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Seasonal Transport of Suspended Solids and Nutrients Between Bear River and Bear Lake

Cody M. Allen
Utah State University

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SEASONAL TRANSPORT OF SUSPENDED SOLIDS AND NUTRIENTS BETWEEN BEAR RIVER AND BEAR LAKE

by

Cody M. Allen

A thesis submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

In

Watershed Sciences

Approved:

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UTAH STATE UNIVERSITY
Logan, Utah
2012
ABSTRACT

Seasonal Transport of Suspended Solids and Nutrients Between Bear River and Bear Lake

by

Cody Allen, Master of Science
Utah State University, 2012

Major Professor: Nancy O. Mesner
Department: Watershed Sciences

Dingle Marsh is a wetland complex separating the Bear River from Bear Lake. Flow direction through the marsh is controlled at four major inflow and outflow sites. These sites were chosen as monitoring sites to assess the suspended solid and nutrient transport through the marsh. High frequency turbidity measurements were collected at each site and used as a surrogate for total phosphorus (TP) and total suspended solid (TSS) concentrations. Loads of TP and TSS were calculated using flow data from the 2008 water year. Load calculations for TP and TSS were compiled at 30-minute intervals and annual mass balances were calculated for Dingle Marsh and Bear Lake. These calculations were used to identify the seasonal loading patterns within this system.

This study found the majority of TSS and TP loading entered the marsh from the Bear River. As flows moved across the marsh, the loading of TSS and TP was greatly reduced. Seasonal flow patterns were analyzed to determine the loading patterns to
Dingle Marsh, Bear Lake, and the Bear River. This study also identified water management strategies aimed at setting a target endpoint for TSS and TP loads.
PUBLIC ABSTRACT

Seasonal Transport of Suspended Solids and Nutrients Between Bear River and Bear Lake

by

Cody Allen, Master of Science
Utah State University, 2012

I measured inflows and outflows of nutrients and suspended solid through the wetland complex, known as Dingle Marsh, at the Bear River Migratory Bird Refuge between October 2007 and September 2008. My analysis of changes throughout the year will help Dingle Marsh managers adjust the timing and volume of water movement between Bear River and Bear Lake to meet defined refuge goals, such as improving bird or other animal habitat. These results could also be used to protect Bear Lake’s water quality.

Automated samplers took readings of dissolved oxygen, turbidity, pH and water temperature every 30 minutes. These values were combined with flow measurements and used to predict suspended solids, total phosphorus and nitrogen.

The high frequency monitoring allowed for the analysis of nutrient and suspended solid concentrations at multiple time scales. Coupling the high frequency water quality data with discharge measurements, the mass of nutrient and suspended solid parameters
could be calculated. The ability to calculate mass at fine time scales allowed for insight into the behavior of this system throughout seasonal changes. This research provided a viable method for tracking nutrient and suspended solid transport in this system, and provided methods that could be applied for uses in similar research.

During my sampling period, most of the suspended solid and nutrient loading entered the marsh from the Bear River. This loading was greatly reduced as flows moved across the marsh. Since the behavior of this system is largely driven by yearly climate patterns, the seasonal loading patterns were defined and analyzed. The results of this study show Dingle Marsh to be an effective retention basin for suspended solids and nutrients. This research also provided a viable method for tracking nutrient and suspended solid transport in this system, and provided methods that could be applied for uses in similar research.
ACKNOWLEDGMENTS

I am grateful for my advisor, Nancy O. Mesner, for the direction and assistance she provided to me. I am thankful for her patience and understanding throughout the process. I appreciated the help I received from my other committee members, David K. Stevens and Wayne A. Wurtsbaugh, who gave of their expertise in this field of research. I want to thank Jeff Horsburgh for installing all of the monitoring equipment, and teaching me how to use it. Jeff also accompanied me on many long field days, which I will always be grateful for. I want to thank the Bear Lake Wildlife Refuge for allowing me access to the monitoring sites and for providing expertise. I give thanks to Connely Baldwin and PacifiCorp for providing locations for monitoring equipment and for providing data throughout the project. I am grateful to Idaho Department of Environmental Quality for funding this project.

I will be forever thankful to my family for their never-ending love and support, and for never letting me get discouraged. I especially want to thank my wife, Shelsie, who has been my main support every day since I began this journey, and I know I could not have done it without her. I also want to thank Josie; she is the source of the inspiration that drives me to become better every day of my life.

Cody Allen
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CHAPTER 1

INTRODUCTION

This study focuses on the dynamics of nutrient and suspended solid transport through a large wetland complex and the role that this wetland complex plays seasonally as a source and/or sink of suspended solids and nutrients to waters downstream. The study area, Dingle Marsh, is used to route water from the Bear River to Bear Lake for irrigation storage and is also used to transport irrigation releases from Bear Lake to the Bear River further downstream. The study’s goal was to better understand the role this specific wetland plays in suspended solid and nutrient transport and retention as water moves through the complex. This improved understanding can potentially be used for improved water management of these wetlands. The site serves as a natural laboratory to explore how the relationship between flow and residence time affects the role of wetland complexes as sources or sinks of suspended solids and nutrients.

The specific objectives of this study are:

1. Examine the overall mass balances for both the Dingle Marsh system, and imports versus exports of suspended solids and nutrients to Bear Lake (including transport/flux and storage).

2. Evaluate implications of the mass balances and fluxes on management of this aquatic system.

3. Identify and test best monitoring techniques to characterize this system.

In order to meet these objectives the following questions must be addressed:

1. Can the fate and transport of total suspended solids (TSS) and total phosphorus (TP) entering Dingle Marsh from the Bear River be quantified, predicted or managed?

2. What is the mass and timing of movement of suspended solid, phosphorus and nitrogen?
3. How does loading change seasonally? Are these changes gradual or rapid?

4. Are seasonal loading changes predictable? What factors are important?

5. What kind of loading behavior was observed in past studies? How do these historic observations compare with new observations?
CHAPTER 2
LITERATURE REVIEW

SUSPENDED SOLIDS

This discussion focuses on the potential impacts of suspended solids with special emphasis on wetland complexes. Transport of suspended solids in a river is a natural physical process of the river; however, excess suspended solids in aquatic systems can alter the physical structure of that system and can impact various ecological functions. Suspended solids can be carried by rivers and streams if the upward force of the current is higher than the settling velocity of the suspended solid particle. As the settling velocity of a suspended solid particle approaches the upward force of the current, the higher the likelihood of that particle being moved as bed load transport. Bed load transport is the movement of suspended solid particles by sliding and/or rolling along the bottom of the flow. As the upward force of the current exceeds the settling velocity of a suspended solid particle, the particle can be moved as suspended solid. The concentration of suspended solids is a function of flow velocities, which must be high enough to keep the particles from settling (Lane, 1955). Therefore, suspended solid concentrations vary seasonally with flow.

Increasing concentrations of suspended solids will lead to a reduction in the visual clarity of the water column. The clarity of the water is affected by the number of particles suspended in the water column. These particles affect light penetration into the water by reflecting and absorbing light photons as they move through the water (Van Nieuwenhuyse and LaPerriere, 1986; Wood and Armitage, 1997; Davies-Colley, 2001). Suspended solid can be measured gravimetrically, but is often closely associated with
turbidity which is a measurement of light scattering through a predetermined volume of water.

One impact of high suspended solid concentrations is the reduction in the water column of light energy needed for photosynthesis (Ryan, 1991; Davies-Colley, 2001). This effect is most noticeable in systems which rely heavily on autochthonous energy sources (Davies-Colley et al., 1992). Reducing the capacity of primary production in an ecosystem may have a ripple effect throughout the food web (Henley et al., 2000). By interrupting the flow of energy to primary producers, high suspended solid concentrations have the potential to reduce energy at multiple levels of an aquatic system (Ryan, 1991).

Suspended solids can further affect the aquatic food web by reducing the range in which visual predators can locate prey. The effectiveness of both terrestrial and aquatic predators is greatly reduced as water clarity is degraded and as prey species become covered in silt (Ryan, 1991). This disruption in the natural balance between predator and prey can allow for the rapid expansion of nuisance species.

High levels of suspended solids may also cover or irritate gills of fish and other aquatic organisms (Chapmann, 1988; Ryan, 1991; Schalchi, 1992; Richards and Bacon, 1994; Davies-Colley, 2001). Increased suspended solid may also cause interstitial spaces in salmonid spawning beds to fill, leading to reduced egg survival and an increased likelihood of predation (Wood and Armitage, 1997).

Accumulation of fine sediment can physically alter the characteristics of an aquatic system. A major concern with excess sedimentation in a wetland system is the resulting channelization. When excess amounts of fine sediments are deposited in a water body, yet water is still forced through, the flow will carve channels through the
once lentic system. These channels become high speed conduits for carrying excess suspended solids, nutrients and other pollutants, thus reducing the wetland system’s potential for filtering and processing potentially harmful substances. An accurate estimate of suspended solid concentrations and loads can help to understand the life span of reservoirs by determining the yearly delivery of fine sediment. These estimates also aid land use managers in determining the success or failure of land use management techniques designed to reduce erosion or soil loss to a water body (Minella et al., 2008).

Problems with channelization and sediment aggradation will continue to plague the health and utilization of the impaired water body as long as suspended solid concentrations exceed natural background levels. Furthermore, the problems from high levels of suspended solid can be cyclic as wind action and flood pulses can re-suspend sediment particles causing increased damage to the health of the wetland.

Suspended solids are also a delivery pathway for other pollutants in aquatic systems. Phosphorus, metals and bacteria may bind to suspended solid particles as they travel with the flow. The binding between pollutants and suspended solid particles usually occurs when negatively charged pollutant ions bind to positively charged ions or surfaces on suspended solid particles (Philips et al., 1999; Christensen et al., 2000). Depending on the type or strength of the bond, phosphorus, metals or other materials may be released with changing conditions in deposited sediments, such as changes in pH, dissolved oxygen, temperature, or biological activity and perturbation.

Suspended solids are usually made up of fine particles of sand, silt, clay and organic matter that have been weathered from the upstream watershed. Anthropogenic activities such as poorly managed agricultural land, forestry clear cuts, unregulated
mining practices, road construction, or the lack of riparian buffer zones can lead to increased suspended solid loads (Ryan, 1991; Davies-Colley et al., 1992; Houser, Mullholland, and Maloney, 2006; Harris et al., 2007). The sources and effects of suspended solid throughout the world are highly variable (Wood and Armitage, 1997) further emphasizing the need for fully understanding the characteristics of the loading of suspended solids in a variety of ecosystems. Considering the potentially damaging effects of excess suspended solid concentrations to aquatic systems, it is imperative to have accurate and precise measurements of these parameters.

NUTRIENTS

Nutrients are elements which plants need to survive and grow. All natural aquatic systems require nutrients in order to be healthy and productive, but over enrichment of these nutrients can greatly deteriorate the physical condition of an aquatic ecosystem. The two most common nutrients of concern in aquatic systems are nitrogen and phosphorus. Both of these nutrients are abundant in nature and in some forms are also quickly assimilated by plants and algae. Natural sources, cycling and transport of nitrogen and phosphorus in aquatic systems are quite different.

Biologically available nitrogen is introduced into aquatic systems through atmospheric deposition, decomposition of plant material and fixation by cyanobacteria (Camargo et al., 2005). The basic forms of nitrogen are: ammonium, ammonia, nitrate, nitrite, organic nitrogen and nitrogen gas. Nitrogen is most abundant in nitrogen gas form which is largely unavailable for use by plants. Nitrogen fixation is the process in which bacteria convert the nitrogen gas into ammonia which is usable by plants. Nitrogen fixation can also occur through the industrial production of fertilizers which
produce the ammonium or nitrate form of nitrogen. Ammonium will eventually go through nitrification by aerobic bacteria producing nitrate. Since nitrate is bioavailable, it may be transported in the water column, taken up by plants, or if anoxic conditions exist it can be denitrified by anaerobic bacteria, creating nitrogen gas. As nitrate is taken up by plants, the form of the nitrogen is changed into organic nitrogen. When the plant dies, bacteria or fungi decompose the plant matter and the organic nitrogen goes through a process of ammonification, changing the nitrogen back into ammonium (Wetzel, 2001).

Unlike nitrogen, phosphorus has no natural gaseous form and enters aquatic systems from the natural weathering of rocks, fertilizers and sewage. Inputs of phosphorus into a system can generally be pinpointed to a source within the watershed whereas the nitrogen cycle takes place on a global scale. The total phosphorus in aquatic systems can be divided into dissolved and particulate forms. Dissolved phosphorus includes the ion orthophosphate, which is easily assimilated by plants, as well as dissolved polyphosphates and organic phosphorus (Wetzel, 2001). Particulate phosphorus includes phosphorus in mineral forms, phosphorus incorporated into biological materials, phosphorus adsorbed to sediment and phosphorus that precipitates when combining with certain metals such as aluminum. The bioavailability of these particulate forms varies. In addition, adsorbed and precipitated phosphorus may be released back to a dissolved form under certain chemical or physical conditions in the water column or sediments. Particulate phosphorus can be transported in rivers for long distances with adequate flow (Grayson et al., 1996). Suspended particulate phosphorus will continue to be transported until reaching an area with reduced flow velocities where it is deposited on the bed of the receiving water. As phosphorus settles in a receiving
water body, it can be processed into biological materials, incorporated into the soils, or moved if plant material is removed. Phosphorus can also be released under anaerobic conditions, high pH conditions, and through biological decomposition. Varying levels of temperature, oxygen, pH and some cations can all affect the levels of dissolved phosphorus (Kronvang et al., 1997; Correll, 1998).

