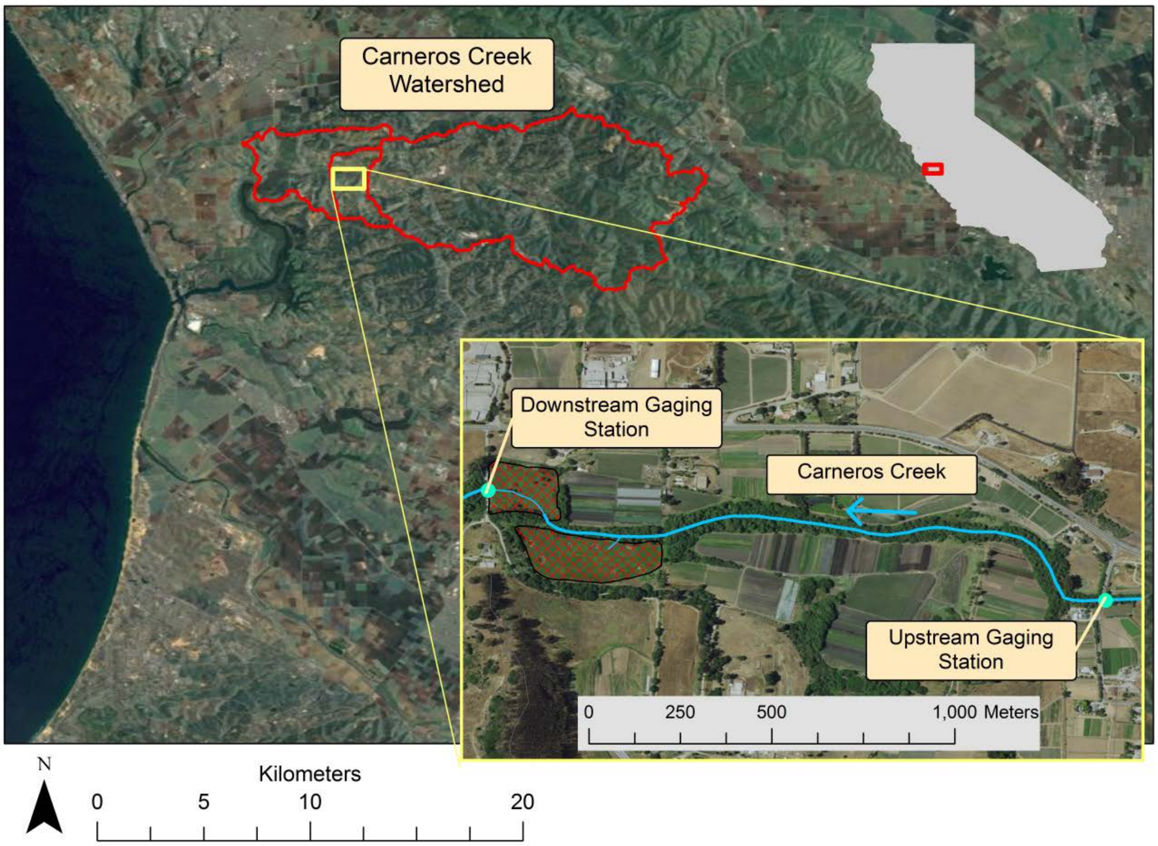

Figure 1. Map of the study location, which is south of the city of Watsonville, in Santa Cruz County, California.

Field Methods

Prior to the first runoff producing rain event in water year (WY) 2012, 59 sample plots consisting of one 30 cm × 30 cm artificial grass mat were installed and placed using a 30 m gridded sampling design with incorporated randomization. This sampling methodology was used to ensure each environmental parameter was sufficiently replicated prior to installation, and the sample sites had a good spatial coverage of the wetlands (Fig. 2). However, three of the 59 sediment traps were not contacted by any sediment laden water as they were placed at locations above the level of inundation.

The ground surface at each sample location was manually exposed by removing vegetation prior to the installation of the sediment trapping mat. Each sediment trap was secured in place using four 18 cm steel pins, with the location surveyed and flagged.

The following parameters were measured at each sediment trap location: (1) position, (2) elevation, (3) mass of vegetation, and (4) predominant type of vegetation (Table 1). Plot position and elevation were measured with a Nikon NPR-362 total station, with the prism pole placed directly on the installed mat. The resulting coordinates were used to measure linear distance to the point of inflow.

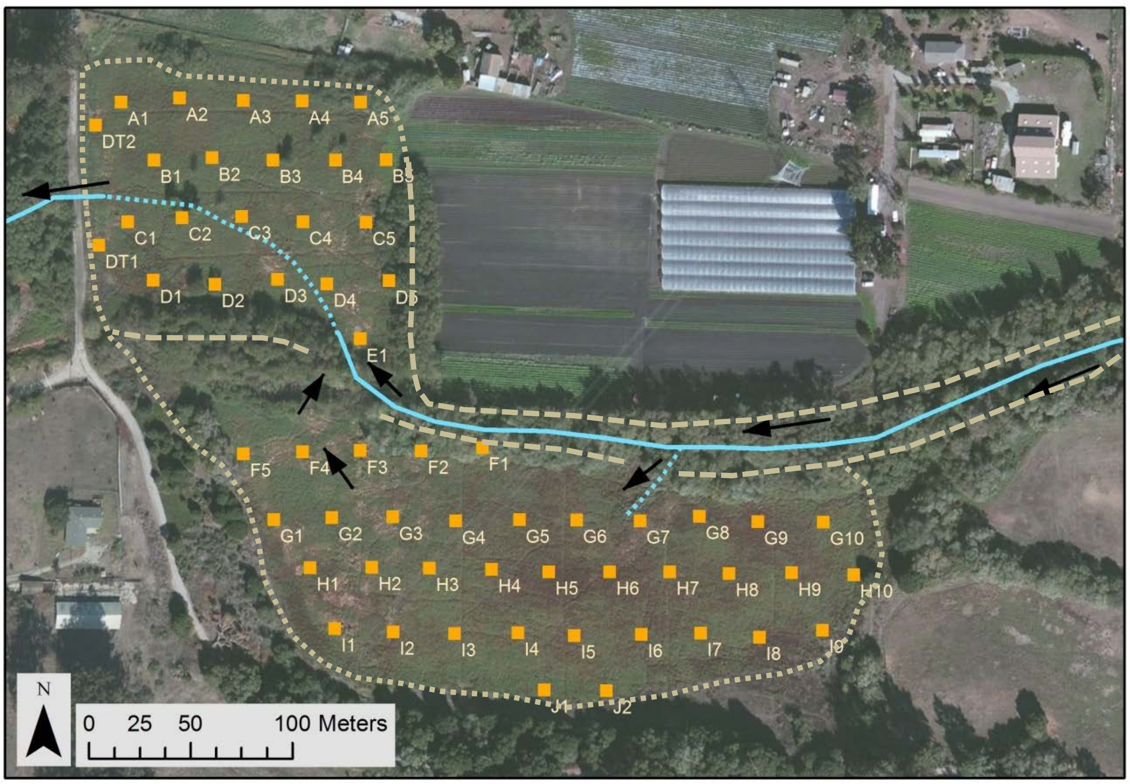

Figure 2. Aerial view of the study site showing the sediment trap locations, with arrows indicating flow directionality. Note the approximate locations of dredge-spoil levees (dashed lines) and the boundary of the wetland basin (dotted line).

Table 1. The parameters measured at each sediment trap plot location.

Parameter	Type of Data	Measured by	Used to Determine
Position	X, Y Coordinate	Field survey	Distance from inflow
Elevation	Continuous Variable (m)	Field survey	Index of relative average water depth
Vegetation Type	Categorical Variable	Field observation	Type of vegetation
Vegetation Density	Continuous Variable (kg m^{-2})	Field measurement	Density of stalk and leaves

For the purposes of this study, vegetation density was defined as the dry mass of plant material per square meter (kg/m^2). We determined the vegetation density by placing a 25 cm × 25 cm quadrat 0.5 m from the sample plot, and removing all above-ground vegetation. Vegetation stalks and leaves were cleared of any clumps of sediment and dried in a laboratory oven. In the winter months, the dominant plant species of the study site, *Polygonum spp*., died back and became dormant. For this reason, the density measurement had to be taken once the vegetation has become dormant, to represent the sediment trapping impact during runoff events. The homogenous distribution of plant material in the wetlands allowed an estimate of plant density at the mat site to be made from a sample near the mats, thereby avoiding direct disturbance on the mat site.

