University of Nevada, Reno

Development of a State and Transition Model for Low Gradient, Perennial Streams in the Northern Great Basin Region

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Abstract

Low-gradient, perennial riparian systems are underrepresented, unique settings in the arid Great Basin region and provide a wealth of uses and services including fertile soils, wildlife habitat, flood attenuation, livestock water, groundwater storage and wildfire buffering. Their desirability has, since the time of western colonization and settlement, led to extensive alterations ranging from channel straightening to displacement of stabilizing riparian vegetation through farming practices and widespread, continuous grazing. This has led to numerous incidences of bank erosion, channel widening and, in many cases, disconnection between the channel and floodplain, greatly reducing the extent of the riparian zone and leading to a shift in dominance from wetland-type species to more upland-preferring plants. These type conversions and degradation are of particular concern as riparian areas, which, despite making up less than one percent of the region, are crucial landscape elements, especially in the face of a warming climate trend. This places a substantial emphasis on maintaining and restoring riparian areas. This is a difficult task considering their complexity, but is considerably aided through the use of State and Transition models that are built around underlying riparian processes and provide a level of predictive power concerning various applications of disturbances, including restoration. This study was focused on developing a STM for low-gradient, perennial streams in the northern Great Basin with the express aim of using quantitative channel/vegetation data and pattern analysis. Chapter 1 provides background information on STM development and riparian theory as well the various elements examined during this project. Chapter 2 documents the process involved with achieving this goal, as well as descriptive elements of the model. Chapter 3 examines the application of remote sensing technology in the context of riparian monitoring and within the context of STM extrapolation across areas outside the study area.

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Contents

Chapter 1: Review of Existing Literature on Riparian Classification, Process Modeling, Ecosystem Service Elements and Remote Sensing

Abstract

This research seeks to contribute to the field of applied riparian ecology in recognition of the vast importance of riparian systems in general, their amplified influence in dry regions of the world, and their ever-increasing role in ecosystem resiliency in the face of global climate warming. In arid/semi-arid regions, the presence of water in these systems has formed a paradox whereby they represent highly desirable landscape components in terms of human interests, but have simultaneously been overused, neglected, modified and generally taken for granted for centuries. Thankfully, this paradigm is shifting towards conservation, restoration, and attentive management that balances benefaction with riparian health. A concomitant effect of this shift is the development of models that can explain observed complexities and predict riparian trajectories based upon a large host of abiotic and biotic factors, including anthropomorphic inputs. In terms of modeling and research, riparia have required additional ponderings not just on their unique processes, but also how they link and relate to the surrounding uplands. This fact, coupled with the prominent need to have functioning riparian systems, places extra emphasis on not just understanding them, but also on creating tools and guidelines that can be used with relative ease and effectiveness by those actively engaged in on-the-

ground management. To do this, it is first necessary to come to terms with prevailing thought and techniques, as well as understand the history and paths that have been taken in pursuit of comprehension. The following sections delve into these matters beginning with the first attempts to organize the associated chaos of riparia, followed by the gradual development of dynamic models for rangelands, including riparian areas. The next section presents a review of riparian ecosystem service elements that this research wishes to incorporate, but that have not previously received much attention within the context of applied, rangeland/riparian-related models (e.g. bird diversity, stream temperature, carbon sequestration). The final section covers the comparatively recent use of remote sensing techniques in monitoring riparian areas. A secondary portion of this project sought to extrapolate the primary riparian modeling effort to remote sensing applications. This was done in an attempt to exploit the greater reach of imagery-related analysis while simultaneously connecting it to a ground-truthed, quantitative-based framework.