The process by which excess nutrients cause the over-fertilization of a water body is called eutrophication. Eutrophication occurs naturally over hundreds to thousands of years and is part of the aging process of lakes as they gradually fill (Correll, 1998). Some nutrients and suspended solids are delivered to an aquatic system naturally through storm and flood events and also from the naturally occurring minerals found throughout the watershed. Problems with nutrients generally occur when extra loading is added to an aquatic system by the overuse of fertilizer in agricultural systems, urban and agricultural runoff, industrial waste and sewage effluence. Excessive nutrients delivered to an aquatic system can accelerate alga growth and algae can reach nuisance levels.

As organic matter accumulates on the benthos of a river, lake or wetland decomposition begins. Bacteria are the primary decomposers and use aerobic respiration to obtain energy from organic material. When this process consumes most of the dissolved oxygen within the water body, fish kills can result. Excess algae also affect water color, odor and taste and can have far reaching consequences on the public perception of the capability of the water body to sustain recreational and municipal uses. These processes have become a more common problem with ever increasing anthropogenic impacts on water quality (Christensen et al., 2002).
Watershed management is critical in reducing nutrient inputs to water bodies. Proper management of reservoir and other hydrologically controlled systems can also reduce the input of excess nutrients. Monitoring programs are critical to identify sources and loads of nutrients, to differentiate between natural and anthropogenic sources and to predict and avoid adverse impacts (Correll, 1998).

**WATER QUALITY SAMPLING**

The cost of water quality samples and the difficulty of maintaining rigorous sampling regimes have resulted in limited or no storm event sampling and poor characterization of seasonal data (Gray, 2002; Ankcorn, 2003; Mesner and Paige, 2011). Factors such as weather events and changes in seasons can also restrict access to sites (Spackman et al., 2008). As a result, data collected for water quality assessment monitoring or Total Maximum Daily Load estimates are often taken at monthly or quarterly frequencies. This sampling frequency is problematic if most aquatic processes involving suspended solids and nutrients take place on much finer time scales. Concentration of suspended solids and associated contaminants often peak during flood and storm events, so the inability to sample at these times can result in a substantial loss of information. Monthly (or less frequent) sampling will likely miss these large events when most of the contaminants are transported (Grayson et al., 1996; Kronvang et al., 1997; Johnes, 2007; Jones, 2008).

Many studies have been conducted to determine the sampling frequency necessary to most closely predict the true load. Johnes (2007) found that sampling at low sampling frequencies could overestimate the load by 196% to 452% of the true load. In contrast, Jones (2008) found that infrequent sampling missed rare but significant loading
events, resulting in underestimates of the actual load. Johnes (2007) found that at sites with low baseflow, a single high flow event could account for up to 20% of the annual load of total phosphorus, with the top five flow events in the year contributing up to 42% of the annual load. In systems with high baseflows, the impact of single events was less significant. In these systems, the highest flow event typically accounted for <1% of the annual load of total phosphorus (Johnes, 2007). Kronvang et al. (1997) compared sampling at two week intervals with intensive (hourly) storm monitoring. The less frequent sampling underestimated annual transport by 24% to 331% for suspended solids and by 8% to 151% for particulate phosphorus. In another study, Scholefield et al. (2005) undertook an intensive hourly monitoring experiment for 90 hours and found extreme variability in both phosphorus and nitrate concentrations. These studies suggest that suspended solid and nutrient concentrations are dynamic on a very short time scale, changing on the order of days and sometimes even hours. As sampling frequency is reduced, the precision of load estimates is also reduced (Grayson et al., 1996; Phillips et al., 1999; Christensen et al., 2000).

One approach for the collection of higher frequency water quality samples is to use a surrogate measure in place of the parameter of interest (Spackman et al., 2008). This approach is only valid if a measurable parameter correlates well to the parameter of interest and if sufficient water quality data are collected to generate these relationships (Gray, 2002). This method is only useful when the surrogate is easier and cheaper to collect than the parameter of interest (Grayson et al., 1996).

Stream discharge has often been used as surrogate for suspended solid and less frequently, nutrient concentrations (Brasington and Richards, 2000; Ankcorn, 2003)
because stream flow is the primary delivery force for transporting particles. The relationship between discharge and concentration is complicated by the first flushing of nutrients, with lower concentrations often seen on the descending limb of a hydrograph (Brasington and Richards, 2000; Ankorn, 2003; Stubblefield et al., 2007). Suspended solid and nutrient concentrations are highly variable, because they vary with the duration and magnitude of the flow event and also the season and the source of water (Grayson et al., 1996). Horowitz (2003) found suspended solid rating curves tended to under predict high, and over predict low suspended solid concentrations.

Historical flow data should only be used to predict suspended solid or other concentrations when it can be confirmed that the relationship between discharge and concentration has not changed over time. Rating curves must be checked regularly to confirm that relationships still hold, and must be updated if the curves are no longer accurate.

Turbidity is another relatively easy measurement that has been correlated to suspended solid and nutrient concentrations (Gippel, 1995; Grayson et al., 1996; Kronvang et al., 1997; Spackman et al., 2008). Because turbidity is a measure of the light scattered by suspended particles, it is possible to correlate the measurement to the amount of suspended solid in the water column (Ziegler, 2002; Minella, 2008). When particulate nitrogen or phosphorus comprises most of the total nitrogen or phosphorus in a system, then turbidity may also be an adequate surrogate for these nutrients (Grayson, 1996). Minella (2008) found that in eight monitored events in the period of one year, turbidity explained from 80.9% to 98.4% of the variability in suspended solid concentrations. Although the relationships between turbidity and suspended solids varied
slightly among different events in this study, turbidity was sensitive to the peaks and troughs of suspended solid concentrations. In testing between turbidity and discharge as suspended solid surrogates, Lewis (1996) found turbidity consistently estimated the loading of suspended solid within 8% of the true load. In contrast, discharge was only able to estimate within 24% of the true load. Stubblefield et al. (2007) used turbidity as a surrogate for TSS and TP concentrations in a watershed subject to flushing and depletion of suspended solids and nutrients. Stubblefield et al. (2007) found correlations of 0.95 and 0.91 between turbidity and TSS, and for TP found correlations of 0.62 and 0.83.

Many studies that use turbidity as a surrogate for suspended solid or nutrient concentrations may also use additional surrogates such as specific conductivity, pH, water temperature, and dissolved oxygen to strengthen correlations, but turbidity is found to be the most important surrogate (Kronvang et al., 1997; Gray, 2002; Jones, 2008).

The size, composition and color of suspended particles can all influence turbidity (Gippel, 1995). Particle size has an influence on turbidity measurement because as individual particle size increases, it also increases the amount of light dispersed from the turbidity sensor beam. The size of suspended particles is directly related to the mineral composition of the catchment that is being monitored, which means turbidity measurements in an individual catchment are not necessarily comparable to other catchments (Gippel, 1995). The composition of suspended particles at a site also influences the turbidity value, because various types of particles will have differing light scattering characteristics. For example, suspended organic matter may only deflect a small portion of the light that it comes in contact with. In contrast, a suspended mineral particle may scatter all of the light that it comes in contact with (Gippel, 1995). Water
color is another factor that may influence turbidity readings because varying colors in the water will absorb diverse amounts of light which will influence turbidity measures (Gippel, 1995). Care should also be taken to assure that turbidity measurements are comparable because different methods and reporting units produce different results (Ziegler, 2002; Ankcorn, 2003).

The relationship between turbidity and suspended solid or phosphorus varies between sites and seasonally at a single site. (Gippel, 1995; Lewis, 1996; Pavanelli and Pagliarani, 2002). Riley (1998) found relationships between turbidity and suspended solid concentrations not only differed among sites in the same watershed, but also found seasonal changes could affect the relationship. Similarly, Jones (2008) found that turbidity could be used as a surrogate for TSS and TP, although the relationships were subject to spatial and seasonal factors. Studying multiple sites on a single river, Grayson et al. (1996) found linear correlations for TP with an $R^2$ of 0.91. Grayson et al. (1996) did not develop site specific correlations for TP and turbidity, and the authors suggest this may have introduced spatial variability. The relationship of turbidity to other parameters may also vary with changes in land use practices upstream of the sampling site (Houser et al., 2006; Harris et al., 2007).

Surrogate sampling is not a direct measurement of the parameter of interest and the predictive power of the relationship is only as good as the calibration of that relationship. To establish an accurate relationship between turbidity and other parameters, water samples must be analyzed over the full range of turbidity values (Gippel, 1995; Riley, 1998; Ankcorn, 2003).
DINGLE MARSH

Dingle Marsh is a natural wetland system located at the northern edge of Bear Lake in southeastern Idaho (Figure 1). The marsh is located in the Bear Lake watershed within the Bear River Basin. Several open water bodies within the marsh include Mud Lake and Bunn Lake (Figure 2).

![Figure 1: Location of Dingle Marsh, Bear Lake National Wildlife Refuge, Idaho](image)

Dingle Marsh was established as a National Wildlife Refuge in 1968 to protect and improve habitat for various waterfowl species (Bear Lake National Wildlife Refuge, 2006). Its watershed has an annual precipitation of 28 to 140 cm. Most precipitation falls at higher elevations and is released throughout spring and summer as the snowpack melts. The land within the Bear Lake watershed is nearly 50% privately owned with the primary land use being rangeland and agriculture (Bear River WIS, 2005). The Dingle Marsh wetland covers 17,600 acres, of which 16,000 acres are wetted land. The Dingle
Marsh complex is public land that is managed by the U.S. Fish and Wildlife Service as National Wildlife Refuge. The refuge was established to protect and manage habitat for waterfowl and other migratory birds, specifically western Canada geese and greater sandhill cranes (Bear Lake National Wildlife Refuge, 2006). In areas of the refuge that are flooded seasonally, refuge managers cut hay to provide short cover feeding sites that provide rearing habitat for waterfowl. The refuge also cultivates barley and alfalfa to provide food crops for waterfowl and to reduce waterfowl feeding in farm lands adjacent to the refuge. Bulrush is the dominant vegetation throughout the marsh and provides cover and nesting material for waterfowl. Herbicide treatments and prescribed fire are the primary methods in controlling invasive species and allowing for the natural succession of wetland plants (Bear Lake National Wildlife Refuge, 2006).

Since 1911, water from the Bear River has been diverted through Dingle Marsh and stored in Bear Lake until needed downstream for agricultural purposes. Currently the flow of water entering and leaving Dingle Marsh is controlled by PacifiCorp at various water control structures. The four main water control structures are the Rainbow Inlet canal (Inlet), Causeway, Lifton pump house, and the Paris Dike Outlet (Outlet) (Figure 2).

Geologic evidence suggests that the Bear River was historically connected with Bear Lake, entering through the marshes (present day Dingle Marsh) to the north. The most recent natural direct connection between the river and the lake is estimated to be at least 10,000 years ago (USGS, 2005; Rehis et al., 2009). The river and marsh appear to have been intermittently connected; however, evidence suggests that during high runoff years Bear River flood waters would enter the marsh and ultimately connect to Bear Lake.
by a small natural channel (Reeves, 1954). Conversely, when water levels were high in Bear Lake, water would flow from Bear Lake into Dingle Marsh.

Figure 2: Location of study sites and general flow paths throughout Dingle Marsh.

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River flood waters would enter the marsh and ultimately connect to Bear Lake by a small natural channel (Reeves, 1954). Conversely, when water levels were high in Bear Lake, water would flow from Bear Lake into Dingle Marsh.

In 1907, construction of a canal began which would eventually direct the flows of the Bear River permanently through Dingle Marsh and into Bear Lake. The purpose of the diversion canals was to use Dingle Marsh and Bear Lake for water storage, releasing water later in the summer for irrigation uses and for secondary power generation. The canal system was operative in 1911, but construction was not completed until 1918. Prior to the construction of the canal system, Dingle Marsh was primarily a fresh water discharge system fed by local runoff and multiple groundwater springs (Reeves, 1954). The canal connections to Bear Lake and the Bear River resulted in a turbid flow-through system (Bear Lake National Wildlife Refuge, 2006).

The Bear River is diverted at Stewart Dam and nearly the entire flow is sent down the Inlet canal towards an open water area of the marsh known as Mud Lake. A small stream of water still flows down the old Bear River channel, mostly the result of the inefficiencies of the dam and local accrual. The hydrology of Dingle Marsh is currently controlled by PacifiCorp in accordance with agreements made with the USFWS Bear Lake National Wildlife Refuge (Bear Lake National Wildlife Refuge, 2006). Releases of stored Bear River water is determined primarily by downstream irrigation demand with power generation as a secondary benefit. Flows are controlled at four main structures: the Rainbow Inlet canal (Inlet) which receives Bear River flows diverted at Stewart Dam, the Causeway where water enters Bear Lake by gravity flow, the Lifton pump house where water is released or pumped into the Outlet canal from Bear Lake, and the Paris
Dike Outlet canal (Outlet) which delivers water back to the Bear River. Management of these different control structures and the relative elevation of water in Dingle Marsh, the Bear River and Bear Lake all determine the direction of flow through this system.

When the Causeway is open (typically winter through spring), Bear River water flows through Dingle Marsh into Bear Lake through the Causeway. When the Causeway is closed, water flows to the Outlet canal and returns to the Bear River. As water travels from the Inlet canal to the Causeway it flows through Mud Lake, which is a large open water area of the refuge. The elevation of Mud Lake is influenced by the management of water levels at all four control sites. During the irrigation season, water is pumped from Bear Lake at the Lifton pumping station into the Outlet canal to the Bear River.