Flow first entered the southern wetland through a well-defined avulsion channel (Fig. 2). The distance from each sample plot in the southern wetland to this inflow point was measured (Table 1). For the northern wetland, the downstream wetland, the point of inflow received a combination of waters that passed through the southern wetland and flows that bypassed the first wetland in the creek channel. The point at which the combined flow enters the northern wetland was considered the point of inflow for this sub-basin. The two sediment traps installed in the ditch immediately downstream of the downstream gage control point were excluded from the modeling portion of the study as they were not representative of majority of the wetland conditions. However, since they provide more detailed information on the total amount of sediment trapped in the system, they were included the net deposition calculation.

Sediment load was measured at locations upstream and downstream of the wetland system. The volume of suspended sediment entering and exiting the study site was measured by collecting water samples at both locations using a DH-48 sampler on a regular basis, with more frequent samples taken during post-storm run-off events. Bedload sediment was also collected at the upstream gaging station using a Helley-Smith sediment sampler (Guy and Norman 1970; CCoWS 2004). Backwater in the channel upstream of the wetlands caused nearly all bedload (sand and small gravel) to be deposited before flow entered the study site, and none to exit the study site (Largay 2007; Holloway 2010).

Pressure gages and staff plates were deployed above and below the wetland system. The upstream gage was located at the nearest location suitable for gaging, which is approximately 1200 m upstream. The downstream gage was placed at the exit of the northern wetland, to capture the total flow exiting the study site. These gages were installed prior to WY 2009 and were maintained throughout this study.

Using standard procedures, we established a stage-discharge relationship by measuring stream discharges at a range of flows and applying this function to logged stage data. The transport of sediment through time was determined in a similar fashion, where the suspended and bedload rating equations were applied to the continuous stage record.

All sediment traps were collected in early summer 2012 once the ground became sufficiently dry. Any sediment clinging to the bottom of the mat was scraped off in the field before the mat is placed in an individual 13 liter bucket. All sediment-laden organic matter was kept on the mat for later processing.

LABORATORY METHODS

We measured the suspended sediment concentration (SSC) of the collected samples where each sample was filtered through coarse and fine glass fiber disks and weighed using standard laboratory techniques (Guy 1969; IAEA 2005). The dry mass of trapped sediment on the artificial grass mats was determined through methods outlined by Steiger et al. (2001). This process included: manually washing mats until clean, retaining the sediment laden water in a bucket and allowing it to settle for 2-3 weeks, siphoning off the clear water, drying the slurry in a laboratory oven, weighing to determine the combined dry mass of trapped organics and sediment, combusting the organic portion in a muffle furnace, and reweighing to determine the total mass of organics-free sediment.

We determined that the mass of vegetation-trapped sediment was orders of magnitude less than the amount collected from the sediment mats, and therefore omitted further measurements of vegetation-trapped sediment. This determination was made following an elaborate laboratory process involving washing and combusting several samples of vegetation in order to quantify the amount of sediment trapped on the vegetation stalks.

DATA ANALYSES

Mass Balance Sediment Budget

A mass-balance sediment budget was used to determine the total amount of new sediment storage on the Triple M Ranch during the study period. The overall sediment budget for the study site may be written as:

$$\Delta S = S_{in} - S_{out}$$

where, ΔS is the change in sediment storage (kg/yr), S_{in} is the total mass of sediment that entered the study site (kg/yr),and S_{out} is the total mass of sediment that exited the wetland system (kg/yr). Since both bedload and suspended sediment entered the study site through the main channel as well as small tributaries, the following expanded equation more accurately describes the change in on-site sediment storage:

$$\Delta S = (S_{s-in} + S_{b-in} + S_{trib-in}) - (S_{s-out} + S_{b-out})$$

where, S_{s-in} and S_{s-out} is suspended sediment entering and exiting the study site, S_{b-in} and S_{b-out} is bedload sediment entering and exiting the site, and $S_{trib-in}$ is the suspended sediment entering the site through tributaries joining Carneros Creek between the upstream and downstream gaging sites. Because the tributaries were small, non-sandy channels, we assumed that they transported only suspended sediment. Given the difference in watershed area between the upstream and downstream sites was 5.8%, the amount of suspended sediment supplied was determined by the following equation as none of the tributaries were directly measured (Largay 2007, Holloway 2010):

$$S_{trib-in} = 0.058 \times S_{s-in}$$

Sediment Rating Equations

Inflow and outflow of suspended and bedload sediment were measured at the upstream and downstream gaging locations from WY 2010 through WY 2012 using an event-based sampling strategy. Instantaneous measurements of bedload transport and suspended sediment concentration (SSC) were regressed against corresponding streamflow values to create rating equations for both classifications of sediment at the entire range of stream flows at both sites. The resulting exponential functions for the rating equations were:

$$S_{s-in} = 263.91\ {Q_{in}}^{1.69}$$

$$S_{s-out} = 94.51\ {Q_{out}}^{1.69}$$

$$S_{b-in} = 19.62\ {Q_{in}}^{1.35}$$

where Q_{in} refers to flow entering the study location, and Q_{out} refers to flow exiting the site. These rating equations were applied to the continuous flow record, yielding a continuous record of sediment transport at both gaging sites (Watson et al. 2005). To avoid biasing the curve to the more frequently observed lower values, the data were binned in 0.1 m^3/s sections from zero to 1.0 m^3/s, after which the values in each bin were averaged. Monte Carlo bootstrapping was used for suspended and bedload rating equations at both sites in order to remove biasing influence of any singular point of data, as well as provide a measurement error (Efron 1982; Wu 1986) Bootstrapping of the rating equations was conducted using the R statistical package (R Development Core Team, 2010).

Total Sediment Deposition over the Wetlands

We attempted to validate the budget-based approach to estimating the mass of sediment trapped by extrapolating the sediment trapped on the mats to the whole wetland. The total amount of sediment trapped per square meter of wetland may be written as:

$$S_{trap} = S_{gr} + T_{veg}D_{veg}$$

where S_{trap} (kg/m^2) is the total amount of sediment trapped, S_{gr} (kg/m^2) is the sediment deposited on the upper surface of the mat, and T_{veg} (kg sediment/kg plant) is the sediment trapped on the leaves and stalks of the vegetation, and D_{veg} is the measured density of vegetation in the field (kg plant/m^2). We assumed that the sediment trapped on the artificial grass mats was representative of the sediment trapped on the ground surface throughout the wetland, and therefore yielded the S_{gr} value. As described above, early laboratory analysis of T_{veg} indicated that it was far too small to significantly impact the total estimated from the sediment-trapping mats alone. We therefore omitted the T_{veg} term from future calculations and modeling.

Using the S_{trap} values obtained from the sediment traps, we created a modeled depositional surface through kriging in ArcGIS. Kriging is a method for interpolating point data over space using weighted local averaging, and is an optimal approach for interpolating sedimentation point data (Oliver and Webster 1990). Additionally, kriging allows the interpolated variance to be estimated and plotted, thus providing some measure of uncertainty. The average value for the interpolated surface was calculated and multiplied by the depositional area, as determined through field surveys, to determine the total mass of sediment deposited over the wetlands.

Model Comparison

We compared a series of generalized linear models to estimate the relative influence of environmental factors on the response variable—area-normalized sediment deposition (g/m^2). These depositional values were obtained by normalizing each 30 cm × 30 cm artificial grass mat such that they were 1 square meter. The models utilized the four predictor variables introduced in Table 1: distance from the point of inflow, elevation, vegetation density, and vegetation type. Distance from point of inflow was defined as the straight-line distance between the sample plot and the nearest point on the delineated inflow polygon as measured using ArcGIS. We used the elevation of each plot to represent the depth of water above the sediment trap. This assumed a planar and level water surface. While this assumption is not completely accurate, the slope between the wetland point of inflow and spill point is quite low (<1 m) and approximates a level water surface. Vegetation density was calculated as the area-normalized dry mass of vegetation stalks and leaves located near the plot. Vegetation type described the dominant plant species at the sampling location, and consisted of either smartweed (*Polygonum spp.*) or reed (*Typha spp.*). Each input variable was described as a continuous positive value, with the exception of vegetation type, which was categorical. The models were constructed and analyzed using the R statistical package (R Development Core Team, 2010) using the following generalized linear model:

$$Y \sim Gamma(\alpha, \beta)$$

$$\beta = \frac{\mu}{\alpha}$$

$$\mu = \frac{1}{\beta_0 + \beta_1 X_1 + \cdots + \beta_n X_n}$$

where Y is the response variable, α is the shape parameter, μ is the scale parameter, β_0 is the model intercept, and $\beta_i \ldots \beta_n$ are the parameter coefficients corresponding to predictor variables $X_1 \ldots X_n$. We elected to use inverse link to avoid convergence problems, and because it is the default gamma distribution in R. The inverse link refers to the relationship between μ and the predictor variables. Due to this inverse relationship, the fitted coefficients (β), and therefore their inferred influences, are inverted such that a positive influence will display a negative coefficient, and a negative influence will display a positive coefficient. We elected to use a gamma distribution because it describes a response variable as only having positive continuous values, unlike a normal distribution, which allows negative values, which in turn are impossible when measuring deposition using a settling plate (Nelder and Wedderburn 1972; McCullagh and Nedler 1989; Halekoh and Hojsgaard 2007).