Riparian Classification and Predictive Modeling

Classification Systems and Channel Evolution Models

Peering back in time, it is evident that humans have long sought to bestow order upon perceived disorder. This aim is most evident in the formation of classification systems. In fact, a pattern reveals itself when viewing the progress of science, especially in the formative stages of emerging branches. Observations are made, then organized into a classification system from which laws are postulated that coalesce into established theory. This is not a singular chain of events, but rather steps along this process are oft repeated with the resulting theoretical body changing and developing as understanding increases (Goodwin 1999). This process is present in landscape ecology where classification has been a key component of understanding how landscape mosaics interact with and affect ecological systems and how changes in scale and composition yield different results. Such knowledge has allowed for extrapolation of finite data over large areas (Turner & Gardner 2015). Riparian studies, being a subset of landscape ecology, have also benefitted from organizational structures. The basic goal of riparian classification is to impose divisions along riparian gradients based upon physical aspects so as to draw inferences about formative processes and resulting abiotic and biotic configurations (Kasprak et al. 2016). During its infancy, those studying riparian ecology sought to link vegetative patterns to specific fluvial features. This work was largely descriptive, but as the role of physical processes in regulating biotic assemblages became clearer, emphasis was placed on process-based, geomorphological classifications. In recent times, models and classification systems are prone to illustrate not only the influence of physical parameters on biotic patterns, but also how biotic factors, primarily vegetation, can alter geomorphology (Bendix & Stella 2013). Though a relatively young discipline, interest in riparian ecology has generated a bewildering proliferation of classification attempts. This has prompted many synthesis papers and guidelines for developing classification schemes (Gebhart et al. 1990; Osterkamp & Hupp 2010; Buffington & Montgomery 2013; Kasprak et al. 2016). Hallmarks of success are aptly summed as 1) the ability to incorporate diverse structural and functional characteristics and how they respond to disturbance; 2) convey knowledge of underlying processes and mechanisms at work in riparian systems; 3) can be applied to a broad range of spatial and temporal scales; and 4) increase understanding without substantial financial expense

(Naiman et al. 2005). As such, this section will focus primarily on those classification systems that meet the above criteria, though space will be devoted to highlighting the seminal research that these systems were built upon. Discussion will begin with geomorphic classification systems, as they are by far the most common.

A basic assumption of geomorphology is that forms are representative of underlying processes. The large variety of existing classification systems based upon geomorphology can be divided into two main categories: descriptive and process-based. Classifying channels in terms of form alone (descriptive) is useful for inventory purposes, pattern recognition, especially in terms of GIS tools and models, and communicating information about a region. Classifications guided by the processes in which forms develop and that demonstrate how channels and floodplains behave within their associated valleys, offer more informed and defensible ways to assess condition and meet management goals (Brierley et al. 2002; Buffington & Montgomery 2013). In particular, process-based systems can be used to identify reaches that function similarly, thus reducing sampling effort, and assessing current conditions in the context of past disturbance and future potential. This ability is important for making informed management decisions and for developing testable hypotheses. However, an important aspect of any classification scheme is that it maintains a degree of flexibility such that atypical data is not forced to conform with the system. Nature is complicated and fitting it into a given construct should not become the end goal, but rather the construct should be used as a tool to better illuminate the complexity (Buffington & Montgomery 2013; Kondolf et al. 2016).