Current management of water in the Dingle Marsh system results in three main flow patterns each year. During the baseflow phase in winter and spring, water moves from the Inlet to the Causeway replenishing storage in both Mud Lake and Bear Lake. The next phase is lake fill and downstream release, which occurs during summer when water demands downstream are relatively low. During this phase, flows through the Causeway are decreased as flows at the Outlet are increased. The timing of this phase depends on the supply of upstream water and the demand downstream. The lake withdrawal phase is the last of the three periods, initiated to supply water during the main irrigation season. Water moves from the Inlet to the Outlet and water from Bear Lake is pumped through the Lifton Station to the Outlet.

Prior to 1993, this pattern was slightly different. Lifton was the primary site for water flow into and out of Bear Lake, and the Causeway was a secondary site for water movement into Bear Lake. In 1993, high flows caused a failure of the Causeway site
resulting in uncontrolled flow from Dingle Marsh into Bear Lake. This failure necessitated the rebuilding of the Causeway site which allowed it to be designed as the primary site for inflow to Bear Lake. From that point on, Lifton has been used only for outflow from the lake. The pattern outlined here summarizes water management during a typical water year although there is considerable inter- and intra-annual variability in the amount of snow pack above Bear Lake, the timing of spring runoff, water elevations in Bear Lake and Dingle Marsh at the beginning of the water year, and downstream irrigation demands.

**WATER QUALITY STUDIES OF DINGLE MARSH**

Bear Lake is home to a trophy sport fishery, four endemic fish species, and several species of endemic invertebrates (Bjornn et al., 1989). The local economy is largely based on fishing and other tourism resulting from the lake’s remarkable blue color. Recreation and fisheries would be adversely affected by eutrophication or other impacts to Bear Lake water quality. The water quality from Bear River may impact waterfowl habitat in Dingle Marsh. Suspended solid accumulation in the marsh also affects the marsh’s storage capacity.

Due to high concentrations of solids and nutrients, multiple studies have addressed the potential impact of Bear River water on Bear Lake’s oligotrophic state (Nunan, 1972; USEPA, 1975; Lamarra, 1980; Lamarra et al., 1983; Birdsey, 1989; Bjornn et al., 1989). Studies have focused both on Bear Lake conditions and on the combination of human and natural impacts that affect the water in the Bear River (Nunan, 1972; ERI, 1992; ERI, 1998).
A 1974-1975 National Eutrophication Study by the United States Environmental Protection Agency found Bear Lake to be oligotrophic but subject to mesotrophic loading from the Bear River (USEPA, 1975). This study predicted that hypolimnetic depression or depletion of oxygen would become increasingly evident if mesotrophic loading from the Bear River continued. These changes were not expected to be rapid because of the large volume of the lake and the lengthy hydraulic retention time of Bear Lake.

Lamarra (1980) found that low Bear River water years were correlated with lower oxygen deficits in Bear Lake’s hypolimnion. In contrast, high Bear River flow years correlated with high hypolimnetic utilization of oxygen.

Lamarra et al. (1983) focused on nutrient loading into Bear Lake and specifically on the role of Dingle Marsh. From April 1982 to June 1983, eight major inflow and outflow sites to the marsh were sampled once per month. In addition, nine sites were located within the interior of Dingle Marsh, intended to capture the processing of suspended solids and nutrients across the marsh. Dingle Marsh was found to be a net sink for total phosphorus (TP), nitrate (NO₃-N), and total suspended solids (TSS). The marsh captured total organic carbon (TOC) and total nitrogen (TN) when water was flowing through the marsh to Bear Lake. The marsh was a net source for TOC and TN during the drawdown of the marsh when water was being transported back to the Bear River. The study found that on average the marsh removed more than 1,000 Kg of TP per day. Lamarra (1980) found that reduced water velocities and presence of emergent vegetation caused sediment to settle, reducing TP and TSS loads to Bear Lake by as much as 50 percent. Lamarra et al. (1983) also included plankton uptake as a factor in the reduction of TP.
Studies by Herron (1985) and Bjornn et al. (1989) focused on Dingle Marsh, measuring inflows and outflows to determine its role as a net sink of suspended solids and nutrients. In 1981, TP loads were 13% greater in the inflow compared to outflow and in 1982, TP loads were 52% greater in the inflow compared to the outflow (Herron 1985). From 1985-86 Dingle Marsh trapped approximately 70% of total suspended solids (TSS), 16% of total phosphorus (TP) and 44% of nitrate (NO₃) that was transported into the marsh via the Inlet (Bjornn et al. 1989). Bjornn’s study focused on the effect of suspended solid and nutrient loading on wildlife production. He estimated that over the next 100 years approximately 1.5 to 2.6 million tons of suspended solids could be delivered to the Dingle Marsh and that the deposition of those suspended solids could lead to a filling up of 2.3 to 3.9% of the total volume of the marsh.

The movement and management of water through Dingle Marsh have a significant effect on nutrient and suspended solid retention in the system. Lamarra (1980) pointed out that the original goal of water management was to move water efficiently through Dingle Marsh. Breaches in canals have resulted in more dispersed water movement across the wetland and have further reduced Bear River loading into Bear Lake.

These past studies done on Dingle Marsh indicate that nutrient and suspended solid fluxes in the system are highly variable and dynamic. Bjornn et al. 1989, Herron 1985, Lamarra et al. 1983, Lamarra 1980, and USEPA 1975 all depended on grab samples collected at two week to one month intervals. Their findings indicate that the source/sink behaviors of the marsh are dynamic and apparently linked to the management of flow through the system. They found similar patterns of nutrients and suspended
solids transport, but the magnitude and durations of these patterns varied. They attributed the variability to the hydrology of the system, which incorporates the influence of past water years as well as the current year. It is likely that the infrequent sampling in these studies could not capture the true dynamics of this system.

High frequency monitoring is the most reasonable methodology in capturing both the small and large scale changes in water quality. Lamarra (1997) used high frequency turbidity monitoring stations at the Inlet, Outlet, and Causeway sites. In addition to these sites four separate organizations collected water quality samples at other sites within Dingle Marsh and Bear Lake. Lamarra only evaluated suspended solid transport and found similar patterns as previous studies. Throughout the year as water passed through Dingle Marsh from the Inlet, 75% of the turbidity and TSS was removed. The marsh fluctuated between trapping and releasing suspended solids throughout the year. The majority of TSS export from Dingle Marsh occurred when water was being drawn from Bear Lake and from Dingle Marsh. Discharge in 1997 was two times higher than the 75 year average (1922-1997) which may explain the wide range of values observed that year.

This study builds on the past understanding of dynamic changes in suspended solid and nutrient transport through the Dingle Marsh system. Use of high frequency turbidity monitoring at the Dingle Marsh site has allowed for the analysis of fine scale changes in water quality as well as providing more accurate annual load estimates of TP and TSS. Turbidity monitoring is easier and more cost effective than collecting water quality samples at high frequency. Turbidity monitoring for use as a surrogate is also more accurate than monthly samples using a linear interpolation method (Ankcorn 2003).
High frequency water quality monitoring can also provide clearer insight in a system such as Dingle Marsh that is influenced by the variable hydrology and management strategies. It is also more accurate than other methods in a system that may experience flashy flows or is influenced by storm events. The limitations and challenges of high frequency turbidity monitoring for use as a surrogate can be overcome by precise water chemistry sampling techniques, and by continuous calibration of the surrogate relationships. This study will further examine the loading characteristics of Dingle Marsh on a variety of temporal scales and determine the current filtering capacity of the system.
CHAPTER 3
STUDY SITE & METHODS

STUDY SITE

This study focused on monitoring the water quality at the four main water control structures of the Dingle Marsh complex: the Inlet, Causeway, Lifton, and the Outlet (Figure 2). Each site is operated independently of the other structures, but flow for all sites is managed by PacifiCorp. Because of the unique characteristics, each site’s hydrology is described below and shown in Figures 3-1, 3-2, 3-3, and 3-4.

The Inlet is the site where the Bear River is directed into Dingle Marsh. Under normal operating conditions, the Inlet site has flow every day of the year. The peak discharges at the Inlet usually occur in April, May or June with the most common peak discharges occurring in May. The peak discharges are the result of spring snow melt occurring upstream of the Inlet site (Figure 3-1).

Once water passes through the Inlet site, it flows into Dingle Marsh and either travels to the Outlet site or to the Causeway site. The Causeway is the intermediary site between Dingle Marsh and Bear Lake. The Causeway is used as the primary inflow site for water entering Bear Lake from the Bear River. The Causeway is generally operational from fall to mid-summer of the following year. Once water is needed downstream of Bear Lake, the Causeway is closed to allow water to flow from the Inlet to the Outlet (Figure 3-2).

Once the water demands downstream of Dingle Marsh exceed the outflow of the system, the Lifton site pumps water out of Bear Lake. The peak discharges of the Lifton site usually occur in July or August. Lifton is currently the only outflow of Bear Lake,
and is typically only used when more water is needed downstream or enacted in anticipation for the need to use Bear Lake for spring flood control (Figure 3-3).

The Outlet is the site which water must pass in order to be returned to the original Bear River channel. The peak discharges at the Outlet usually occur in July or August depending on the demand for water downstream. The Outlet receives flow from both the Inlet and Lifton sites, but is most influenced by flows at the Lifton site (Figure 3-4).

METHODS

All sampling occurred during the 2008 water year (Oct. 2007 – Sept. 2008).

Turbidity:

In order to capture turbidity values at high frequency, a turbidity sensor (Forest Technology Systems DTS-12 SDI-12) was installed at each of the four monitoring sites. The sensors use a laser based 90 degree nephelometer reading and are able to cover a range of 0 to 1600 NTU (Forest Technology Systems 2007). The sensors are factory calibrated and did not require recalibration during the study. The probes were cleaned and maintained with each monitoring event (one to two week frequency). The sensors are equipped with a motorized wiper blade that sweeps away debris and cleans the lens prior to taking a measurement. Every 30 minutes the sensor is programmed to calculate mean, variance, median and minimum turbidity values of 100 readings taken over a period of five seconds (Forest Technology Systems, 2007). In addition, these sensors have the ability to record water temperature, which was also recorded in 30 minute intervals. Turbidity sensors were housed at the end of 6-inch PVC tubing to protect the sensors from debris during high flow conditions.
Figure 3-1: Daily discharge averages at the Inlet for 1922-2008.

Figure 3-2: Daily discharge averages at the Causeway for 1997-2008.

Figure 3-3: Daily discharge averages at Lifton for 1997-2008.

Figure 3-4: Daily discharge averages at the Outlet for 1922-2008.
The end of the pipe was perforated to allow for adequate flow through, and the bottom of the pipe was open to prevent buildup of fine sediment. The sensor housings were vertically anchored to the side walls of the water control structures. This meant the sensors were located on one side of the channel rather than the middle of the flow. This had to be done to minimize damage done to the sensor housings by floating debris. The turbidity sensors were located approximately two feet from the bottom of the channel.

_Dataloggers and power sources:_

All sensors were powered by a solar panel and a rechargeable six-volt battery. Data were stored on site by Campbell Scientific CR200X data loggers and data were downloaded daily by way of radio telemetry to a local station within the Bear Lake NWR where data could be remotely accessed through an internet connection. Data loggers and batteries were sealed in enclosures which were connected to the sensor through conduit tubing.

_Water Quality Sampling:_

Water quality subsurface grab samples were collected at each site on a weekly basis except when discharge was near peak flows, when samples were collected two or three times per week. Grab samples were collected at the surface as close as possible to the turbidity sensor location. Immediately after samples were collected, portions of the water samples were filtered through a 0.7 μm glass fiber filter for analysis of the dissolved parameters. All samples were then placed on ice in a dark cooler until they could be placed in a 0°F freezer later that same day.
During the early part of the study (October to December), three samples were collected and were processed at the Utah Veterinary Diagnostic Laboratory (http://www.usu.edu/uvdl/). Sampling was limited in October through December due to ice cover and no grab samples were collected from December through April because all sites were covered with thick ice. For the remainder of the study, grab samples were consistently collected at a minimum interval of two weeks and were processed by the Baker Analytical Labs at Utah State University (http://www.biology.usu.edu/htm/labsites/baker-lab/). All samples collected were analyzed for total phosphorus (TP), dissolved total phosphorus (DTP), total suspended solids (TSS), total nitrogen (TN), dissolved total nitrogen (DTN), and nitrate (NO$_3$). TSS was analyzed using EPA Method 340.2 (mass balance) and TP and DTP were analyzed using EPA Method 365.2 (Ascorbic Acid). DTP was analyzed from the filtered samples. NO$_3$ was analyzed using EPA Method 353.2. TN and DTN were analyzed using EPA Method 351.2 on unfiltered and filtered samples respectively.

During each sampling trip, a set of duplicates, blanks, and spikes were taken at a single site chosen by scheduled rotation. The greatest variation in duplicates was 2.6%; all blanks were below detection limits (TP 0.0025 mg/L, TN 0.0057 mg/L, NO$_3$ 0.0009 mg/L) and spike recoveries ranged from 99% to 108%.

This study used discharge estimates provided by PacifiCorp (C. Baldwin, personal communication, December 30, 2009). PacifiCorp recalibrates any of its discharge relationships anytime there is an alteration to hydraulic head, canal bed elevations, or improvements made to water control structures. Methods for calculating discharges differed at each site and are described below.
At the Inlet site, PacifiCorp has established a stage-discharge relationship for estimating discharge. PacifiCorp considers this relationship to be most accurate when flows exceed 300 cfs, but considered estimates adequate for discharges less than 300 cfs. PacifiCorp later confirms these estimates by comparing discharge estimates against elevation changes in Mud Lake and in Bear Lake.