Sixteen models (Table 2) were compared using Akaike's Information Criterion (AIC), which ranks models according to a combination of accuracy and parsimony. AIC was used to determine which combination of parameters created the model that best predicts sediment deposition with the fewest number of parameters (Burnham and Anderson 2002). We decided *a priori* to compare all sixteen models representing all possible combinations of parameters, as well as to compare the relative importance (RI) and log evidence ratios (LER) for each predictor variable (*DIST*, *ELEV*, V_{Mass}, and V_{Type}).

Table 2. The sixteen models compared in this study. DIST refers to the distance from the point of inflow, ELEV refers to the elevation of the sediment trap, V_{Mass} refers to the mass of vegetation at the trap location, and V_{Type} refers to the type of vegetation at trap location (*Polygonum* or *Typha*).

Model	Parameters Included
m0	Null (constant value)
mD	DIST
mE	ELEV
mM	V_{Mass}
mT	V_{Type}
mDE	DIST + ELEV
mDM	DIST + V_{Mass}
mDT	DIST + V_{Type}
mEM	ELEV + V_{Mass}
mET	ELEV + V_{Type}
mMT	V_{Mass} + V_{Type}
mDEM	DIST + ELEV + V_{Mass}
mDET	DIST + ELEV + V_{Type}
mDMT	DIST + V_{Mass} + V_{Type}
mEMT	ELEV + V_{Mass} + V_{Type}
mDEMT	DIST + ELEV + V_{Mass} + V_{Type}

CHAPTER 3

RESULTS

HYDROLOGY

Water year 2012 produced below average rainfall throughout the region. There was 39.8 cm precipitation recorded on site (Fig. 3), which is below the median of 53 cm/yr as historically measured at a site located 7 km to the northwest (Largay 2007). Stream flow initiated at the upstream gaging location on 5 Oct 2011, and ceased for the season on 1 May 2012. However, the creek remained completely dry for a substantial portion of the water year in October through mid-January as well (Fig. 3). Five major flow events were measured by the study pressure transducers, with runoff volume totaling 836 ML. We report gaged streamflow volumes to the nearest thousand cubic meters, which may overestimate precision. The seasonal peak flow was approximately 2.3 m^3/s. In contrast, the downstream gaging location measured a total of 537,000 m^3 over four major flow events that began 21 January 2012 and ceased 27 April 2012, with a peak of 2.5 m^3/s. Losses between the gages resulted from some combination of evapotranspiration, infiltration to groundwater, and slow leakage through the sand deposited and occluding the canal downstream of the study site.

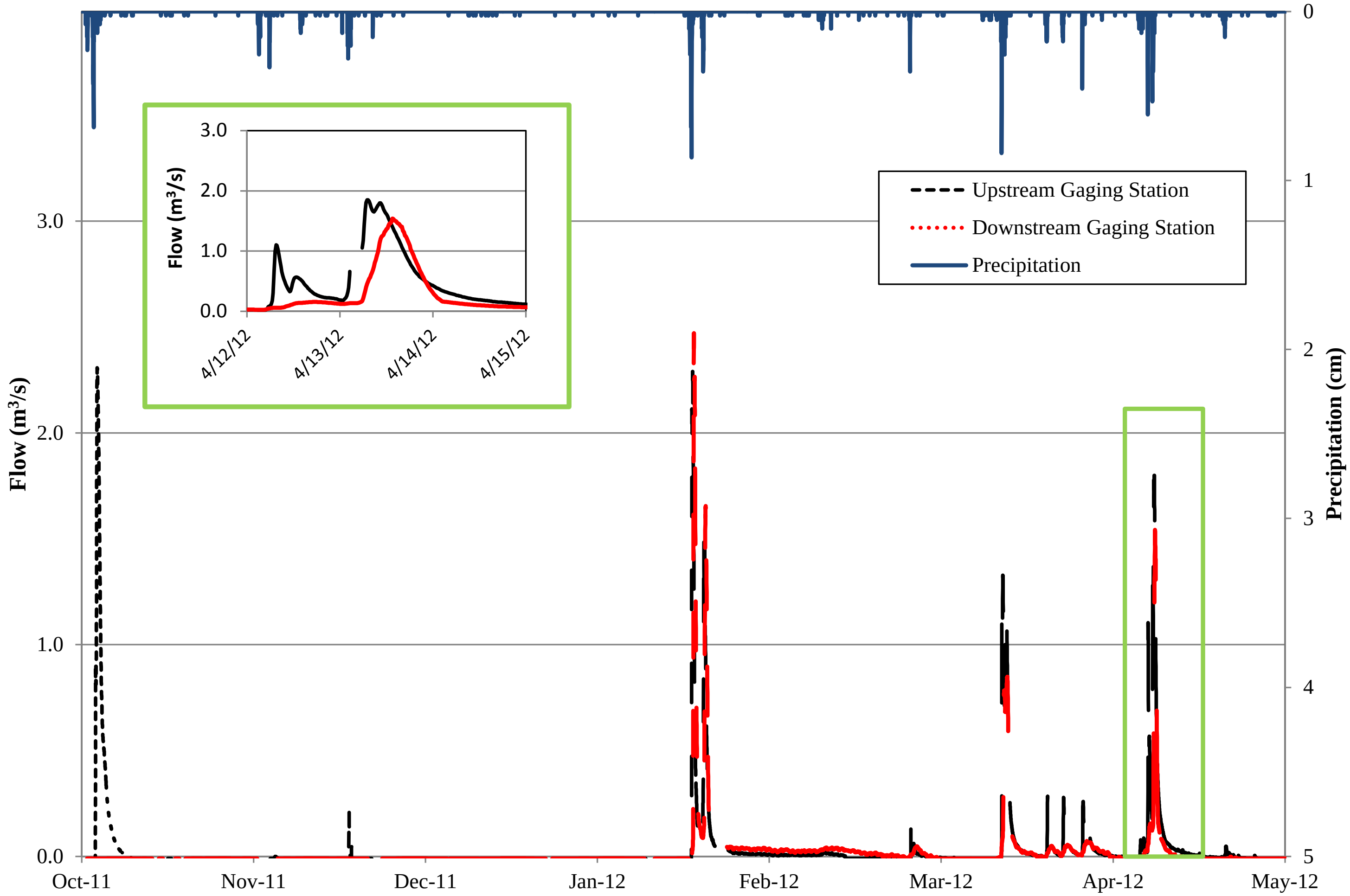

Figure 3. Hydrographs for the upstream and downstream gages. Note the downstream gage did not receive runoff from the first storm due to the insufficient volume of water required to inundate the study wetlands. An April storm event is highlighted in the inset box, illustrating the lag between peaks and the lack of downstream flow as the wetland basin became inundated.

SEDIMENT BUDGET

Holloway (2010) determined that there was great uncertainty in calculations of sediment mass transport using standard stream gaging techniques. There was less absolute uncertainty in the current study because there was less runoff and lower peak discharge values, making field measurements easier to obtain. We report gaged sediment transport values to the nearest 1 tonne, which may overstate the true precision of the values.

A total of 204 tonnes of suspended sediment, and 14 tonnes of bedload sediment entered through the upstream gaging location. At the downstream end, 38 tonnes of suspended sediment, and no bedload sediment exited the study site after passing through the wetland system. Accounting for the additional suspended sediment supplied by the tributaries located between the gaging sites (12 tonnes), 83% of the suspended sediment and 100% of bedload sediment was retained in the study site. Table 3 summarizes the sediment budget, and Figure 4 presents the sediment data and rating curves.

Table 3. A summary of the suspended sediment, bedload sediment, and flow entering and exiting the study location.