Some of the earliest work focusing on form-based classification was the channel pattern system developed by Leopold and Wolman (1957), which divided rivers into straight, sinuous, meandering, and braided patterns based upon stream discharge and load. This classification was plan-form and looked at how different patterns were associated with each other. Breakthroughs of the study included confirmation that channel shape is driven largely by flows seeking to establish a quasi-equilibrium between discharge, sediment load, and transport ability. This process is observed in differences in slope where below a certain threshold meanders form, while steeper slopes lead to stream braiding. These findings were expanded in the following decades and included other channel patterns, such as anabranching and anastomosing, as well as bed stability (Kondolf et al 2016). Schumm (1985) devised a process-based classification for alluvial rivers (largely in the Midwest) based upon sediment load, flow velocity, and stream power. This system has five classes: suspended load, straight channel; mixed load, straight channel; meandering (suspended load and mixed load); meander-braided transition; and braided (bar-braided and island-braided). Stability tends to decrease moving from straight to braided channels, though the author recognizes that stability is a matter of degrees in alluvial systems where sediment is being actively eroded, transported, and deposited. Recognizing that stable, alluvial channels will still exhibit lateral shifts as part of their natural behavior, Schumm cautions that it is important to identify stable patterns when classifying a given system in order to understand the relationships between function and form. By first identifying the natural range of pattern variability along a river, it should be possible to locate and replicate the most stable or appropriate channel configuration. Engineered patterns are limited by a river's potential and pushing past these limits requires the river to adjust by way of radical means, such as severe down cutting in straightened sections and blowouts in induced sinuosity. Similarly, identifying areas that are close to pattern thresholds may indicate reaches with the greatest potential for modification. Applications of, and references to, Schumm's model and concepts are widespread, though restricted to lower gradient reaches of rivers where alluvial processes are the dominant drivers of morphology (Carlson 2009; Buffington & Montgomery 2013; Castro & Thorne 2019). Seeking to address the issue of applying data at different spatial and time scales, as well as selecting representative sampling locations in inherently diverse systems, Frissell et al. (1986) developed a classification scheme that relates stream systems to their associated watersheds, ranging from channel networks down to microhabitat. An underlying assumption of the system is that biological patterns are reflections of the physical patterns and these are directly tied to the surrounding watershed of a given stream. A goal of this work was to design a classification system robust enough to incorporate multiple management objectives ranging from pollution control to restoration of fish habitat. Design takes a hierarchical approach, which set a trend for many classification schemes to follow (e.g. Paustian et al. 1992; Hogan et al. 1996; Rosgen 1994). Geomorphic classifications lend themselves well to hierarchical schemes as small units, such as microhabitat, can be placed within a larger unit like channel reach, which falls within a valley segment and so on. At the same time, each higher unit exerts some degree of control upon the lower ones, such that the processes involved with smaller unit dynamics need to be placed within the context of the catchment in which it occurs (Kondolf et al. 2016). Thus, variables selected at higher levels are applicable at lower levels in the system; data collected at differing levels of resolution can be integrated; and suitable starting points of scale can more easily be selected to fit the given needs of a project.

Variables selected by Frissell et al. (1986) included dominant underlying substrate, dominant overlying substrate, water depth, water velocity, and overhead cover.

The work of Frissell et al. (1986) was followed by other process-based systems, such as the floodplain research conducted by Nanson & Croke (1992). They developed three floodplain classes based upon the correlation between sediment transport and erosional resistance of channel-bounding alluvium. On the opposite end of the stream spectrum, Whiting & Bradley (1993) created a classification that sought to address both alluvial and hillslope processes specific to small, forested headwaters. Domains were delineated based upon hillslope gradient, sediment size, valley width, channel width, channel gradient, and channel depth. The underlying assumption is that each domain contains a specific channel type and that each channel type will respond to disturbance in like fashion and function similarly in transporting water and sediment. Concern that landscape processes were being overlooked in sampling strategies, Montgomery (1999) presents the Process Domain Concept (PDC). A central issue illustrated in this study is how to assimilate geomorphic processes into the framework for delineating habitat condition and landscape state. The author notes that this framework is often structured, out of necessity, around localized disturbance, but argues for inclusion of a larger-scale spatial context that is inclusive of downstream disturbance history and upland impacts. The PDC is essentially an expansion of the system developed by Whiting & Bradley (1993) in that landscape units can be separated out by differences in landscape processes. Process domains are portions of the landscape that adhere to specific geomorphic processes that dictate site dynamics, structure, and habitat. Examples provided are floodplains, channels, hillslopes, and hollows, each capable of being further subdivided as needed. At the watershed level, the differing geomorphic process, disturbance regimes, and resiliency can be mapped out into these domains and thus, have significance for management, including sampling strategy.