At the Causeway, PacifiCorp has established a relationship between the level in which the gates are open and the amount of water passing through. When the Causeway was re-built in 1993, the contractors provided PacifiCorp with a gate-level to discharge relationship. This relationship is re-calibrated on a yearly basis.

PacifiCorp estimates discharge at Lifton using a relationship between the number of pumps operating and discharge measurements downstream of the pumping station. This relationship can be complicated by the water levels and backflow from Dingle Marsh. The estimation of discharge at Lifton is reviewed by comparing the discharge levels at Lifton with discharges at the Outlet site.

The Outlet site discharge is based on stage-discharge relationships. The relationships at this site are in need of regular calibration due to the sedimentation occurring within the Outlet canal which can alter the shape of the canal and ultimately change the relationship between stage and discharge. In light of the confounding factor of sedimentation, PacifiCorp performs regular discharge measurements for approximately every 200 cfs change in discharge.

Site specific linear regression models were developed between turbidity and response parameters of interest using Statistical Analysis Software (SAS). Each model was tested for the significant ability to explain the variance in the parameter of interest
and also to analyze the model fit. Models were determined to be significant if they could explain up to 50% of the variability in the data and were found to be statistically significant (p-value \( \leq 0.05 \)). The linear regression models from each site were further tested in an analysis of covariance to test the need for site specific relationships.

For parameters with non-significant regressions, concentrations were determined by linear interpolation between sample values. No attempt was made to determine the accuracy of these estimates. Models found to be statistically sound were applied to the turbidity data to estimate parameter concentrations, resulting in a data set of modeled concentrations at 30 minute intervals at each site. Interpolated data was also calculated at a 30 minute interval. These high frequency concentrations coupled with the flow data provided by PacifiCorp provided loading estimates using Equation 1:

EQN 1. \[
L_{avg} = \frac{\sum_{i=1}^{n} Q_i C_i}{n}
\]

where \( L_{avg} \) = Average pollutant load for a time period

\( Q_i \) and \( C_i \) = Paired observations of flow and concentration

\( n \) = Number of instantaneous flow/concentration pairs.

To analyze the effect of sampling frequency on total load determination, a random sampling platform was programmed using SAS. This random sampler compiled summary statistics on 5,000 random samples taken from the complete set of calculated loads for TSS and TP. The program took random samples from the complete data set at the 30 minute, hourly, daily, weekly, and monthly time scales.
CHAPTER 4
RESULTS

DISCHARGE

The overall discharge during the 2008 was only 77% of mean historic discharges, however, patterns of seasonal flow closely resembled historic flows. At the Inlet site, a total of 164,000 acre-feet of water passed through during the 2008 water year. At the Inlet, the average discharge was 225 cfs, with the maximum discharge totaling 1,000 cfs and the minimum discharge totaling 25 cfs (Figure 4-1). The 2008 monthly average discharge was similar to historic data collected at this site; however, in 2008 the peak discharge occurred approximately a month later than the average (Figure 4-2).

Figure 4-1: Inlet hydrograph for the 2008 water year.

Figure 4-2: Comparison of historic (1922-2008) Inlet water years to the 2008 water year. For comparison, both curves in the lower frame show monthly average values.
During the 2008 water year, a total of 126,000 acre-feet of water was pumped through Lifton. Lifton had a maximum discharge of 1,248 cfs, a minimum discharge of 0 cfs, and an annual average of 174 cfs. There were two peaks of maximum discharge during the 2008 water year; one occurred during mid July 2008 and the other during mid August 2008. Water was pumped from Bear Lake from July 2008 to the end of September 2008 (Figure 6-1). Compared to historical flows (1997-2008), discharge at Lifton was lower than average in the early part of May and June, but rose quickly to peak at discharges above average in August (Figure 6-2). A total of 164,000 acre-feet passed through the Outlet during the 2008 water year. The Outlet had a maximum discharge of 1,476 cfs, a minimum discharge of 0 cfs and an annual average of 225 cfs (Figure 7-1).
The maximum discharge for the 2008 water year at the Outlet occurred during mid-July 2008 with another significant peak in mid-August 2008.  The flow patterns at the Outlet closely resembled historic flows (Figure 7-2) and flows at the Lifton site.

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Flow control points in the Dingle Marsh system resulted in three main water management strategies for the 2008 water year (Figures 8-1, 8-2, and 8-3). During low flows (fall through winter), flow was diverted from the Bear River, moving through Dingle Marsh and through the Causeway into Bear Lake. The Lake Fill period occurred during spring runoff, when high flows from the Bear River were diverted into Dingle Marsh and on to Bear Lake. During the Lake Withdrawal period, water was pumped at the Lifton station from Bear Lake into the Outlet canal. Low Bear River flows moved directly from the Inlet to the Outlet canal. The absolute timing of water management will vary between years according to existing storage in Dingle Marsh, Bear Lake runoff flows, and irrigation demands.
Figure 8: Schematic of three major periods of different water management in the Dingle Marsh system. The arrows represent flow direction and the width of the arrows represent the relative average flows for the time period specified.
Figure 9-1 shows discharge at all four monitoring sites during the 2008 water year. The three flow management strategies are indicated at the top of the graph. Figure 9-2 compares discharge between the Inlet and the Outlet.

Figure 9-1: Continuous discharge for all sites during the 2008 water year.

Figure 9-2: Comparison of Inlet & Outlet discharges.

Figure 9-3: Comparison of Causeway & Lifton discharges.
The difference between these total flows is the annual net change of Bear River water in the Dingle Marsh/Bear Lake system. Figure 9-3 compares discharge at the Lifton and Causeway sites. The difference between these total flows is the change in the volume of Bear Lake during the 2008 water year.

For the 2008 water year, 164,000 acre-feet of water passed through both the Inlet and Outlet stations. 126,000 acre-feet passed through both the Causeway and Lifton sites during the same period (Figure 10).

![Total Discharge for Each Site during the 2008 Water Year](image)

**Figure 10:** Total discharge for all sites during the 2008 water year.

**TOTAL SUSPENDED SOLIDS**

The relationships between turbidity and TSS measured by grab samples taken during the study are shown below. The data for the Inlet and Outlet are shown in Figure 11-1 and for the Causeway and Lifton in figure 11-2. Discharge versus TSS at all sites is shown in Figure 12.
Turbidity was a significant predictor of TSS at all sites within the study area (Table 1, Figure 11-1 and Figure 11-2). At three of the sites (Inlet, Causeway, and Outlet) turbidity alone accounted for 81% to 94% of the variability seen in TSS (Table 1). Hybrid models were also tested at all sites, adding discharge as a second predictor. At all but the Lifton site, turbidity and discharge were found to be collinear and therefore the models were rejected. At the Lifton site, the hybrid model was significant and was the most powerful model. Discharge alone was also a significant predictor of TSS at all four sites, but did not explain as much variability as the TSS or hybrid models. However, the discharge models are useful in comparing the range of TSS concentrations across the spectrum of discharges (Figure 12). The decreasing levels of TSS as flows move from the Inlet to the other sites illustrates some of the settling capacity of the marsh. The models chosen to calculate TSS concentrations for further analysis in this study are indicated in Table 1 with a (1).
In an analysis of covariance, the slopes of the TSS versus Turbidity regression lines were found to be significantly different $F_{3,44} = 2.77, P = 0.052$. This result reinforces the need for site specific regression models to accurately predict TSS. The predicted concentrations of TSS for each sampling date are shown for each site in Figure 13. As suggested by the $R^2$ for these relationships, predicted values varied more for the Causeway and Lifton sites than the Inlet and Outlet. TSS concentrations at the Causeway and Lifton were on average less than half of the concentrations observed at the Inlet and Outlet.

The most powerful predictive model at each site was used to estimate concentrations of TSS for each of the turbidity and/or discharge measurements taken throughout the study. This produced a dataset of TSS concentrations at 30 minute intervals (Figure 14).
Table 1: TSS predictive models. Units: TSS (mg/L), Turbidity (NTU), Discharge (cfs). The (1) denotes the model used at each site to calculate TSS.

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<tr>
<th>Sites</th>
<th>TSS Model</th>
<th>R²</th>
<th>Model fit</th>
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<tr>
<td>Inlet</td>
<td>(1) TSS = 1.2058*Turbidity – 2.2291</td>
<td>0.94</td>
<td>P &lt;0.0001</td>
</tr>
<tr>
<td></td>
<td>TSS = 0.1098*Discharge + 10.003</td>
<td>0.81</td>
<td>P &lt;0.0001</td>
</tr>
<tr>
<td>Causeway</td>
<td>(1) TSS = 0.809*Turbidity – 0.9273</td>
<td>0.83</td>
<td>P = 0.0001</td>
</tr>
<tr>
<td></td>
<td>TSS = 0.0096*Discharge + 3.2908</td>
<td>0.42</td>
<td>P = 0.0428</td>
</tr>
<tr>
<td>Lifton</td>
<td>(1) TSS = 0.707<em>Turbidity + 0.0065</em>Discharge – 0.54719</td>
<td>0.70</td>
<td>P = 0.0035</td>
</tr>
<tr>
<td></td>
<td>TSS = 0.0076*Discharge + 5.6673</td>
<td>0.51</td>
<td>P = 0.0129</td>
</tr>
<tr>
<td></td>
<td>TSS = 0.8439*Turbidity +2.8228</td>
<td>0.41</td>
<td>P = 0.0349</td>
</tr>
<tr>
<td>Outlet</td>
<td>(1) TSS = 0.9462*Turbidity + 0.0595</td>
<td>0.93</td>
<td>P&lt;0.0001</td>
</tr>
<tr>
<td></td>
<td>TSS = 0.0388*Discharge + 3.1937</td>
<td>0.79</td>
<td>P&lt;0.0001</td>
</tr>
</tbody>
</table>

Figure 12: TSS versus Discharge at all sites.
Figure 13: Comparisons between modeled predictions and TSS concentrations from grab samples.
Throughout most of the spring, TSS concentrations at the Causeway were two to three times lower than those at the Inlet. From mid May until mid June, concentrations at the Causeway were up to ten times lower than those at the Inlet. Since there is no significant increase in flow from the Inlet to the Causeway, the reductions in concentrations appear to result from settling due to the diffuse flow through Dingle Marsh and especially through Mud Lake.

The Outlet was opened in mid-June, and until about mid July, patterns of TSS concentrations at the Outlet mirrored the concentrations at the Inlet, but at lower absolute concentrations. This pattern shifted in late July, corresponding to reduced Inlet flows and the initiation of pumping at Lifton. By early August, the Outlet concentrations exceeded those at the Inlet site. As higher flows eased, concentrations at all sites were low and patterns were similar (Figure 14).

TSS loading through Dingle Marsh was driven mostly by seasonal conditions leading to high variability. During winter baseflow conditions, TSS loads at all sites were very low and stable (Figure 15). The highest TSS loads, both in yearly load and instantaneous peak, occurred at the Inlet. As TSS loading at the Inlet increased with spring flows, these same flows did not cause a similar spike in TSS loading at the Causeway site. As flows moved from the Inlet to the Causeway across Dingle Marsh during this period, TSS loading was greatly reduced due to the diffuse flow and resulting lower velocities of water moving through Dingle Marsh thus resulting in increased settling of solids. When flow was closed at the Causeway and the Outlet gates were opened, TSS loads at the Outlet were also reduced compared to loads at the Inlet; although the differences were not as substantial as those between the Inlet and Causeway.
The path from the Inlet to the Outlet is more constrained within a channel and flow velocities remain higher than they are when flow moves to the Causeway, resulting in reduced settling capacity.

Although loading at the Outlet was initially lower than the Inlet, TSS loads at the Outlet continued to increase until they exceeded loads at the Inlet and Lifton combined (September 2008). For the months of July and August, TSS loading at the Outlet was on average 32 MT/day higher than loads at the Inlet and Lifton combined. Since this study only collected data at the four monitoring sites, the source of this excess TSS load could not be precisely identified. As TSS loads declined at the Inlet during late summer, the loads at the Outlet appeared to be most influenced by the Lifton site loads. From August through September, Lifton loads were quite variable. The Outlet loading pattern is smoother and does not reflect the variability in Lifton loading (Figures 15 & 16).

During the 2008 water year, a total of 13,600 metric tons (MT) of TSS passed the Inlet monitoring site (Figure 16). The assumption in this study is that this was all delivered to Dingle Marsh without any short circuiting or loss to other areas. An additional 3,010 MT of TSS was added to Dingle Marsh from Bear Lake through the Lifton Pumping Station. 1,200 MT of TSS were exported from the marsh into Bear Lake, and nearly 8,000 MT of TSS were exported through the Outlet and back into the Bear River. During the baseflow period, TSS loading into Dingle Marsh and into Bear Lake remained relatively low (Figure 17-1). As flows began to increase in the lake fill period, the TSS loading reached the highest levels observed throughout the study, but loading into Bear Lake remained low (Figure 17-2). During the lake withdrawal period, Dingle Marsh exported more TSS than it received (Figure 17-3).
Figure 14: Predicted continuous total suspended solids (TSS) concentrations for all sites.

Figure 15: Predicted continuous total suspended solids (TSS) loads for all sites.

Figure 16: Yearly TSS loading at all sites.
Figure 17: Schematics of the seasonal dynamics of TSS loading during the three main water management strategies. The arrows represent flow direction and the total width of the arrows represent the relative average TSS loads for the time period specified.
PHOSPHORUS

The linear relationships between turbidity and TP measured in grab samples taken during the study are shown below. The data for the Inlet and Outlet are shown in Figure 18-1 and for the Causeway and Lifton in Figure 18-2.

![TP vs Turbidity Relationships at Inlet & Outlet Sites](image)

Figure 18-1: TP versus turbidity relationships for the Inlet and Outlet sites.