Flow - Upstream (IN)	836	ML
Flow - Side tribs (IN)	49	ML
Flow - Downstream (OUT)	537	ML
Flow – difference	348	ML
Suspended Sediment - Upstream (IN)	204	tonnes
Suspended Sediment - Side tribs (IN)	12	tonnes
Suspended Sediment - Downstream (OUT)	38	tonnes
Suspended Sediment – difference	178	tonnes
Bedload Sediment - Upstream (IN)	14	tonnes
Bedload Sediment - Downstream (OUT)	0	tonnes
Bedload Sediment – difference	14	tonnes
Total Sediment Trapped	192	tonnes

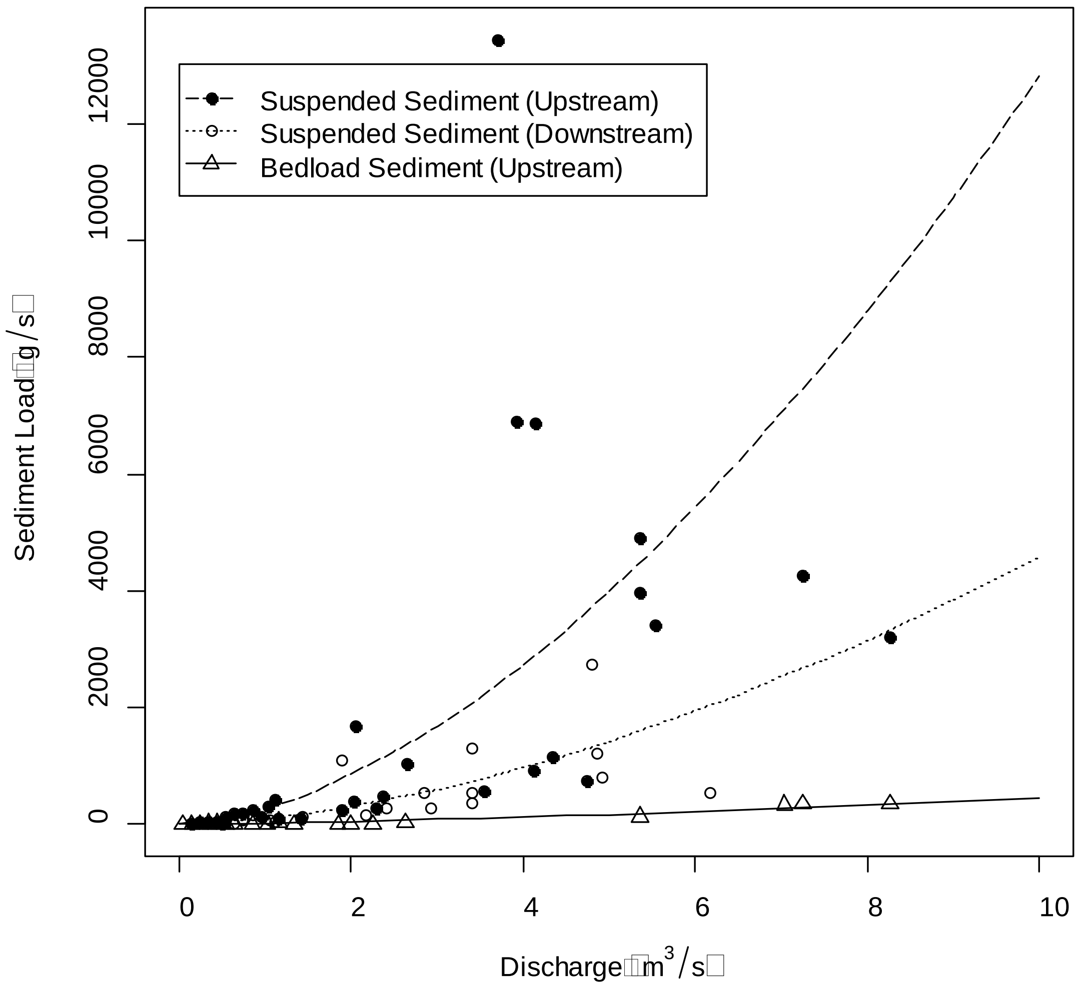

Figure 4. Measured sediment load and discharge at upstream and downstream sites, and fitted curves describing the relationship between the two.

SEDIMENT TRAPS

Of the 59 sediment traps installed, 56 were recovered and cleaned of sediment for analysis. Three of these collected mats were placed slightly above the level of inundation reached during the study; these mats were excluded from further analysis. Trapped sediment ranged from 22.9 g to 258.8 g, with an average of 94.9 g, equating to a coverage of 254.0, 2875.2, and 1054.6 g/m^2, respectively (Fig. 5). The mass of the trapped organic component ranged from 3.0 g to 32.4 g, with a mean of 13.2 g. On average, the sediment traps collected 87% sediment and 13% organics by mass.

As determined through GIS surface interpolation (Fig. 6), the southern wetland retained a total of 35 tonnes of sediment, with an average of 1050 g/m^2, a minimum of 230 g/m^2, and a maximum of 1812 g/m^2. The northern wetland retained a total of 17 tonnes of sediment, with an average of 851 g/m^2, a minimum of 332 g/m^2, and a maximum of 2875 g/m^2. Both wetlands combined trapped 52 tonnes of sediment, as determined through the interpolation of the sediment trap data (Fig. 6). Vegetation density ranged from 0.19 to 7.09 kg/m^2, and averaged 3.35 kg/m^2. *Polygonum* was the dominant vegetation type at 43 sample locations, whereas *Typha* was the dominant species at six locations (Fig. 7). Mat elevation ranged from 1.378 - 2.828 m, with an average elevation of 2.373 m (Fig. 8). The sediment traps ranged from 18.4 to 193.1 meters away from the point of inflow (Figs. 9 and 10). Figures 9 and 10 illustrate how well each variable predicted sediment trapping.

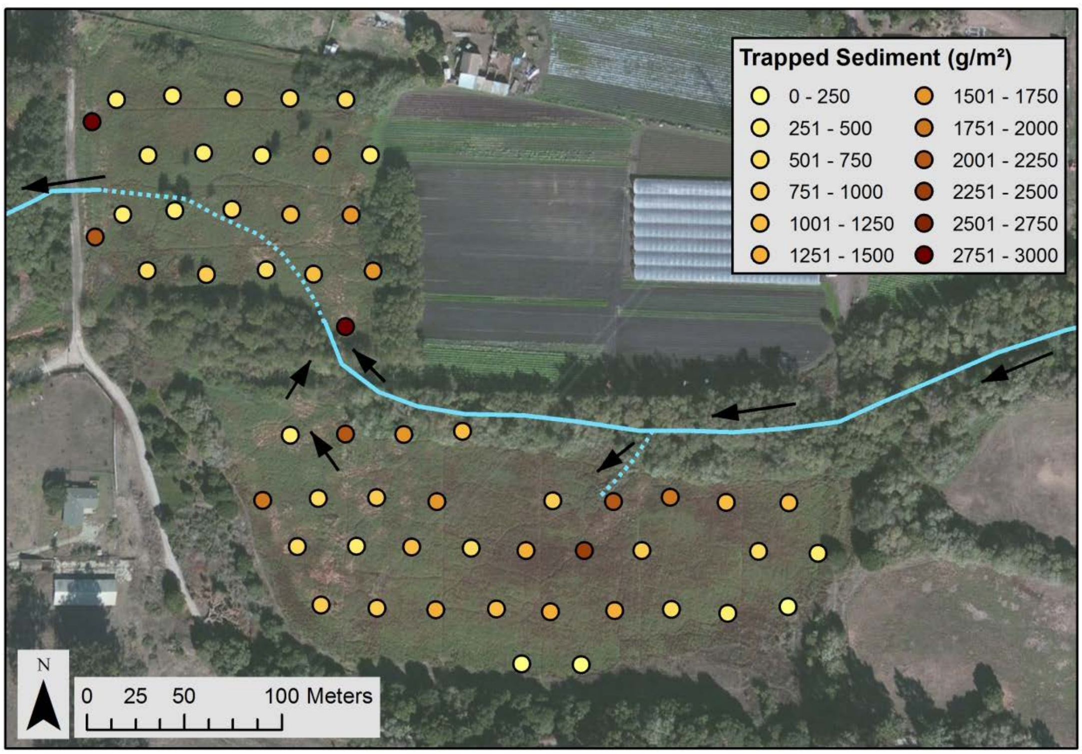

Figure 5. Mass of trapped sediment (g/m^2) at each of the recovered sample plots.