In more recent times, GIS and remote sensing tools are being utilized and trained to identify fluvial processes across large areas and diverse landscapes. As part of a collaboration between the U.S. Forest Service and Colorado State University, Carlson (2009) reviewed numerous process-based classification systems and developed a GIS-centric approach designed to be applied at various scales across the western United States. This system organizes geomorphic variables into three groups: Energy, hillslope coupling, and lateral confinement. The first group uses valley gradient to explain longitudinal features (e.g. sediment transport, sediment aggradation/floodplain development, planform, bedform). Much research has yielded generalized thresholds between gradient and expected geomorphic features. For example, valley sections exceeding 3-4 percent slope tend to have confined channels with step-pool and cascade features. Where at least one hillslope is coupled (channel intersects with toeslope), colluvium can enter directly into the stream, whereas uncoupled channels are affected by materials transported from upstream locations and supply of new materials is dependent upon bank erosion and bed scour. Dissipation of energy during flood events is controlled in part by valley width. Lateral adjustments are constrained by confined valleys as sinuous channels require approximately seven channel widths of space in order to freely meander. The applicability of the system to diverse landscapes makes it useful for identifying and mapping priority sites for inventory, monitoring applications, and locating at-risk resources within riparian settings (Bledsoe & Carlson 2012). However, problems inherent within GIS digital elevation models can lead to misclassifications of valley types. Steep, dissected slopes are often underestimated, while shallow slopes tend to be overestimated (Brardioni & Hassan 2006).

Intertwined with the development of riparian classification systems are channel evolution models. In simple terms, these models use data to predict channel response following disturbance. Applied to non-simple riparian ecosystems, they, much like classifications systems, are variable. Initial development began primarily in response to issues arising from management of large alluvial rivers, especially in developed areas of the Midwest and southeastern U.S (Schumm 1985; Simon & Hupp 1986; Schumm 1994; Van Dyke 2013). Holding much in common with early rangeland ecological theory, the first evolution models followed a linear mode whereby a functional channel experiences disturbance, adjusts and eventually returns to a state of equilibrium or quasi-equilibrium. This process does occur and linear models are sufficient in some situations, but are unable to predict channel alterations in all systems, especially in areas where spatial and temporal paradigms differ significantly from conditions found where the models were created (Phillips 2006; Hawley et al. 2012; Toone et al. 2014; Fryirs et al. 2016). Additionally, the steps outlined in early models (i.e. vertical incision, bank instability followed by horizontal widening, aggradation, stabilization), though again an accurate depiction in many instances, do not capture evolution that often continues past the models' end points (e.g. additional bank sloughing, increased channel roughness and sinuosity, reduced sediment mobility). In a move beyond discrete linear models adapted to single-thread channels, some newer evolution models adopted the concept of multiple pathways in an attempt to place channel evolution into a broader context that better incorporates landscape features (e.g. braided and meandering channels, spatial discontinuities) and convoluted disturbance histories (Van Dyke 2013). A good example is presented in Makaske et al. (2002) where studies along upper Columbia River demonstrated a long history of channel avulsions with a signature channel succession pattern of avulsion, widening and deepening of the channel, infilling, and eventual abandonment. Being an anastomosing river, there are at least two channels present at each avulsion, often many more, with each being in a different stage of channel evolution. There are also multiple pathways a channel will take during the infilling stage based upon levels of sedimentation which dictate whether infilling is vertical or lateral and whether a channel remains distinct following abandonment or fills in entirely. This level of complexity and presence of multiple channels is beyond the scope of the classic, single-thread models.

To recognize both the possibility of multi-thread channels and existence of non-linearity in many systems, Cluer and Thorne (2014) proposed the Stream Evolution Model (SEM). This model considers stream hydrology, morphology, and vegetative attributes and places evolution into a cyclic context. This model builds upon the concepts in classic models (Schumm 1985; Simon & Hupp 1986) by adding an initial, pre-disturbance stage (Stage 0) and continuing the spectrum of evolution with two additional stages (Stages 7 & 8) beyond the traditional quasi-equilibrium stage. Evidence has shown that many alluvial stream systems in the U.S. were likely multi-threaded prior to disturbances associated with European settlement. Thus, presenting the undisturbed stage of a model as single-thread, may be misleading and actually a representation of a post-disturbance condition. Late-stage evolution is further described in the SEM by a stage of further channel widening, followed by anastomosing channels that somewhat represent an analogue of the multi-threaded, pre-disturbance Stage 0. Flexibility is built in such that stages can be skipped when not present, regress back to previous stages, and stages within the cycle may repeat.