Turbidity was the best predictor for TP at both the Inlet and Outlet sites, accounting for over 60% of the variation in TP (Table 2). At the Causeway site, turbidity was also a significant predictor of TP, but discharge proved to be a more powerful predictor, accounting for 82% of the variation in TP. Lifton TP concentrations could not be predicted by turbidity or discharge alone or by a hybrid model.
Figure 18-2: TP versus Turbidity relationships for the Causeway and Lifton sites.

Figure 19: TP versus discharge relationships for all sites.
Table 2: Total phosphorus (TP) predictive models. Units: TP(mg/L), turbidity(NTU), discharge(cfs). The (1) denotes the model used at each site to calculate TP.

<table>
<thead>
<tr>
<th>Sites</th>
<th>TP Model</th>
<th>R²</th>
<th>Model fit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inlet</td>
<td>(1) TP = 9.8*10^{-4}*Turbidity + 0.024</td>
<td>0.63</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td></td>
<td>TP = 7.9*10^{-5}*Discharge + 0.045</td>
<td>0.43</td>
<td>0.0032</td>
</tr>
<tr>
<td>Causeway</td>
<td>(1) TP = 0.0051*Discharge + 0.0081</td>
<td>0.82</td>
<td>0.0020</td>
</tr>
<tr>
<td></td>
<td>TP = 0.0014*Turbidity + 0.014</td>
<td>0.77</td>
<td>0.0087</td>
</tr>
<tr>
<td>Lifton</td>
<td>TP = 5.3*10^{-6}<em>Discharge + 9.1</em>10^{-4}</td>
<td>0.12</td>
<td>0.3674</td>
</tr>
<tr>
<td></td>
<td>TP = 4.8*10^{-4}*Turbidity + 0.0085</td>
<td>0.09</td>
<td>0.4455</td>
</tr>
<tr>
<td>Outlet</td>
<td>(1) TP = 7.6*10^{-4}*Turbidity + 0.016</td>
<td>0.67</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td></td>
<td>TP = 5.4*10^{-4}*Discharge + 0.015</td>
<td>0.57</td>
<td>0.0015</td>
</tr>
</tbody>
</table>

I also tested the significance of relationships between particulate phosphorus (the difference between TP and DTP at 30 minute increments) and turbidity or discharge, reasoning that the dissolved phosphorus component of the total phosphorus concentration may not be well correlated with either turbidity or discharge. All significant models of the relationship between particulate phosphorus and turbidity or discharge are shown in Table 3. None of these models had greater explanatory power than models with TP alone as the response variable. Predictive models for dissolved total phosphorus were also evaluated and none were found to be significant.

The predicted concentrations of TP for each sampling date using the best regression models for each site and parameter were compared with the grab sample data to observe the accuracy of the models (Figures 20). As suggested by the R² value, the best relationships are at the Causeway and Outlet sites. At the Inlet site, the relationship between sampled and predicted becomes less accurate as flows and concentrations peaked in June and July.
Table 3: Particulate Phosphorus Predictive Models. Units: PartP(mg/L), Turbidity(NTU), Discharge(cfs).

<table>
<thead>
<tr>
<th>Sites</th>
<th>Particulate Phosphorus Model</th>
<th>R²</th>
<th>Model fit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inlet</td>
<td>PartP = 0.00053*Turbidity + 0.021</td>
<td>0.22</td>
<td>0.0356</td>
</tr>
<tr>
<td></td>
<td>PartP = 0.000038*Discharge + 0.035</td>
<td>0.13</td>
<td>0.1624</td>
</tr>
<tr>
<td>Causeway</td>
<td>PartP = 0.000022*Discharge – 0.0025</td>
<td>0.73</td>
<td>0.0144</td>
</tr>
<tr>
<td></td>
<td>PartP = 0.00092*Turbidity + 0.0027</td>
<td>0.53</td>
<td>0.0258</td>
</tr>
<tr>
<td>Lifton</td>
<td>PartP = 0.000033*Discharge + 0.0030</td>
<td>0.11</td>
<td>0.3877</td>
</tr>
<tr>
<td></td>
<td>PartP = 0.00025*Turbidity + 0.0031</td>
<td>0.06</td>
<td>0.5373</td>
</tr>
<tr>
<td>Outlet</td>
<td>PartP = 0.00071*Turbidity + 0.0016</td>
<td>0.55</td>
<td>0.0016</td>
</tr>
<tr>
<td></td>
<td>PartP = 0.000036*Discharge – 0.0038</td>
<td>0.57</td>
<td>0.0018</td>
</tr>
</tbody>
</table>

TP concentrations calculated at 30 minute intervals using the phosphorus models are shown in Figure 21. The patterns of TP concentrations at the four monitoring sites were very similar to those seen for TSS (Figure 14). A pattern of increasing TP concentrations was observed with increasing discharges. The highest TP concentration (0.18 mg/L) occurred at the Inlet site in early June during the peak of runoff. A second peak in TP concentrations came with the higher elevation runoff which typically occurs near mid-summer in this region. TP concentrations were highest at the Inlet site during most of the year. As TP moved across Dingle Marsh from the Inlet to the Causeway, it was reduced up to 50% throughout the year. This same pattern was observed with TSS concentrations and can also be attributed to settling within Mud Lake. Similar to the TSS concentrations, TP concentrations were also lower at the Outlet site compared to the Inlet site during the high flow periods. Since the TP concentrations at Lifton were consistently low, the pumping later in the year did not seem to have much of an effect on
concentrations at the Outlet. As flows were reduced towards the end of summer, the TP concentrations at the Inlet and the Outlet were nearly identical (Figure 21).

Figure 20: Comparisons between modeled predictions and TP concentrations from grab samples.
Figure 21: Predicted continuous total phosphorus (TP) concentrations for all sites.

Calculated TP loads for each site are shown in Figure 22. The Inlet had the highest total TP loading followed by the Inlet. TP loading ramped up during spring runoff flows and remained responsive to flows throughout the year. The highest peaks in TP loading occurred at the Inlet and corresponded to the peak during spring runoff discharge and the peak in discharge which was related to the higher elevation snowmelt. TP loading at the Causeway tracked changes in loading at the Inlet, but the magnitudes were much lower at the Causeway.

As with TSS loads, once the Outlet was opened and the Causeway closed, TP loads at the Outlet increased dramatically and subsequently dropped as Inlet loads declined rapidly. Initiation of releases from Lifton did not appear to affect TP (or TSS) loads in late July. By August, however, TP loads at the Outlet increased and appeared to reflect Lifton loads, as well as additional loading as the flow moved from Bear Lake to the Outlet site. Once TP loading at the Outlet reached these levels, it stayed consistently higher than both the Inlet and Lifton combined for the rest of the water year (Figure 22 & 24).
Figure 22: Continuous TP loading at all sites. Predicted continuous total phosphorus (TP) loads for all sites.

During the 2008 water year, a total of 16,300 Kg of TP passed through the Inlet site, and an additional 2,400 Kg was added from Bear Lake through the Lifton Pumping station. 4,600 Kg of TP was exported to Bear Lake through the Causeway, and 10,000 Kg of TP passed through the Outlet back into the Bear River. Annual TP loads between the Outlet and the Causeway declined by 62%, compared to the 88% decline observed for TSS loads.

TP loading during the baseflow period remained low and steady during this study (Figure 23-1). However, as flows increased due to spring runoff, loading of TP from the Bear River spiked (Figure 23-2). During the lake withdrawal period, inputs of TP into Dingle Marsh were low, but the marsh became a net source of TP during this period (Figure 23-3).
Figure 23: The seasonal dynamics of TP loading during the three main water management strategies. The arrows represent flow direction and the total width of the arrows represents the relative average TP loading for the time period specified.
Figure 24: Annual total phosphorus loads at each sampling station.

NITROGEN

No significant predictive models were found for TN or nitrate. Linear interpolation between grab samples was used to estimate concentrations throughout the sampling period and loads were then calculated using continuous discharge measurements and interpolated concentrations. These estimates in this section, therefore, are not based on models with significant predictive power. The loading estimates made here for TN and nitrate are subject to a greater level of error compared to the modeled parameters. Only TN and nitrate data from grab samples collected from May through October 2008 were used in this analysis.
Figure 25 shows estimated concentrations as well as the concentrations of each analyzed sample. The highest observed concentration of TN occurred at the Lifton site in late August, when the concentration exceeded 1.0 mg/L. TN was the only estimated parameter with a higher concentration at a site other than the Inlet.

![Figure 25: Estimated TN concentrations at all sites.](image)

Estimated TN loads leaving Bear Lake at Lifton exceeded all other loads during most of the period of Lifton pumping, with the highest peak of TN loading in late August. However, during the period when samples were collected (May – October), the lake was not a net source of TN to the Bear River, as the annual TN load at the Inlet site exceeded the annual load at Lifton (Inlet 119,000 Kg, Lifton 93,000 Kg). During the 80 days of pumping from Bear Lake at Lifton, however, Bear Lake exported an estimated 20,000 Kg more TN than it received. As flows declined at the Inlet site in late July, TN loads at Lifton were nearly double those at the Outlet. The flows of Lifton and the Outlet were nearly identical, indicating a large amount of TN was removed as water passed down the outlet canal towards the Outlet site (Figures 26 & 27).
Between May and October, a total of 119,000 Kg of TN was delivered to the Dingle Marsh from the Bear River through the Inlet, and an additional 93,000 Kg of TN was delivered from Bear Lake through the Lifton pumping station. Unlike patterns seen with other parameters in this study, TN loads did not decrease as water moved from the Inlet to the Causeway. This would indicate that the Dingle Marsh complex did not remove nitrogen as water moved from the Bear River to Bear Lake.
NITRATE-N

Nitrate-N (NO\textsubscript{3}) concentrations were less than 10% of TN concentrations and remained very low throughout the study. NO\textsubscript{3} concentrations were estimated through linear interpolation of grab sample data since no significant turbidity based models were found. The interpolation was based from 21 sampling events and is considerably less certain than the modeled concentrations of TP and TSS. In contrast to TN, almost all NO\textsubscript{3} was removed as water moved from the Inlet to the Causeway (Figure 28). When water moved from the Inlet to the Outlet directly, however, very little nitrate was removed. The changes in NO\textsubscript{3} concentrations further confirm the filtering potential of Dingle Marsh and Mud Lake compared to the results when flows are short circuited down the bypass canal (Figure 28). Concentrations of NO\textsubscript{3} leaving Bear Lake at Lifton were very low at all times.

Figure 28: Estimated NO\textsubscript{3} concentrations at all sites.

Through the 2008 water year, a total of 6,600 Kg of NO\textsubscript{3} was delivered to the Dingle Marsh from the Bear River through the Inlet, with only 300 Kg of that being delivered to Bear Lake (Figure 29). Only 200 Kg were exported from Bear Lake through
the Lifton pumping station during the same period with 1,000 Kg being exported through the Outlet. Dingle Marsh appears to have removed almost all the nitrate load from the Bear River during the period when the lake was filling (Figure 30). When the Causeway was closed, nitrate loading at the outlet peaked, indicating relatively little processing of nitrate except in the Mud Lake complex of Dingle Marsh.

Figure 29: Estimated annual NO₃ loading at all sites. NOTE: these estimates are based on linear interpolation of grab samples.

Figure 30: Estimated NO₃ loading at all sites.
TEMPERATURE DIFFERENCES

Water temperatures varied slightly between the Inlet and Causeway (Figure 31-1). During most of the year, Causeway temperatures were slightly higher than inlet temperatures. Inlet temperatures were higher than Causeway temperatures for a brief period during spring runoff. Water temperatures at the Lifton station and the Outlet site were almost identical (Figure 31-2).

Figure 31-1: Continuous water temperatures at the Inlet & Causeway.

Figure 31-2: Continuous water temperatures at Lifton & the Outlet.
ANNUAL MASS BALANCES

Mass balances of TSS, TP, TN, and NO₃ were calculated for both Dingle Marsh and for Bear Lake (Table 4). Net changes in loads were calculated as follows:

EQN 2: Dingle Marsh annual net change = \( L_{\text{in}} + L_{\text{dif}} - L_{\text{cause}} - L_{\text{out}} \)

EQN 3: Bear Lake annual net change = \( L_{\text{cause}} - L_{\text{dif}} \).

EQN 4: Dingle Marsh retention of Bear River constituents = \( L_{\text{in}} - L_{\text{cause}} \).

Where \( L_{\text{in}} = \) Annual Inlet Load (MT)

\( L_{\text{dif}} = \) Annual Lifton Load (MT)

\( L_{\text{cause}} = \) Annual Causeway Load

\( L_{\text{out}} = \) Annual Outlet Load.

Note that the Bear Lake mass balance did not include other tributaries to the lake.

During the 2008 water year, a net gain was measured in Dingle Marsh for TSS, TP, TN and NO₃ (Table 4). In contrast, Bear Lake experienced a net gain in TP and NO₃ and a net loss in TSS and TN.

Table 4: Dingle Marsh and Bear Lake Mass Balance for 2008 water year, using equations 2 and 3 respectively.

<table>
<thead>
<tr>
<th>Sites</th>
<th>TSS Loading</th>
<th>TP Loading</th>
<th>TN Loading</th>
<th>NO₃ Loading</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dingle Marsh</td>
<td>7,700 MT</td>
<td>4,100 Kg</td>
<td>52,700 Kg</td>
<td>5,500 Kg</td>
</tr>
<tr>
<td>Bear Lake</td>
<td>-1,800 MT</td>
<td>2,200 Kg</td>
<td>-19,900 Kg</td>
<td>80 Kg</td>
</tr>
</tbody>
</table>

The load of every parameter was reduced as water moved from the Bear River into Bear Lake (Table 5).
Table 5: Dingle Marsh filter effect of Bear River water delivered to Bear Lake.