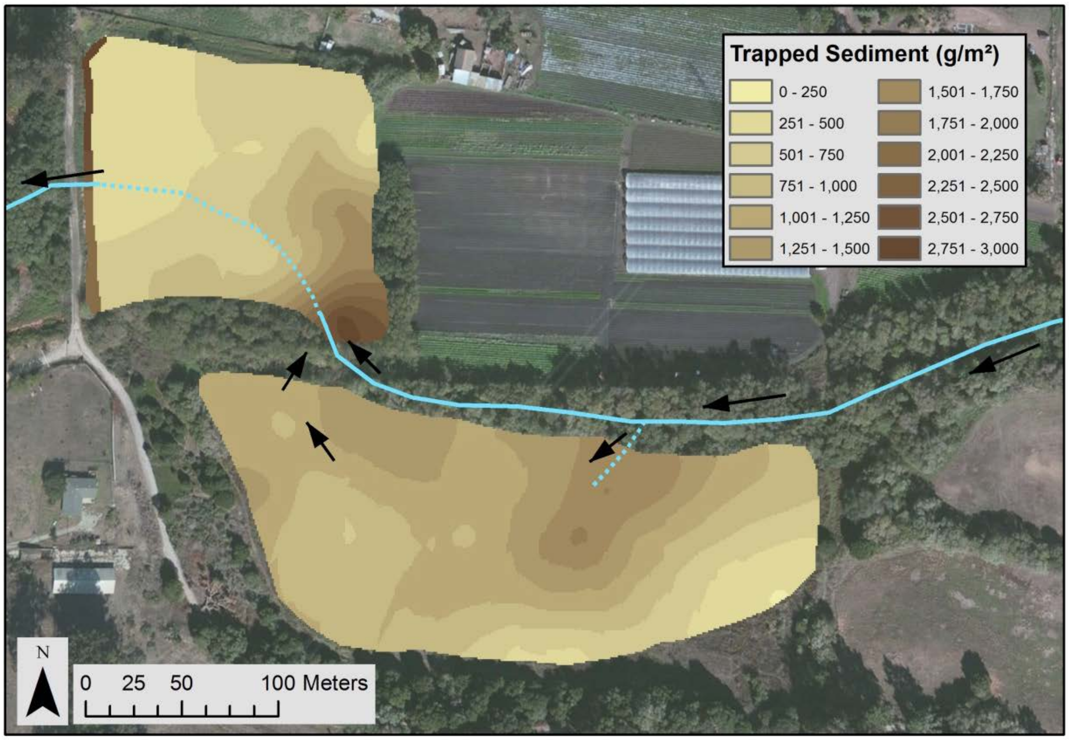

Figure 6. Interpolated surface of trapped sediment generated through kriging the sediment trap data.

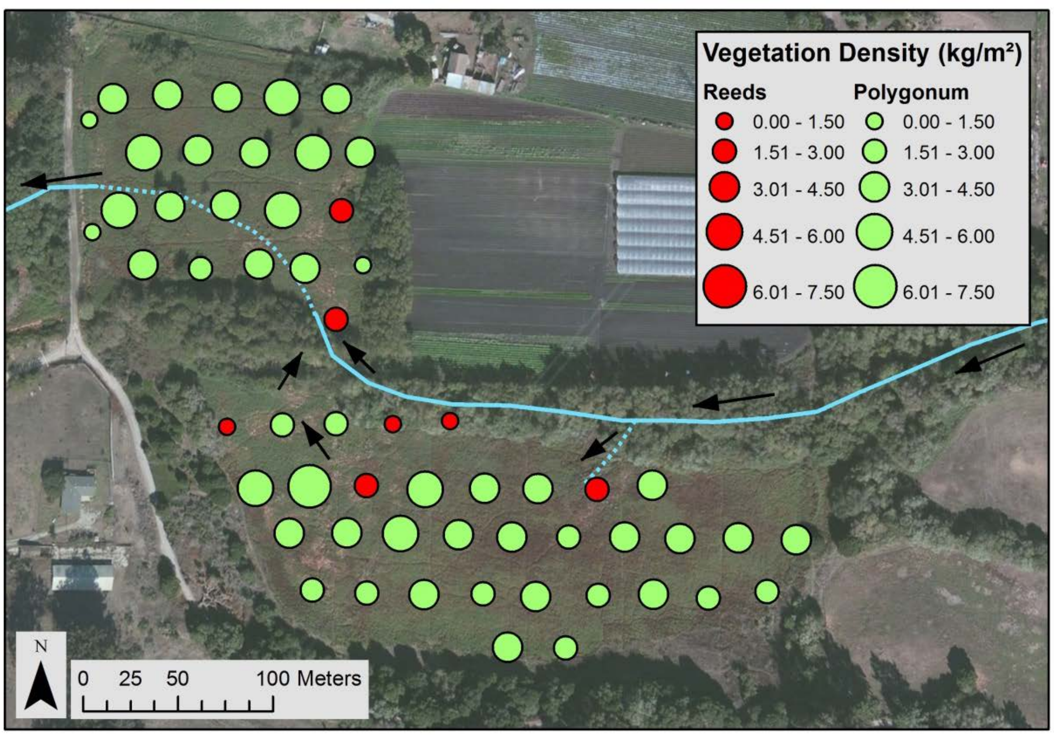

Figure 7. Vegetation type and density at each of the recovered plots.

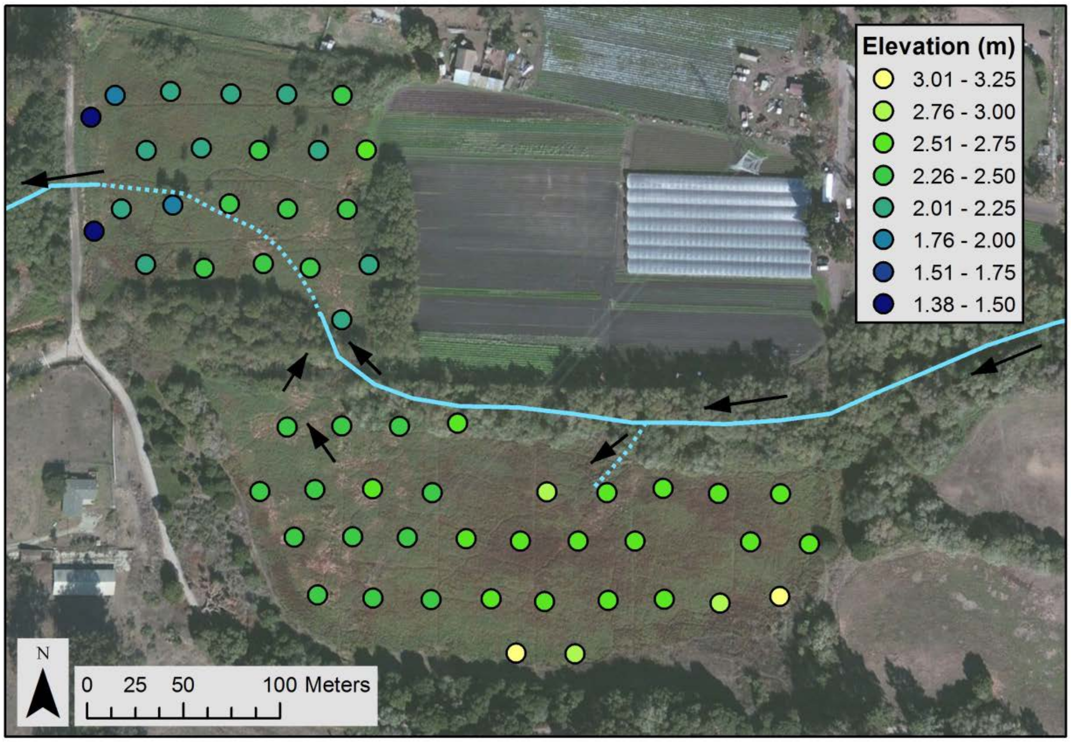

Figure 8. Elevation of each of the recovered sediment traps.