Consistent with and complementing models like the SEM, is the application of biogeomorphology. This recent field of study focuses upon the feedbacks and interactions that exist between the living and nonliving components of the landscape (Coombes 2016). Certainly not a novel concept as demonstrated by the literature covered in this review, which represents but a tiny fraction of the greater whole. However, its adoption into riparian ecology helps bring together disparate elements and more firmly establishes the interplay between physical and biological factors. This is demonstrated in Castro and Thorne (2019) who recognize that hydrology and geology are primary drivers of fluvial processes, but include biotic interactions as a third, main driver in their stream evolution model. The model takes the form of the Stream Evolution Triangle (SET) with each of the three drivers occupying a corner of the triangle. This format allows a wide variety of applications within the triangle. For example, a simple examination of a stream reach can be sufficient to identify the primary driver(s) at that point in time. An incised channel that is little effected by flow regime or surrounding biotic communities would be more under geologic constraints, even though the valley itself may not be confined nor the channel restricted by bedrock. A partially confined channel in a monsoonal climate and with woody vegetation along the banks, would plot towards the middle of the triangle. This exercise helps develop an understanding of the dominant processes at work and their relative contribution to stream morphology and evolution. An assumption of the model is that any one of the drivers can change, such that a channel driven by hydrology, or portion of channel, may lose stream power and move towards a more biological driven system as vegetation becomes established along the banks and bars. Additionally, and perhaps more importantly, existing stream classification and evolution models can be accommodated by the SET.

Placing stream classifications from geomorphologic systems within the SET model, such as Rosgen (1994), can offer new insights as to the influence of each driver and the ability of a system to adjust. Channel types that identify strongly with one corner of the model tend to be more resistant to overall change even though hydrologically driven, alluvial systems often see frequent changes within the planform. Conversely, channels plotting towards the center of the triangle are under the influence of all three drivers and are much more susceptible to change as the influence of each driver waxes and wanes. Placing Cluer and Thornes' (2014) Stream Evolution Model in the triangle not only reveals the influence of each driver during each stage of evolution, but also gives an indication of temporal pattern with stages closer to corners likely to persist, where those towards the middle are more transitory. For example, the multi-threaded, wooded, pre-disturbance, "Stage 0", is predominantly under the influence of stabilizing biotic drivers, while the degraded-incised stages move toward the geology corner and movement back towards multi-threaded, laterally active channels is more under the domain of hydrology. The key point being that stream evolution is influenced, but not predetermined and that SET assumes a level of uncertainty based upon natural variability. The SET also has implications for understanding response and recovery to disturbance at different scales. One essential component of recovery is a positive response in biological process.

Flooding can remove vegetation and shift/widen channels. This represents temporary process control by hydrological factors. Should flood frequency represent a fairly constant disturbance, the system may stay within the sphere of influence presided over by hydrology. A resulting deepening of the channel, may trend towards geological drivers resulting from incision, and post-flooding colonization and subsequent growth represents a biological uplift that moves the system towards either pre-disturbance conditions or some quasi-stable analogue. Recognizing the relative influence of the three primary drivers can inform managers as to whether fairly straight-forward approaches are sufficient to initiate lasting recovery or if the level of complexity warrants a more complicated approach, or even discourages certain approaches, such as forcing channel stability (Shafroth et al. 2002; Castro & Thorne 2019). This model does not provide pat answers to problems. The benefit that the SET offers is a space in which to think about a system and form theories that explain the underlying processes.