<table>
<thead>
<tr>
<th></th>
<th>TSS Loading</th>
<th>TP Loading</th>
<th>TN Loading</th>
<th>NO3 Loading</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Delivered From Bear River through Inlet</strong></td>
<td>9,800 MT</td>
<td>11,900 Kg</td>
<td>90,200 Kg</td>
<td>5,800 Kg</td>
</tr>
<tr>
<td><strong>Deposited In Bear Lake</strong></td>
<td>1,200 MT</td>
<td>4,600 Kg</td>
<td>73,000 Kg</td>
<td>300 Kg</td>
</tr>
<tr>
<td><strong>% Filtered By Dingle Marsh (EQN 4)</strong></td>
<td>88%</td>
<td>62%</td>
<td>20%</td>
<td>95%</td>
</tr>
</tbody>
</table>

SEASONAL CHANGES

In order to better understand changes in TSS and nutrient loading throughout the water year, mass balances were generated using modeled load estimates calculated at 30 minute intervals. Mass balances were calculated on a 30 minute time step using the same equations as used for the annual mass balances. Percent captured was also calculated and graphed. These values were determined by using equation four calculated at a daily time step, then analyzing the filtered loads as a percentage.

DINGLE MARSH MASS BALANCE

The highest TSS loading for Dingle Marsh occurred during Bear River’s spring runoff from May to July, peaking at 300 MT per day at the end of May (Figure 32). During this period, nearly 9,000 MT of TSS passed through the Inlet, but only 1,400 MT was recorded at the Causeway indicating that Dingle Marsh retained approximately 84% of the TSS bound for Bear Lake during the high flow. During baseflow conditions,
Dingle Marsh trapped an estimated 58% of the 1,000 MT of TSS that entered through the Inlet.

Once the lake withdrawal period started, Dingle Marsh began to export TSS (Figure 32). Forty percent more TSS was exported through the Outlet than delivered from the Inlet and the Lifton site combined, indicating the marsh was releasing TSS that had previously been settled. Even including this export, Dingle Marsh retained 49% of the TSS delivered during the 2008 water year (Figure 33).

The TSS loading to Dingle Marsh is closely correlated to the hydrograph and the TSS loading at the Inlet. Once the Causeway is closed and the Outlet is open, the export of TSS from Dingle Marsh closely resembles the pattern at Lifton. These patterns suggest that the major influence to Dingle Marsh during runoff is the dynamics of the Inlet. During baseflow or drawdown of the system, the dynamics are more closely related to Lifton.

Figure 32: Dingle Marsh continuous TSS mass balance calculated using Eqn 2 at a daily time step. Values greater than zero indicate periods of accumulation in Dingle Marsh, and values less than zero indicate periods of export.
Figure 33: Percent TSS retained by Dingle Marsh calculated using Eqn 4, calculated at a daily time step.

The peak loading rate of TP to Dingle Marsh was approximately 300 Kg of TP per day which occurred at the beginning of June 2008 (Figure 34). During the 2008 water year, Dingle Marsh was able to retain about 29% of the TP that was delivered both through the Inlet and the Lifton sites (Figure 35).

During the time that the Causeway was open (22 Oct 2007 through 29 June 2008), 11,900 Kg of TP came through the Inlet canal, and 4,600 Kg of TP passed through the Causeway into Bear Lake (38% removal of TP by Dingle marsh).

Figure 34: Dingle Marsh continuous TP mass balance calculated using Eqn 2 at a daily time step. Values greater than the zero indicate periods of accumulation in Dingle Marsh, and values less than zero indicate periods of export.
Figure 35. Percent TP retained by Dingle Marsh, using Eqn 4, calculated at a daily time step.

TN loading to Dingle Marsh was relatively stable through most of the baseflow period with similar daily load estimates at the Inlet and Causeway sites (Figure 36). From April 2008 through June 2008, some TN was retained in the system and when the Lifton pumps were turned on in early July, the marsh retained even more of the TN released from Bear Lake. The high level of TN loading at Lifton was due to high TN concentrations coming out of Bear Lake (once exceeding 1.0 mg/L).

Figure 36: Dingle Marsh estimated TN mass balance calculated using Eqn 2 at a daily time step using less precise interpolated data. Values greater than the zero indicate periods of accumulation in Dingle Marsh, and values less than zero indicate periods of export.
NO₃, a significant bioavailable form of nitrogen, represented only 4% of the TN at the Inlet. The highest NO₃ concentrations were observed at the Inlet where they were nearly 10 times higher than any other site. When the Causeway was first closed, there was a short period when Dingle Marsh exported NO₃ to the Outlet. As water was routed from the Inlet to the Causeway nearly all NO₃ was trapped, but as water was routed from the Inlet directly to the Outlet nearly all the NO₃ was being transported through the system. During baseflow NO₃ concentrations (Figure 28) and loads (Figure 37) were low at all sites.

Figure 37: Dingle Marsh estimated NO₃ mass balance calculated using Eqn 2 at a daily time step using less precise interpolated data. Values greater than the zero indicate periods of accumulation in Dingle Marsh, and values less than zero indicate periods of export.

**BEAR LAKE MASS BALANCE**

Bear Lake has several small tributaries which were not monitored as part of this project. In this thesis, therefore, the Bear Lake mass balance refers only to the difference between the Causeway (Inputs) and the Lifton (Outputs).
In total during the 2008 water year, 1,200 MT of TSS entered into Bear Lake through the Causeway and 3,000 MT were exported from Bear Lake through Lifton. Therefore, Bear Lake exported 1,800 MT more TSS than it received from the Bear River. However, it should be noted the TSS that Bear Lake exported was much different that the TSS entering from the Causeway. Although the composition of the TSS was not explicitly analyzed during this study, field observations showed the Causeway TSS to be made up primarily of plant material, where the Lifton TSS was primarily made up of mineral particles.

Figure 38 shows the continuous TSS loading into Bear Lake throughout the 2008 water year. TSS loading into Bear Lake was highest between May and June, peaking at 57 MT per day on June 5. During this short period, 870 MT of the total 1,200 MT of TSS was delivered to Bear Lake.

![Figure 38: Bear Lake continuous TSS mass balance calculated daily using Eqn 3. Values greater than zero indicate periods of accumulation in Bear Lake, while values less than zero line indicated periods of export from the lake.](image)

During the lake fill/downstream release period (May – July), 8,900 MT of TSS passed through the Inlet site. When the Outlet site was opened on June 17, the loading at the Causeway was reduced to approximately 2 MT per day. The Causeway was closed
on June 29 and thus ended the loading into Bear Lake for the 2008 water year. Lifton was opened on July 5 and marked the beginning of TSS export from Bear Lake. The export of TSS from Lifton peaked at 130 MT per day on August 10 and averaged 37 MT per day during operation.

The patterns of TP loading to Bear Lake were similar to those for TSS, except that Bear Lake was a net sink for TP, receiving 4,600 Kg of TP in the 2008 water year and exporting 2,400 Kg. TP loading to Bear Lake through the Causeway peaked at 147 Kg/day on June 5. The overall average for TP loading to Bear Lake through the Causeway was 16 Kg/day. The average TP loading during the highest discharge (May-June) was 45Kg/day (Figure 39).

Lifton exported an average of 29 Kg of TP per day from Bear Lake with a peak of 68 Kg per day on August 10. The levels of TP loading from Lifton are more a function of high flows rather than high concentrations of TP.

During the time that the Causeway was open, TP loading was on average 30 MT/day higher at the Inlet than at the Causeway.

![Figure 39: Bear Lake continuous TP mass balance calculated daily using Eqn 3. Values greater than zero indicate periods of accumulation in Bear Lake, while values less than zero line indicated periods of export from the lake.](image)
Bear Lake exported more TN than it received during the 2008 water year (Figure 40). As with TSS and TP, TN loading into Bear Lake was minimal throughout the baseflow period and increased as flow increased with spring runoff. The significant decrease in TN loading to Bear Lake coincided with the Outlet opening in mid June. Concentrations of TN were similar between Lifton and the Causeway, and TN was present at similar concentrations at all monitoring stations, so differences in loadings were apparently due to changes in flow. TN concentrations, however, are based on interpolated values (from 21 sampling events) and are considerably less certain than the modeled TP and TSS concentrations.

![Bear Lake TN Loading for 2008 Water Year](image)

Figure 40: Bear Lake estimated TN mass balance. (Equation 3 calculated at a daily time step using less precise interpolated data). Loading above the zero line identify periods of accumulation in Bear Lake, whereas loading below the zero line identify times of export.

On average, as flow moved from the Inlet through Dingle Marsh and Mud Lake, Dingle Marsh retained almost all NO₃. On average, loads of NO₃ at the Causeway were only 8% of those at the Inlet site during this period. NO₃ loading at Bear Lake was never higher than 5 Kg/day and the export was never higher than 8 Kg/day. Both the low imports at the Causeway and low exports at Lifton are due to the low average NO₃.
concentrations at each of those sites (Causeway 0.0015 mg/L, Lifton 0.0013 mg/L) compared to an average 0.021 mg/L at the Inlet site. Even though loading of NO₃ in Bear Lake is relatively low, it is still possible to see the patterns in loading. The loading of NO₃ follows the hydrology of the Inlet for the most part, and again it can be seen that when the Outlet site is first opened (mid June); it immediately switched the balance to a net loss (Figure 41). As with TN, NO₃ concentrations were interpolated between 21 sampling events of actual measured concentrations.

![Bear Lake NO₃ Loading for 2008 Water Year](image)

Figure 41: Bear Lake estimated NO₃ mass balance (Equation 3 calculated at a daily time step using less precise interpolated data). Loading above the zero line identify periods of accumulation in Bear Lake, whereas loading below the zero line identify times of export.
LINEAR REGRESSION MODELS

Turbidity was significantly correlated to TSS at all four sites in this study. The strongest correlations were found at the Inlet site, which is most similar to a natural river system. The weakest turbidity to TSS relationships were observed at the Lifton site, which is the most artificially controlled site and influenced by the water chemistry of Bear Lake. The turbidity and TSS levels at Lifton were much lower than at the other sites resulting in larger impacts from the errors inherent in the methods used. Discharge was a significant predictor of TSS at all sites, although the predictive power was less than provided by turbidity.

Significant correlations between turbidity and TP were found at three of the four sites (Inlet, Causeway, and Outlet). These same three sites had significant correlations between discharge and TP. At the Lifton site, neither turbidity nor discharge was a significant predictor of TP. Turbidity was found to have the strongest predictive power at the Inlet and Outlet sites, and discharge had the strongest relationship at the Causeway. Neither turbidity nor discharge was found to be significant predictors of the other water quality parameters (TN, and NO₃).

When using turbidity as a surrogate measure, the need to account for seasonal fluxes within the models is a point often emphasized (Kronvang et. al 1997, Christensen et al., 2000, Jones 2008). Only one site (the Inlet) had flow throughout the entire year, therefore, seasonal variability at the other sites was not measured. The seasonality of discharge was tested as a possible explanatory variable for the Inlet site. Discrete
variables representing high and low discharge were introduced into the models at the Inlet, but no increase in the explained variance or model fit was observed.

TOTAL SUSPENDED SOLIDS MODELS

Significant models for predicting TSS as a function of turbidity were established at all sites in this study. ANCOVA results suggested that the regression slopes of the site-specific relationships were significantly different leading to the conclusion that TSS versus turbidity relationships needed to be developed at each site. These models were then used to estimate TSS concentrations for the entire water year at 30 minute intervals. The only site in which turbidity was not the most powerful predictor of TSS was the Lifton site, where the best fit was produced from a hybrid model using both discharge and turbidity as predictors. At this site, turbidity alone only explained 41% of the variability in TSS (Table 1) while the hybrid model was able to explain 70% of the variability. The differences in the Lifton model compared to the other sites may be related to the fact that this is the only site within the system in which water is directly pumped through the site. Water pulled through the pumps may draw in lake sediments, which may explain the relationship between flow and TSS at this site.

TOTAL PHOSPHORUS MODELS

Significant turbidity versus TP regressions were found at the Inlet, Outlet, and Causeway sites. ANCOVA results suggested the slopes and Y-intercepts of the three site-specific relationships were not significantly different. This results shows that an overall turbidity versus TP model could statistically be used in this system. However, since the site specific models were only able to account for 60% to 80% of the variability
of TP, a general overall model would only increase the variability in the predictions. There is already a considerable tradeoff between high frequency sampling that the turbidity sensors provide, with the lack of accuracy in the turbidity to TP relationships. An overall model would be useful for practical purposes, but in this case combining the models would only exacerbate the error already present in the relationships. TP is a difficult parameter to model using turbidity in this system, so any effort to reduce variability in the models would be beneficial. For this reason, site specific models for turbidity versus TP were used for this study.

Turbidity was found to be the best predictor for TP at the Inlet and Outlet sites, while discharge was the best predictor of TP at the Causeway. Neither turbidity nor discharge resulted in a significant model for predicting TP at the Lifton site. This is likely due to the natural chemistry of Bear Lake, which has high pH levels and high levels of calcium carbonate. These two conditions combined can cause phosphorus to precipitate out of the water column in the form of calcium phosphate, or phosphorus can coprecipitate with calcium carbonate by adsorbing onto the mineral (Birdsey, 1985; Dean et al., 2009). This would be consistent with the low TP concentrations in all samples taken at the Lifton site. These mineral particles may also scatter more light than organic particles, causing irregularities in the relationships (Gippel, 1995). Although it is typical that these mineral particles will settle faster than organic particles, the pumping of water at Lifton likely resuspended the calcium carbonate minerals therefore causing high turbidity caused by mineral particles. The combination of low TP concentrations in the water column coupled with the light scattering effect of mineral particles may explain the high variability of the models. Because no predictive models could be found for TP at
the Lifton site, Lifton TP values were estimated through linear interpolation which, although not being the best method, has been found to be a sound method for calculating phosphorus loads (Kronvang and Bruhn, 1996; Eads and Lewis, 2002).