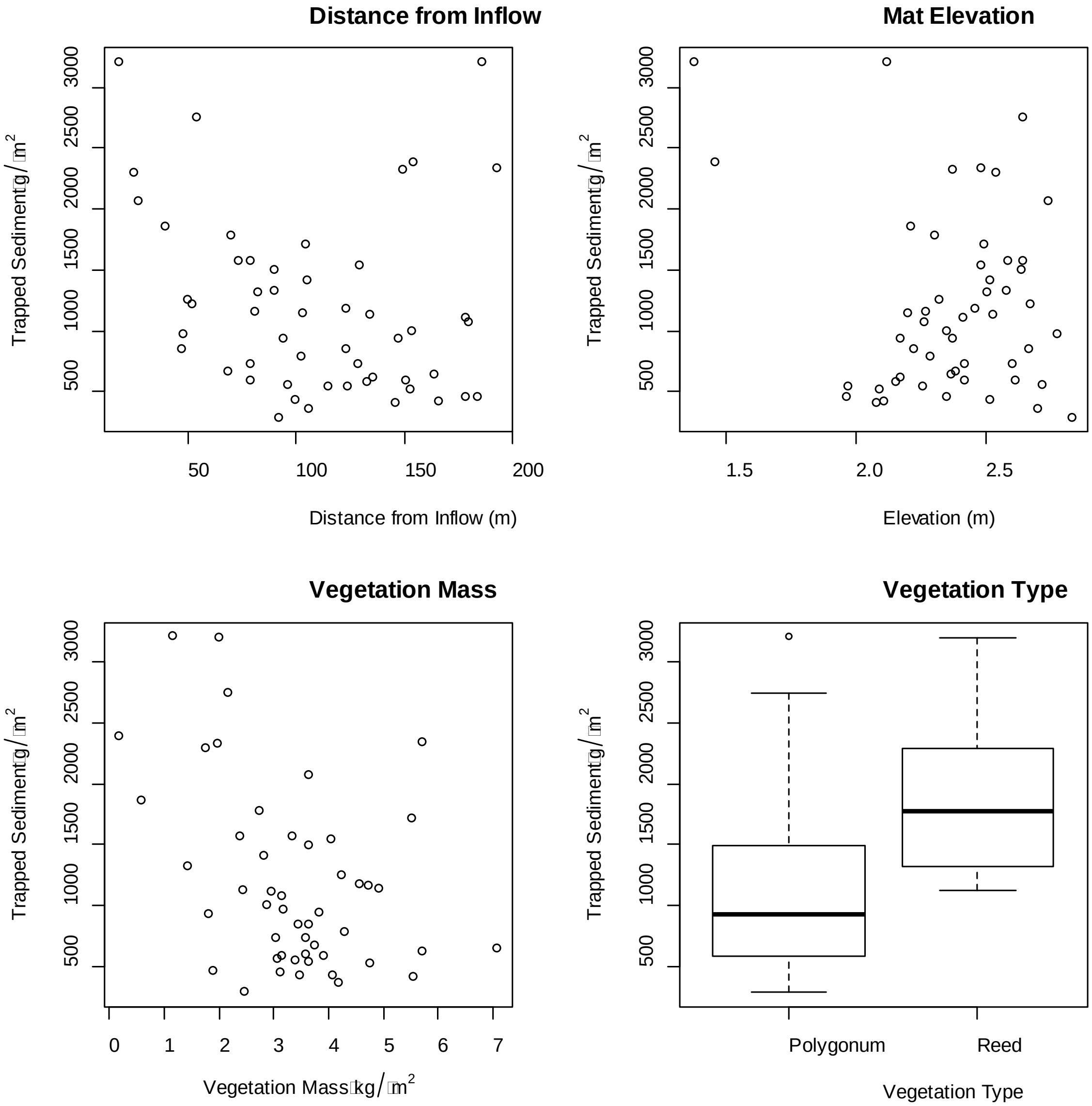

Figure 9. Plots displaying the distance from the point of inflow, sediment trap elevation, vegetation mass, and a box plot of the predominant type of vegetation at each sample location.

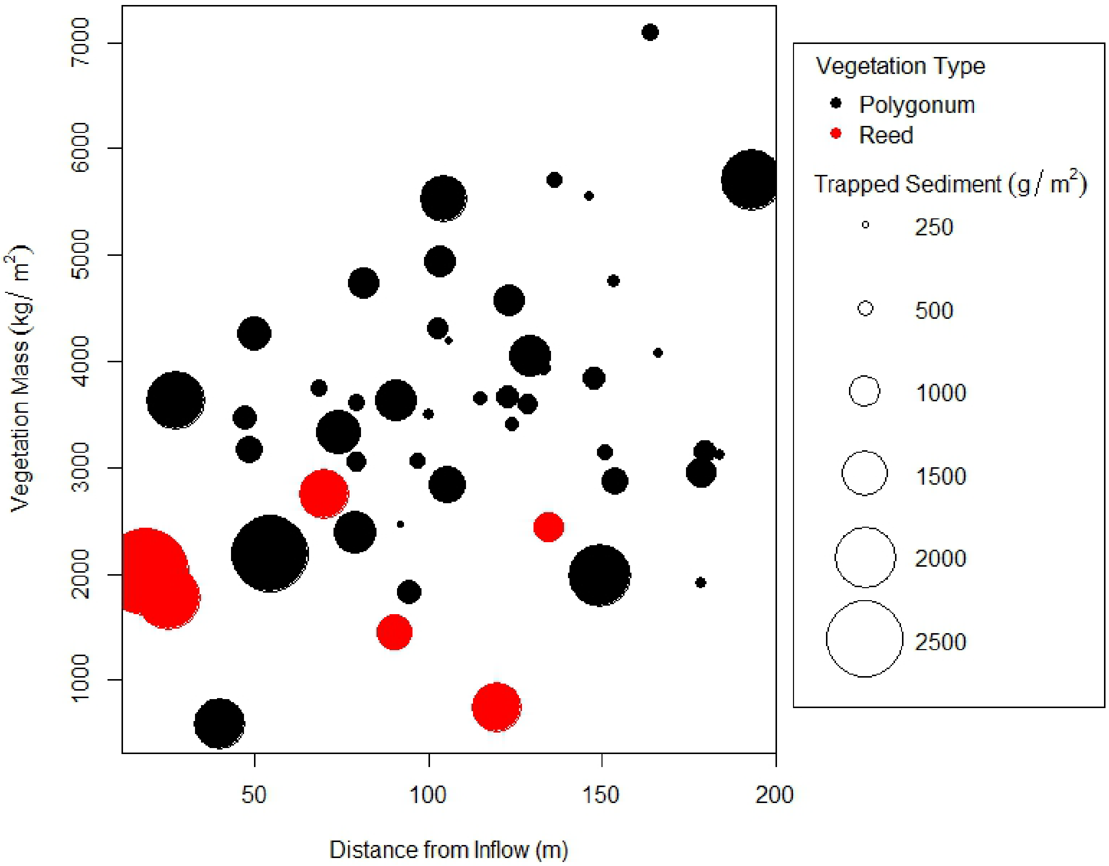

Figure 10. The distance from inflow plotted against vegetation mass where amount of trapped sediment is represented by dot size. Color denotes vegetation type.

MODEL COMPARISON

There was strong evidence that sediment trapping was related to distance, some evidence that it was also related to vegetation type, and also some evidence that it was not related to vegetation mass or elevation (Tables 4 and 5). However, there is some uncertainty regarding the influence of vegetation on sediment trapping due to a correlation between vegetation and distance from the point of inflow (see below). The model incorporating distance from inflow and vegetation type (mDT) performed the best in the AIC comparison, with models mDM (distance and vegetation mass), mD (distance), mDMT (distance, vegetation mass, and vegetation type), and mDET (distance, elevation, and vegetation type) also receiving some support. Only model mE (elevation) performed worse than the null model (Table 4).

Table 4. AIC table displaying degrees of freedom (ν), AIC value, sample size corrected AIC value (AIC_c), difference from best model (ΔAIC), weighted AIC (AIC_w), model shape parameter (α), and model coefficients ($\beta_0 - \beta_i$). Note, the signs of the model coefficients are inverted such that a negative value represents a positive relationship, and a vise versa.

Model	Covariates Included	ν	AIC	AIC_c	ΔAIC	AIC_w	α	β_0	β_{DIST}	β_{VType}	β_{VMass}	β_{ELEV}
mDT	DIST, V_{Type}	4	743.1	744.0	0.0	0.28	3.896	6.3×10^{-4}	4.6×10^{-6}	-3.4×10^{-4}		
mDM	DIST, V_{Mass}	4	744.4	745.3	1.3	0.15	3.803	2.2×10^{-4}	4.9×10^{-6}		9.8×10^{-8}	
mD	DIST	3	744.9	745.4	1.4	0.14	3.630	4.2×10^{-4}	6.1×10^{-6}			
mDMT	DIST, V_{Mass}, V_{Type}	5	744.3	745.7	1.7	0.12	3.951	4.6×10^{-4}	4.3×10^{-6}	-2.6×10^{-4}	6.0×10^{-8}	
mDET	DIST, ELEV, V_{Type}	5	744.4	745.8	1.8	0.11	3.947	1.4×10^{-3}	4.3×10^{-6}	-3.7×10^{-4}		-2.8×10^{-4}
mDEM	DIST, ELEV, V_{Mass}	5	746.2	747.6	3.6	0.05	3.813	5.4×10^{-4}	4.8×10^{-6}		1.0×10^{-7}	-1.3×10^{-4}
mDE	DIST, ELEV	4	746.8	747.7	3.7	0.04	3.634	6.2×10^{-4}	6.0×10^{-6}			-7.7×10^{-5}
mDEMT	DIST, ELEV, V_{Mass}, V_{Type}	6	745.8	747.8	3.8	0.04	3.989	1.1×10^{-3}	4.1×10^{-6}	-2.9×10^{-4}	5.4×10^{-8}	-2.4×10^{-4}
mT	V_{Type}	3	748.6	749.1	5.1	0.02	3.388	1.1×10^{-3}		-5.42×10^{-4}		
mET	ELEV, V_{Type}	4	748.5	749.4	5.4	0.02	3.523	2.5×10^{-3}		-5.3×10^{-4}		-5.5×10^{-4}
mMT	V_{Mass}, V_{Type}	4	748.9	749.8	5.8	0.01	3.493	8.0×10^{-4}		-3.8×10^{-4}	9.2×10^{-8}	
mEMT	ELEV, V_{Mass}, V_{Type}	5	749.5	750.8	6.9	0.01	3.590	2.0×10^{-3}		-4.0×10^{-4}	7.5×10^{-8}	-4.6×10^{-4}
mM	V_{Mass}	3	750.4	750.9	7.0	0.01	3.274	4.9×10^{-4}			1.6×10^{-7}	
mEM	ELEV, V_{Mass}	4	751.2	752.1	8.2	0.00	3.346	1.5×10^{-3}			1.6×10^{-7}	-4.1×10^{-4}
m0	Null (Constant Value)	2	755.0	755.3	11.3	0.00	2.898	1.0×10^{-3}				
mE	ELEV	3	755.1	755.6	11.6	0.00	3.005	2.4×10^{-3}				-5.6×10^{-4}

Table 5. Relative importance and log evidence ratios for each covariate.

Covariate	Relative Importance	Logarithm of Evidence Ratio
DIST	0.920	1.064
V_{Type}	0.613	0.199
V_{Mass}	0.385	-0.204
ELEV	0.276	-0.419

In general, the modeled sedimentation values matched the observed values reasonably well. However, the modeled values had generally poor precision at lower sedimentation values (Fig. 11).

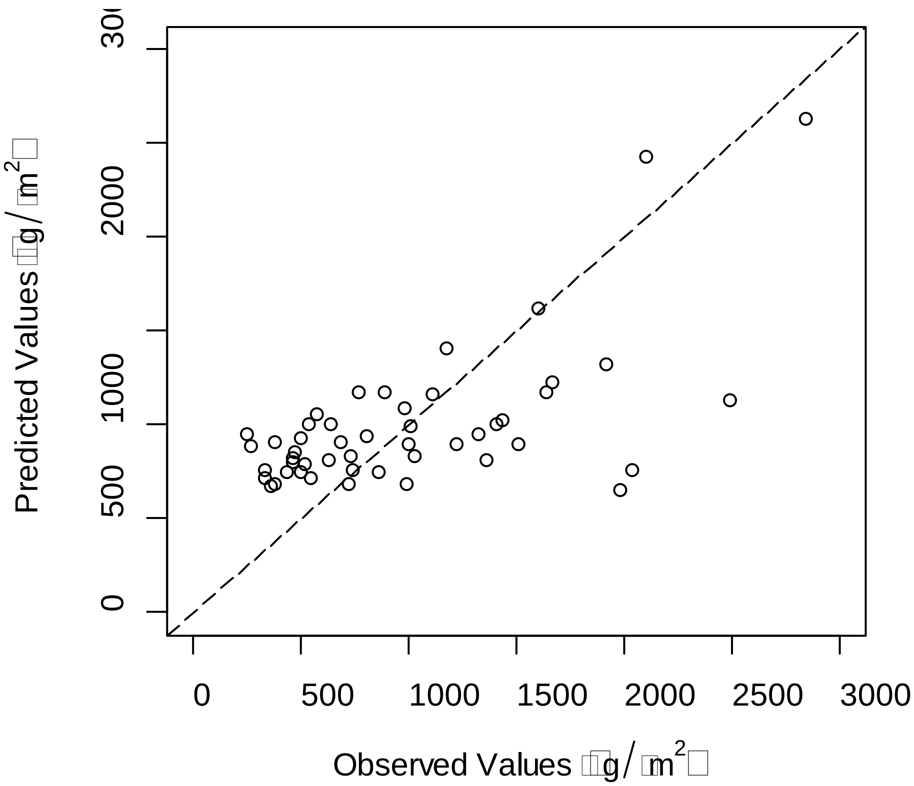

Figure 11. Predicted sediment deposition (g/m^2) plotted against observed sedimentation (g/m^2) displaying the variation between the best performing model and field observed values. The dashed line represents a 1:1 relationship.

We determined there was some degree of correlation between many of the model covariates. Vegetation type was moderately correlated with vegetation density (R = 0.46). Similarly, distance from the point of inflow was moderately correlated with sediment trap elevation (R = -0.42). We found distance from the point of inflow was somewhat correlated with both vegetation density (R = 0.33), and vegetation type (R = 0.27). Elevation and vegetation type were the only covariates not correlated with one another to some extent (R = 0.03) (Table 6).

Table 6. R values displaying the degree of correlation between covariates. R values for the categorical V_{Type} covariate were obtained by fitting linear models, whereas the remainder of the values were obtained through Pearson's analysis of correlation using r statistical software.

	DIST	ELEV	V_{Mass}	V_{Type}
DIST	1.000			
ELEV	-0.423	1.000		
V_{Mass}	0.325	-0.208	1.000	
V_{Type}	0.273*	0.032*	0.457*	1.000

CHAPTER 4

DISCUSSION

Given the dry and flashy nature of the precipitation in WY 2012, the year represented a somewhat abnormal season in regard to the flow pattern. The large basin geometry and elevated spill point of the wetlands required a large volume of water to fill the study site before flowing through the control point of the downstream gage. As the hydrograph receded following the storms, water remained trapped in the wetlands to evaporate and gradually infiltrate, leaving the sediment trapped in the wetlands. Given the precipitation events were short in duration and spaced some time apart, a greater percentage of suspended sediment remained trapped in the wetlands as compared to the previous two years of monitoring. Additionally, the total input of sediment was less in WY 2012 as compared to the previous two years. It is possible that the basin geometry and elevated spill point, combined with the short duration precipitation events observed in WY 2012 led to even less flow passing through the downstream gage than would have if there were larger and more closely spaced precipitation events.

Depending on the method used, we found a substantial difference in the total amount of sediment trapped in the wetlands. As determined through the mass balance sediment budget approach, 192 tonnes of sediment remained trapped on the study location. In contrast, only 52 tonnes of sediment was determined to be trapped in the wetlands through the surface interpolation of the sediment trap data. This difference may be partially accounted for by unmeasured sedimentation between the study wetlands and the upstream gaging station. Further, there are unstated uncertainties in both the gage-based calculations and turf mat-based calculations. Using bootstrapping techniques, Holloway (2010) found that the 95% confidence interval about the annual sediment transport mass was +/- 72% of the average value. If the same relative error applied to the current study, the gage-based value could be as low as 54 tonnes of trapped sediment, which is still more than the interpolated sediment trap data yielded.

One advantage to conducting a mass-balance sediment budget while collecting sediment trap data is the ability to compare each respective result, such that each acts as an independent check of one another. In this case, we gained greater insight into the variability of each approach. We can expect a wide range of error in both the mass balance sediment budget, and the sediment trap data interpolation approaches. Having conducted both approaches concurrently, we have a greater understanding of the sediment trapping ability of the study wetlands than if only one method had been used individually.

Qualitatively, it appears that the 30 m gridded sampling design sufficiently represented the range of elevation, vegetation, and distances from the point of inflow in the study wetlands. Field observations verified that the conditions at each sediment trap were representative of the adjacent area, and all major environmental features within the wetlands were sampled. These observations lend support to the efficacy of the sampling design. However, a greater number of recovered sediment traps were placed on *Polygonum* (43 sites) as compared to *Typha* (6 sites). This under sampling of *Typha* dominated plots was largely caused by the greater abundance of *Polygonum* in both study wetlands, and could only be corrected if a stratified-random sampling methodology had been used.

Through examination of the covariate evidence ratios, log evidence ratios, and AIC model comparison results, we found strong evidence that sediment trapping was influenced most by distance from inflow, some evidence that it was also related to vegetation type, and also some evidence that it was not driven by vegetation mass or elevation. However, there is some uncertainty regarding the influence of vegetation on sediment trapping due to the correlation between vegetation type and density, and distance from the point of inflow. It is also possible that given the low topographical relief of the study site, there was not a large enough range in elevation to play a significant role in sedimentation. However, Courtwright and Findlay (2011) found vegetative and physicochemical differences across small changes of elevation in a riverine wetland indicating that a large variation in wetland elevation may not be necessary to observe variations.

Given the degree of correlation present between several of the model covariates, care should be given when interpreting the modeling results. While it appears apparent that sedimentation patterns are driven primarily by distance from the point of inflow, it is less clear to what extent vegetation drives sedimentation values. We observed a moderate correlation between elevation and distance from the point of inflow ($R = -0.423$), and vegetation type and density ($R = 0.457$). We found somewhat of a correlation between vegetation type and distance from the point of inflow ($R = 0.273$), and vegetation density and distance ($R = 0.325$). Models containing vegetation type and vegetation density performed well. However, since both the type of vegetation and its density were correlated to some degree with distance from the point of inflow, the model performance may have been influenced by this correlation (Table 6). Given the natural (non-constructed) setting of the study wetlands, it is understandable that variables examined may be correlated with one another. Similarly, modeled variables may be spatially autocorrelated to some degree, which was not accounted for and may further undermine the certainty of the influence of vegetation on sedimentation in this system.