LOADING ANALYSES

Seasonal fluctuations were most pronounced at the Inlet site which affected the loading of TSS, TP, TN, and NO₃. The greatest overall and instantaneous loading for both TSS and TP occurred at the Inlet site, coinciding with the peak seasonal runoff. Seasonal changes in loads at the Causeway appeared to be somewhat buffered due to water traveling through Dingle Marsh. The Lifton and Outlet sites are not immediately affected by these same seasonal flows because the sites are generally not used until later in summer when runoff flows have mostly subsided.

Unlike the other parameters, the TN loading rate was highest at the Lifton site and consistently remained higher than any other site during the period of pumping at Lifton. Even with the elevated loading rates of TN at Lifton, the Inlet site had the greatest cumulative TN loading because the Inlet site was operated all year long whereas the Lifton site was only in operation for 80 days during this study. The loading of TN at the Inlet and at the Causeway were very similar during the times both sites were operational. TN was the only parameter that was not greatly reduced as flows traveled across Dingle Marsh.

NO₃ also exhibited unique seasonal patterns. As NO₃ moved from the Inlet to the other sites, it was removed very effectively. During the time in which water was moving from the Inlet to the Causeway, very little NO₃ from the Bear River was delivered to Bear Lake. The only time NO₃ was observed to move effectively through Dingle Marsh was
when the Outlet was opened and flows were being reduced at the Causeway. This water management likely changed flow paths from moving across Mud Lake to move through the bypass canal.

With the availability of high frequency data collection, continuous mass balances can be analyzed giving further insights into system behavior. Two pulses in the runoff accounted for more than half of the overall loading. When the pumps at Lifton began operating, Dingle Marsh shifted from a TSS and TP sink to a source. This may be due to the shift in flow paths through the marsh as the Causeway closed and Lifton began to pump. How this changed nutrient and TSS retention within Dingle Marsh is unknown. The shift from source to sink also occurred after runoff as flows were decreasing. The lower net flow through the system may also have affected the net source to sink shift.

For most of the year, Dingle Marsh appeared to be very efficient at retaining TSS entering the system. At multiple time periods, Dingle Marsh was trapping in excess of 90% of the TSS being delivered to the system through the Inlet. Once the Lifton site was opened, Dingle Marsh began to export TSS from the system and the efficiency remained highly variable through the end of the year.

**BEAR LAKE**

Cumulatively, Bear Lake was a sink for TP and NO₃, and a source for TSS and TN. Out of the four sites monitored, the Lifton and Causeway site had the lowest cumulative loading of all parameters except TN, which was second highest at the Lifton site. The same loading from runoff discharge that affected Dingle Marsh had an effect on the loading of Bear Lake as well. However, as the water moved across Dingle Marsh and Mud Lake, TSS, TP, TN, and NO₃ concentrations were all reduced. The peak loadings of
all parameters to Bear Lake occurred when discharge at the Causeway was at its highest. Concentrations of all parameters were also at their highest point during the peak discharge for the Causeway. The same correlation between peak discharge and peak loading was observed at the Lifton site when water was leaving Bear Lake.

**DINGLE MARSH IS A SUSPENDED SOLID AND NUTRIENT SINK**

On an annual basis, Dingle Marsh was found to be a net sink for nutrients and TSS. There were times throughout the year, however, when the marsh was a source for TSS and nutrients. The marsh’s retention efficiency also varied throughout the year. When our data is compared with past studies on Dingle Marsh, the retention capacity observed appears to be highly variable from year to year (Table 6).

Table 6: Net total suspended solids and nutrients retained by Dingle Marsh. Positive values indicate that a greater percent was retained by the marsh while negative indicates that a greater percent left the marsh. *For this study TN and NO₃ results were based off of linear interpolation of grab samples.

<table>
<thead>
<tr>
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<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>-1%</td>
<td>56%</td>
<td>35%</td>
<td>60%</td>
<td>49%</td>
</tr>
<tr>
<td>TP</td>
<td>5%</td>
<td>34%</td>
<td>16%</td>
<td>51%</td>
<td>29%</td>
</tr>
<tr>
<td>TN*</td>
<td>-93%</td>
<td>-19%</td>
<td>-9%</td>
<td>19%</td>
<td>38%</td>
</tr>
<tr>
<td>NO₃*</td>
<td>13%</td>
<td>42%</td>
<td>44%</td>
<td>19%</td>
<td>84%</td>
</tr>
<tr>
<td>Water Year % of Avg.</td>
<td>37%</td>
<td>200%</td>
<td>155%</td>
<td>328%</td>
<td>77%</td>
</tr>
</tbody>
</table>
The variations in the annual load calculations to Dingle Marsh can be greatly attributed to the dynamics of the system. The annual hydrology of the system is a driver to the amount of loading and to sink locations of those loads. The annual flows for each of the study years in Table 6 are vastly different, ranging from 37% of normal to 328% of normal. The results in Table 6 also illustrate the high variability in this system and further highlight the need for some type of annual monitoring plan. The differences in water supply also make direct comparisons difficult, but allow for insight into the effect of hydrology on the nutrients retained by Dingle Marsh.

The ability of Dingle Marsh to retain TSS and nutrients in Bear River water has an important impact on the water quality of Bear Lake, because concentrations of TSS and nutrients in the Bear River are much higher than those of Bear Lake. During the period of time that water was being transported to Bear Lake from the Bear River (27 October 2007 to 29 June 2008), Dingle Marsh trapped a significant portion of all parameters measured (Table 7).

Table 7: Fraction of total suspended solids and nutrients retained by Dingle Marsh as water traveled from Bear River to Bear Lake during the 2008 water year. *TN and NO₃ results were based on linear interpolation of grab samples.

<table>
<thead>
<tr>
<th>Net trapped going from Bear River to Bear Lake</th>
<th>Dingle Marsh</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>88%</td>
</tr>
<tr>
<td>TP</td>
<td>62%</td>
</tr>
<tr>
<td>TN*</td>
<td>20%</td>
</tr>
<tr>
<td>NO₃*</td>
<td>95%</td>
</tr>
</tbody>
</table>
During baseflow (September 2007 – May 2008), there were very low concentrations of each parameter and some portion of each was trapped as it passed through Dingle Marsh. These findings are similar to those found in Bjornn et al., 1989 during the same period of the hydrograph (Table 8).

Table 8: Fraction of total suspended solids and nutrients trapped by Dingle Marsh during baseflow. *TN and NO$_3$ results were based on linear interpolation of grab samples.

<table>
<thead>
<tr>
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</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>56%</td>
<td>60%</td>
<td>53%</td>
</tr>
<tr>
<td>TP</td>
<td>57%</td>
<td>62%</td>
<td>20%</td>
</tr>
<tr>
<td>TN*</td>
<td>16%</td>
<td>32%</td>
<td>4%</td>
</tr>
<tr>
<td>NO$_3$*</td>
<td>52%</td>
<td>6%</td>
<td>92%</td>
</tr>
<tr>
<td>Water Year % of Average</td>
<td>155%</td>
<td>328%</td>
<td>77%</td>
</tr>
</tbody>
</table>

During the runoff period (May 2008 – July 2008) when the greatest discharge was occurring at the Inlet site, net trapping for each parameter was increased in Dingle Marsh. The percent load of TSS, TP, and TN trapped within Dingle Marsh were all higher during the runoff period than compared to the baseflow period. The runoff period brought a much higher load of TSS and nutrients than the baseflow, but Dingle Marsh was able to trap a higher percentage of all parameters measured. In the Dingle Marsh system, as discharge increases and loads increase, the percentage of loads trapped also increases.
Results from the 2008 water year are shown with data from Bjornn et al. (1989) for comparison (Table 9).

Table 9: Fraction of total suspended solids and nutrients retained by Dingle Marsh during runoff. *TN and NO₃ results were based on linear interpolation of grab samples.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>43%</td>
<td>89%</td>
<td>91%</td>
</tr>
<tr>
<td>TP</td>
<td>59%</td>
<td>77%</td>
<td>72%</td>
</tr>
<tr>
<td>TN*</td>
<td>16%</td>
<td>68%</td>
<td>29%</td>
</tr>
<tr>
<td>NO₃*</td>
<td>48%</td>
<td>82%</td>
<td>96%</td>
</tr>
<tr>
<td>Water Year % of Average</td>
<td>155%</td>
<td>328%</td>
<td>77%</td>
</tr>
</tbody>
</table>

Spring runoff consistently appears to be a time of TSS and nutrient retention for Dingle Marsh. Although significant differences in annual water supply exist between each of the studies, the overall retention rate is positive for all parameters in each of the three years. The high water levels within Dingle Marsh during spring runoff likely create areas of the marsh where water velocities are slowed and depositional areas are created.
At each site within this study there was a correlation between times of peak discharges and peak loading. This was observed at least to some extent for every parameter measured (TSS, TP, TN, NO₃). To increase loading rates at least 100 times higher than those at baseflow conditions, each site had to reach a specific discharge. The discharges at each site which were responsible for this phenomenon include the Inlet 450 cfs, the Causeway 350 cfs, Lifton 700 cfs, and the Outlet 780 cfs. Although a large portion of the discharges at each site is driven by water supply and demand, there is also a potential of maintaining lower loading rates by maintaining discharges below these suggested thresholds. The limitations of this study cannot assure that these same thresholds are constant every year, but these numbers do provide a potential guideline for managers of this system to follow in order to possibly decrease the extreme high loading events observed in this system.

**MANAGEMENT IMPLICATIONS**

Dingle Marsh has proved to be a TSS and nutrient sink in past and present studies. The slower discharge velocities of Dingle Marsh consistently result in the settling of particles being transported by flow. Since the discharge and water levels of Dingle Marsh can be controlled, loading rates can also likely be controlled. If the elevation of water in Dingle Marsh is increased, the water surface area and emergent vegetation area also increase. Both of these factors play a crucial role in the amount of particles that will be filtered or trapped. This information could be used to strategize management goals directed at routing TSS or nutrient loading. Three different management scenarios with different loading goals are outlined below.
In the current flow patterns, Dingle Marsh acts as a sink for TSS and nutrients during the critical loading periods. In this design, much of the excess loading from the Bear River is prevented from entering Bear Lake. The main factor in this pattern is the routing of Bear River inflows through the marsh and across the Mud Lake unit. Although these current strategies may help Bear Lake, the excess loading of TSS and nutrients into Dingle Marsh may be detrimental to the marsh system by increasing eutrophication.

If a management plan was desired to reduce Bear River loading to Dingle Marsh, then reducing the water surface area of the marsh and increasing flow velocities could route more loading to Bear Lake. In this scenario, the elevation of Dingle Marsh would be kept as low as possible to still allow for desired flow management. Water velocities would be increased which theoretically would reduce any settling of TSS and nutrients within Dingle Marsh. Lower water elevations in the marsh would also decrease the opportunity of the biological uptake of available nutrients. This management plan would conceptually create a closer link between the Bear River and Bear Lake and minimize the filtering effect of the marsh system.

A third possible management plan is the flushing of settled particles out of Dingle Marsh back into the Bear River. In a phenomenon not fully understood within the contexts of this study, a large flushing event of TSS and TP occurred when the Outlet and Lifton sites were both initially opened. During this time, the Outlet exported more TSS and TP than was being delivered from both input sites. It is hypothesized that the source of this flushing was from particles that had settled throughout the year on the north end of Dingle Marsh within the bypass canal. Although this event is not fully understood, it may be possible that strategic timing of the increase of Outlet and Inlet flows could push
this stored load to the Outlet canal. In this plan, the bypass canal would be used as a temporary settling area for TSS and TP until water is needed downstream. During the downstream release phase, water could be routed from the Inlet to the Outlet in an attempt to flush the particles out of Dingle Marsh and back to the Bear River. Increasing flows at Lifton could possibly enhance this effect. The goals of this management plan would be to attempt to minimize loading to Dingle Marsh and Bear Lake, and return a large portion of the loads back to the Bear River system.

None of these management plans were scientifically tested as part of this study. The recommendations here are based from observations made of the behavior and loading patterns of the system. All of these management plans could be tested in future studies using the methods outlined in this study.

**MONITORING FREQUENCY**

One of the objectives of this study was to determine the sampling frequency necessary to accurately represent the actual loads. In order to estimate nutrient loading, a common sampling method is to collect grab samples for nutrient concentrations, calculate loads using discharges on those same dates and then interpolate the values between each sample. Typically, the interpolation is assumed to be linear. This method is useful in determining a load, but ignores the variability in concentrations between sample times. Linear interpolation was used to estimate the loading of TN and NO₃ for this study. Although the interpolation allows for a nutrient load estimate for that season, it does not provide a way to estimate future loads in the way the predictive models do.

An additional part of this research was to determine the sampling frequency necessary to accurately calculate the loading at each site of this system. Traditional water
quality sampling requires the manual collection of a grab sample at each site in the system. This methodology is very labor intensive and expensive. The difficulty and cost in collecting highly frequent grab samples has typically forced the researchers to reduce the sampling efforts to specific seasons or fewer locations. Through the use of correlations between surrogate measurements such as turbidity and TSS or nutrient concentrations, a high frequency dataset of water quality values can be determined less expensively. Throughout the duration of this study, turbidity measurements were taken every 30 minutes and surrogate relationships were used to correlate those data to TSS and TP concentrations by way of linear mathematical models.