It would appear that, we observed high sedimentation values at locations of low vegetation density, which is counter intuitive, but may be partially explained by the correlation between covariates. Sample locations with high vegetation densities tended to occur at locations further away from the point of inflow (Pearson's $R = 0.33$). Previous studies have found differing effects of vegetation on sedimentation, where vegetation may prevent the resuspension of particles and thus increase local sedimentation rates (Braskerud 2001), or inhibit the flow of sediment laden water resulting in reduced sedimentation outside of preferential flow pathways (Harter and Mitsch 2003).

In general, our findings support the results of similar previous studies conducted in both natural systems and regulated treatment wetlands. The distance from the point of inflow was commonly found to be the primary factor driving sedimentation patterns (Asselman and Middlekoop 1998; Braskerud 2001; Stieger et al. 2001). Bannister et al. (2015) had similar findings, with increased sedimentation found near the main channel, and no relationship between elevation and soil properties.

Additional work must be conducted in natural wetlands to determine the influence vegetation has on depositional patters. We have—as have several others—found sedimentation patterns to be driven by the proximity to the point of inflow. Future studies should be carefully designed in order to minimize or account for correlations between vegetation type/density and the point inflow, which would confound the inference one could draw about specific effects.

CHAPTER 5

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APPENDIX A

TABLE OF SEDIMENT TRAP DATA

Mat ID	Elev (m)	Veg Mass	Vegetation Type	Dist from Inflow (m)	Dry Sed w/ Organics (g)	Trapped Sediment (g)	Normalized Trapped Sediment (g m^{-2})
A1	1.967	3.114	Polygonum	183.9	41.1	32.5	360.7
A2	2.109	4.085	Polygonum	165.9	38.2	29.8	331.3
A3	2.170	3.832	Polygonum	147.7	84.7	66.5	739.4
A4	2.170	5.708	Polygonum	136.0	55.8	46.9	520.8
A5	2.415	3.594	Polygonum	128.8	65.5	56.4	627.2
B1	2.091	4.752	Polygonum	153.2	47.0	39.2	435.3
B2	2.153	3.928	Polygonum	133.3	52.7	41.6	462.0
B3	2.257	3.649	Polygonum	115.1	48.6	42.7	474.3
B4	2.199	4.937	Polygonum	103.3	102.6	90.5	1005.6
B5	2.513	3.493	Polygonum	100.0	38.8	34.7	386.0
C1	2.079	5.562	Polygonum	146.3	37.0	30.6	339.7
C2	1.971	3.401	Polygonum	124.3	49.1	41.9	465.8
C3	2.287	4.307	Polygonum	102.5	70.8	61.6	684.1
C4	2.269	4.734	Polygonum	81.3	104.1	90.7	1007.6
C5	2.304	2.751	Reed	69.9	160.3	144.1	1600.8
D1	2.221	3.658	Polygonum	123.0	76.0	65.6	728.7
D2	2.374	1.829	Polygonum	94.4	83.8	72.7	807.9
D3	2.385	3.747	Polygonum	68.6	60.4	51.8	575.5
D4	2.322	4.258	Polygonum	49.8	112.4	100.5	1116.6
D5	2.212	0.581	Polygonum	40.0	167.4	149.9	1665.1
DT1	1.461	0.188	Polygonum	154.3	214.6	184.8	2053.7
DT2	1.378	1.155	Polygonum	186.3	288.5	258.8	2875.2
E1	2.122	2.02	Reed	18.4	288.3	255.7	2841.4
F1	2.580	1.446	Reed	90.1	119.0	106.1	1178.9
F2	2.464	0.744	Reed	119.6	171.0	147.8	1642.8
F3	2.374	1.976	Polygonum	149.6	208.9	183.1	2034.9
F4	2.348	1.906	Polygonum	178.4	41.3	34.5	382.8
G1	2.478	5.707	Polygonum	193.1	210.1	178.2	1980.2
G2	2.364	7.084	Polygonum	164.2	58.0	49.5	549.8
G3	2.524	2.437	Reed	134.5	101.3	88.2	980.4

G4	2.494	5.53	Polygonum	104.3	154.1	135.5	1505.4
G6	2.772	3.171	Polygonum	48.3	87.1	80.4	893.0
G7	2.537	1.774	Reed	25.1	206.1	189.5	2105.4
G8	2.740	3.638	Polygonum	26.9	186.0	172.5	1916.8
G9	2.672	-	Polygonum	51.9	109.0	100.5	1116.4
G10	2.503	-	Polygonum	82.3	118.7	105.1	1167.9
H1	2.261	3.15	Polygonum	179.7	96.8	65.2	724.3
H2	2.417	3.15	Polygonum	150.9	53.1	45.0	499.9
H3	2.458	4.567	Polygonum	123.5	105.8	93.0	1033.1
H4	2.714	3.061	Polygonum	96.6	50.0	45.2	501.8
H5	2.640	3.335	Polygonum	73.9	141.5	129.1	1434.8
H6	2.642	2.183	Polygonum	54.2	247.0	223.8	2486.2
H7	2.663	3.465	Polygonum	47.1	76.0	69.2	769.2
H9	2.610	3.604	Polygonum	79.2	53.8	48.3	536.6
H10	2.700	4.188	Polygonum	106.0	32.6	24.4	271.1
I1	2.411	2.956	Polygonum	178.5	99.8	89.1	990.1
I2	2.349	2.869	Polygonum	153.7	89.8	77.6	862.1
I3	2.483	4.048	Polygonum	129.2	138.4	122.3	1359.3
I4	2.517	2.83	Polygonum	105.4	127.0	110.4	1226.7
I5	2.638	3.637	Polygonum	90.4	134.8	119.5	1327.7
I6	2.582	2.384	Polygonum	78.8	141.4	127.0	1411.5
I7	2.603	3.048	Polygonum	79.1	65.6	57.9	643.5
I8	2.828	2.462	Polygonum	91.9	25.9	22.9	254.0
I9	3.095	2.053		108.9	0	0	0
J1	3.016	3.619		120.4	0	0	0
J2	2.963	2.358		109.6	0	0	0

APPENDIX B

AIC MODEL COMPARISON R CODE

```
# mGr: model Gamma reciprocal (i.e. inverse)
mGr0 = glm( Sed ~  1, data = dat, family = Gamma);
mGrD = glm( Sed ~  Dist, data = dat, family = Gamma)
mGrE = glm( Sed ~  Elev, data = dat, family = Gamma)
mGrM = glm( Sed ~  VEG_Mass, data = dat, family = Gamma)
mGrT = glm( Sed ~  VEG_Type, data = dat, family = Gamma)
mGrDE = glm( Sed ~  Dist + Elev, data = dat, family = Gamma)
mGrDM = glm( Sed ~  Dist + VEG_Mass, data = dat, family = Gamma)
mGrDT = glm( Sed ~  Dist + VEG_Type, data = dat, family = Gamma)
mGrEM = glm( Sed ~  Elev + VEG_Mass, data = dat, family = Gamma)
mGrET = glm( Sed ~  Elev + VEG_Type, data = dat, family = Gamma)
mGrMT = glm( Sed ~  VEG_Mass + VEG_Type, data = dat, family = Gamma)
mGrDEM = glm( Sed ~  Dist + Elev + VEG_Mass, data = dat, family = Gamma)
mGrDET = glm( Sed ~  Dist + Elev + VEG_Type, data = dat, family = Gamma)
mGrDMT = glm( Sed ~  Dist + VEG_Mass + VEG_Type, data = dat, family = Gamma)
mGrEMT = glm( Sed ~  Elev + VEG_Mass + VEG_Type, data = dat, family = Gamma)
mGrDEMT = glm( Sed ~  Dist + Elev + VEG_Mass + VEG_Type, data = dat, family =
      Gamma)

aic = AIC(
      mGr0,mGrD,mGrE,mGrM,mGrT,mGrDE,mGrDM,mGrDT,mGrEM,mGrET,mGrMT,mGrDEM,mGr
      DET,mGrDMT,mGrEMT,mGrDEMT )

fredsAICtable <- function( aic, n) {
  K <- aic$df
  AICc <- aic$AIC + 2 * K * (K+1) / ( n - K - 1 )
  delAIC<- AICc - min( AICc )
  AICw <- exp(-0.5*delAIC) / sum( exp(-0.5*delAIC))
  #This is the AIC table to be published:
  data.frame( aic, AICc, delAIC , AICw)
}

aic=fredsAICtable( aic, length(dat[,1]) )
round(aic[order(aic$AICw,decreasing=T),],2)
```