After selecting the best models at each site for predicting both TSS and TP, these models were applied to the turbidity data to calculate TSS and TP values at the 30 minute sampling frequency. These high frequency values were totaled to represent the best estimate of TSS and TP loading at each site. Then hourly, weekly, daily, and monthly sampling rates were modeled in order to test the error introduced by changing the sampling frequency. Using the specific time scales as strata, 5,000 iterations of random data selection were run for each sampling frequency at each site. The mining of data at varying time scales allows for the quantification of the increase in error brought about by less frequent water sampling (Figure 42 and 43).

At all sites there were no significant differences between hourly and half-hourly sampling. Sampling at decreasing frequencies produces increasingly more variable estimates with a low likelihood that the samples collected at monthly intervals would produce results within 5% of the reference load (Table 10). Deviations from the true load brought about by decreased sampling frequency are shown visually in Figures 44 and 45.
Figure 42: TSS Sampling frequency and error variations. The boxes represent the first and third quartiles (25th and 75th percentiles) and the whiskers correspond to the lower and upper adjacent levels of the estimation of the loading value.

Figure 43: TP Sampling Frequency and error variations. The boxes represent the first and third quartiles (25th and 75th percentiles) and the whiskers correspond to the lower and upper adjacent levels of the estimation of the loading value.
Table 10: Probabilities of estimating the true load.

<table>
<thead>
<tr>
<th>Site</th>
<th>Variable</th>
<th>Probability of being within 5% of the reference load</th>
<th>Probability of being within 50% of the reference load</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Sampling Frequency</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Daily</td>
<td>Weekly</td>
</tr>
<tr>
<td>Inlet</td>
<td>TSS</td>
<td>0.99</td>
<td>0.45</td>
</tr>
<tr>
<td></td>
<td>TP</td>
<td>1.0</td>
<td>0.58</td>
</tr>
<tr>
<td>Causeway</td>
<td>TSS</td>
<td>0.97</td>
<td>0.31</td>
</tr>
<tr>
<td></td>
<td>TP</td>
<td>0.99</td>
<td>0.51</td>
</tr>
<tr>
<td>Lifton</td>
<td>TSS</td>
<td>0.82</td>
<td>0.25</td>
</tr>
<tr>
<td></td>
<td>TP</td>
<td>1.0</td>
<td>0.47</td>
</tr>
<tr>
<td>Outlet</td>
<td>TSS</td>
<td>0.99</td>
<td>0.55</td>
</tr>
<tr>
<td></td>
<td>TP</td>
<td>1.0</td>
<td>0.65</td>
</tr>
</tbody>
</table>

Figure 44: Probabilities of deviations from the true value with TSS sampling.

These results show that in order to produce results within 5% of the reference load, samples would need to be taken at least on a daily basis. This result is in contrast to results in Jones (2008) which found sampling at the daily scale was too influenced by
diurnal fluxes to produce accurate loads. The difference in this study and Jones (2008) may be that due to the less dynamic nature of this system, daily monitoring may still prove to be adequate. This analysis indicates, however, that monthly or weekly sampling programs could not be used to accurately reflect the reference loads.

Figure 45: Probabilities of deviations from the true value with TP sampling.

Jones (2008) also found weekly or monthly monitoring to be inadequate. An analysis of deviations from the true value was not possible for TN or NO$_3$ in this study because predictive models based on flow or turbidity were not found. Despite that, the conclusions about optimal TSS and TP sampling frequency in this system can likely be applied to TN and NO$_3$ as well (Table 10 and Figures 44 and 45).
CHAPTER 6

CONCLUSIONS

ADEQUACY OF TURBIDITY MONITORING FOR SURROGATE USE

Turbidity proved to be a satisfactory surrogate measurement for TSS in the Dingle Marsh system. At the Inlet, Causeway and Outlet sampling sites, turbidity was found to be the best predictor of TSS. At Lifton, a hybrid model of turbidity and discharge was found to make the best predictions. Models developed at each site explained 70% to 94% of the variability in TSS. This not only allowed for calculating accurate totals of TSS, but concentrations could also be predicted at high frequencies. The only other significant predictor of TSS in this system was discharge, which was able to account for 42% to 81% of the variability in TSS. Hybrid models of discharge and turbidity were analyzed at each site, but single explanatory variable models were stronger at all sites except for Lifton.

For TP, turbidity was the best predictor at the Inlet and Outlet sites. At the Causeway site, discharge was found to be a more accurate predictor than turbidity although both models were significant. No significant models could be found for prediction of TP at the Lifton site. This is likely due to the combination of low turbidity and TP levels, coupled with the highly variable light scattering effect of mineral particles found specifically at the Lifton site. These low levels of each parameter would have magnified any error associated with the modeling process. Turbidity measurements were able to account for 63% to 67% of the variation in TP at the Inlet, Causeway and Outlet sites. Discharge measurements could account for 43% to 57% of the variability of TP at the Inlet and Outlet sites and also could account for 82% of TP variability at the
Causeway. Using a surrogate to measure a parameter at high frequencies will produce a greater number of data points, but it comes with the trade off of error introduced by the models. This trade off needs to be considered with the targets for any monitoring program.

Because no predictive models could be found for TP at the Lifton site, TP values between grab samples were estimated through linear interpolation. This has been found to be a sound method for calculating phosphorus loads (Kronvang and Bruhn, 1996; Eads and Lewis, 2002). However, the sampling frequency analysis suggests these samples would need to be collected at far greater frequencies to represent the true load. At the other three sites (Inlet, Causeway, and Outlet), predictive models were found for TP. The concentration estimates of TP at these three sites would be more precise than those estimated at the Lifton site.

The variations in TSS and TP models and coefficients from site to site reiterate the importance of site specific surrogate relationships for this type of study. In an analysis of covariance, the TSS models were found to have significantly different regression line slopes. The TP models did not have significantly different regression line slopes, but the explained variability of the models was already low and using a single surrogate relationship would have only decreased the explained variability. Particle size and composition can vary over short distances in aquatic systems and the variations in models from site to site confirm this point. If a single surrogate relationship is used to predict a parameter at diverse sites, it may still allow for a general analysis of a system, but by doing so a large source of error will be introduced into the analysis.
Through an analysis done on the data collected, daily water quality sampling could produce load calculation within 5% of 30 minute sampling. If the sampling was decreased to weekly sampling, the reliability of the estimates dropped off considerably. The turbidity sensors could be programmed to collect measurements on a daily schedule, but a more frequent program may still be better. More frequent measurement would better allow for the detection of malfunctioning equipment, the wiper blades would operate more often keeping debris clear, and the final load estimate would be more accurate. For this study, turbidity measurements were taken every 30 minutes by an automated sampler 24 hours a day, all year round by use of a solar panel and battery back-up. Through regular weekly maintenance of the turbidity sensors, the wiper blades sustained a clear window for measurements even during times of high algal growth. At times when the surface of the water became frozen, the turbidity sensors were still able to take measurements from the volume of water flowing under the ice. The methods used in this study showed the potential of using this technology to move towards the real-time water quality of this system. These monitoring methods could be used by managers of this system to directly observe the response of water quality when changes are made to the flow management.

**MANAGEMENT STRATEGIES**

Through an analysis of data collected, three management plans regarding loading strategies were formulated. In these plans, TSS and nutrient loads would be directed at Dingle Marsh, Bear Lake, or back to the Bear River. Before any of these plans are applied to this system, further research would be necessary to determine the loading capacity of each of the target end points. This further research would need to include
eutrophication rates of each of the loading end points and also would need to categorize the effects to fish, wildlife, recreation, and other water uses. Repeatedly in this study, the relationship between the hydrology of the system and the loading destinations were illustrated. Further explaining this link may lead to the development of modeling tools that could predict loading and perhaps designate loading sinks.

CONTRIBUTIONS THIS THESIS HAS MADE

Throughout this study, multiple analyses showed the effectiveness of Dingle Marsh at trapping TSS and some nutrients. This study also shows how the trapping effectiveness is greatly reduced if flows are routed through channelized sections of the marsh. Dingle Marsh is still capable of trapping large loads of TSS and nutrients, but seasonal management of flows through the system can increase or decrease this trapping capability. The management implications of this study are that loading of TSS and nutrients can either be directed to Dingle Marsh, or can be short-circuited through the marsh and transferred back to the Bear River through the Outlet. Which of these scenarios used would depend on seasonal or annual management goals.

This research has confirmed the validity of surrogate sampling for parameters such as TSS and TP. Surrogate measures for TSS and TP result in cost reduction for each individual sample while at the same time producing results at a high frequency. The work of this project and the high frequency monitoring therein produced a detailed view of the variability of the loading of TSS and nutrients throughout Dingle Marsh. Understanding the timing and magnitude of these loading periods is essential for future management decisions regarding water use and resource preservation. Furthermore, a continuous water quality data set allows for the comparison of before and after best
management practices. The high frequency monitoring approach can capture small or seasonally dependent changes brought about by a best management practice that might be missed by traditional monitoring approaches. The mass balance approach provides insights into annual loading information, but also the loading behavior within the data collection period. The information provided by this study can be useful to both upstream and downstream water users and aids in the overall understanding of this dynamic system. The entire flow of the Bear River is diverted through Dingle Marsh year round and the impacts on the water quality are quite variable, depending on total flow and direction of flow. Being able to grasp these changes can influence decisions made on the water management. These data are also useful to upstream and downstream users of this system to help understand their impact to the river.

Additional research questions that remain include:

1) Is there a relationship between turbidity and bacterial contamination in this system? Christensen et al., 2002 were able to find a relationship between turbidity and fecal coliform concentrations. These data coupled with continuous in-stream monitoring allowed them to create a high frequency dataset for fecal coliforms. Bear Lake is a popular recreational area with an abundance of beach recreation near the Causeway site where water is delivered to Bear Lake from the Bear River. There are many agricultural operations upstream of Dingle Marsh which could introduce pathogenic bacteria to the river. It would also be interesting to compare concentrations of fecal coliforms as they are transported throughout the Dingle Marsh complex. A study of this type could be conducted with the use of the monitoring equipment already in place. It would only require determining if a relationship exists between turbidity and fecal coliforms.
2) How does the TSS size and composition affect the relationships found in this study? This study was focused on the overall fate of TSS and nutrients throughout the system. One aspect of TSS transport that was not examined in detail was the TSS particle size and composition differences at each site. It is likely that there would be many similarities between sites at some times of the year, but with the variable flows in the system it is possible that as flow patterns shifted that particle composition could change as well. One example in changes of flow patterns would be when the flows at the Lifton site were greater than those at the Inlet. It can be assumed that the particle composition would differ between the riverine sediment of the Inlet and the lakebed sediment of Bear Lake. Another interest would be to determine the fate of TSS as they travel through the marsh. This research could try to determine the settling and resuspension behavior of the differing TSS types.

3) What is the role of resuspension of previously deposited materials in the suspended solid dynamics in this system? During this study, there was a 15 day period in which the Inlet, Causeway, and Outlet sites were all open. This occurred during a change in water management when water was beginning to be returned to the Bear River. As soon as the Outlet site was open, there was a drop in suspended solid loading at the Causeway from 25 tons/day to two tons/day. At the same time, the Outlet loading increased from zero tons/day to 25 tons/day. During this time, 9,300 acre-feet of water passed through the Causeway and 11,200 acre-feet through the Outlet. Once the Causeway site was closed, the loading at the Outlet increased to an average of 100 tons/day. Six days later, the Lifton site began releasing water, and for the next month the Outlet site exported on average 45 tons/day more suspended solid than was being
supplied from both the Inlet and the Lifton site. This suggests that there was another large source of suspended solid during this period. One possibility is that during the time when only the Inlet and Causeway sites are open, suspended solid is deposited in the area of the bypass canal which is resuspended once flow is directed down the bypass canal to the Outlet. Future studies could attempt to locate areas of focused suspended solid deposition throughout the marsh.

4) What is the long term capacity of Dingle Marsh to process nutrients and trap suspended solids? This research found Dingle Marsh to be an excellent sink for suspended solids and nutrients being carried by the Bear River. The ability to trap these parameters can likely be attributed to uptake by wetland plants and periphyton and by settling due to lowered discharge velocities. The precise flow patterns and depositional areas of Dingle Marsh are currently unknown making it difficult to predict the life span of this wetland. If an excess of suspended solids are delivered to the system, it could alter the wetland and cause a drastic change in the ability of Dingle Marsh to filter suspended solids and nutrients. This change would not only impact water users of this system, but could also diminish the benefits this wetland provides as a National Wildlife Refuge. Quantifying the depositional areas of the marsh could help identify the effect of sedimentation and could also help to determine the lifespan of the wetland.

5) Does the elevation of Mud Lake affect retention of TSS or TP in the system? This study found a correlation between the elevation of Mud Lake and the percent retention of both TP and TSS was found. Mud Lake elevation was a significant predictor of retention of TSS (P <0.0001, R² of 0.48) and of TP (P <0.0001, R² of 0.47) (Figure 46). This may be because when there is more water inside of the system, the water
velocity is slowed and allows for the settling of suspended solids and the uptake nutrients. This relationship between Mud Lake elevation and percent retention of TSS and TP could be used in managing this system. When water is coming into the system, the elevation of Mud Lake is higher; when water is being released from the system later in the year, the elevation of Mud Lake is lower. The question is to determine whether the percent retention is any part dependent upon the elevation of Mud Lake, or if both of these factors are simply the consequences of the water management in this system.

Figure 46: Mud Lake elevation & Dingle Marsh filtering efficiency for TSS and TP.
REFERENCES


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