Ultimately, it is this space in which to think is what embodies and advances scientific endeavors, especially in complex, dynamic systems. The choice of classification system, or decision not to use one, ultimately rests upon the goals of the researcher and/or land manager. However, from an applied standpoint, limitations do exist in the form of practicality and financial resources. Adoption of new ideas can also take time, especially in the public arena. Advancements made in academic settings may not be implemented in actual policy change for decades. This is clearly demonstrated with the long-standing persistence of equilibrium-based models in range management, that only gave way to a different paradigm towards the end of the 20th century (Westoby et al. 1989; Vetter 2005). Unfortunately, it is not possible for management to keep pace with discovery. The available tools for private and public land custodians must go through a process of development, implementation, and refining before eventual use. Coupled with this process is the need for standardization, especially across governmental agencies. This goal has never been fully achieved, but close examples exist, such as the interagency use of ecological sites across much of the United States. These sites serve as form of land classification based upon soils, climate, and physiography and are just beginning to be applied in riparian settings (Stringham & Repp 2010). More in-depth discussion about riparian ecological sites is presented in the next section, but the point here is that in practice, riparian management tends to use tools that are established, available, and that can translate to a variety of settings, at least on a regional scale. As such, it is not surprising that the STM model within riparian-based ecological site descriptions have made use of the Rosgen Stream Classification System (RCS).

The RCS is widely used by land managers as it provides a common language for describing streams, can be applied to diverse settings, and the standardized measurements can generally be readily applied in the field. In brief, this system contains four levels: 1) geomorphic characterization; 2) morphological description; 3) stream state or condition; and 4) validation and monitoring. Level 1 categorizes streams into eight possibilities based upon longitudinal profile, channel cross-section, and plan view. In Level 2, the stream types are further subdivided into six categories related to number of channels, entrenchment ratio, width-to-depth ratio, sinuosity, dominant particle size, and water surface slope. Level 3 assesses condition in terms of stability, potential, and function by way of such variables as bank erosion potential, meander pattern, flow regime, vegetation, etc. The final level uses field data to verify initial categorizations and develop a monitoring plan (Rosgen 1994). The overarching goal is to provide a universal system that allows the user to predict stream behavior based upon appearance and determine response to disturbance, restoration potential, and numerous other management actions (Mersel et al. 2014). Despite its widespread use, the RCS has come under much scrutiny since its inception. General concerns are that the field measurements are time intensive and that some of the metrics, especially “bankfull”, can be difficult to determine in unstable channels, thus increasing the likelihood of misclassification (Naiman et al. 2005). A consistency trial conducted in a mountainous region of Oregon found that with three monitoring groups composed of several monitoring crews, the same stream type was derived by all crews in only 4 out of 12 streams. The largest source of discrepancy was in calculating the entrenchment ratio, which incorporates bankfull depth and is the first variable used in determining stream type. Small differences in this measurement translated to large differences in flood-prone width and resulting entrenchment values, which led to much variation in identification of primary stream types. Additionally, without applying the allowable variation in classification parameters permitted under this system (±0.2 units for entrenchment and sinuosity ratios; ± 2.0 units for width/depth ratios), over 40 percent of the sampled sites would have fallen between two stream types, thus requiring the observer to select one or the other based on their own judgement and further reducing consistency. Furthermore, many sites that fell out as stream type B (moderate-gradient, riffle dominated), were visually more representative of stream type A (steep, cascading, step/pool). This calls into question the validity of the parameter ranges, at least for application within the study area, and devalues the utility of the system if the final classification is dependent upon ocular estimation (Roper et al. 2008). There is also evidence that the cross-section method used for determining particle size and distribution confounds results by mixing together bank and bed materials, thus ignoring the processes and forces involved with hydraulic erosion of different sediments (Simon et al. 2007). Deeper concerns are evident when using the system to guide management action. A portion of the problem is attributed to misuse of the classification scheme. Far too often, full-scale restoration projects are implemented after collecting the morphological data associated with Level 2, thus foregoing stream stability assessments and data validation (Juracek & Fitzpatrick 2007). The lack of follow-through has resulted in failed projects ranging from disappointing to catastrophic (Gillilan 1996; Simon et al. 2007). Misuse is compounded by its widespread adoption by State and Federal agencies coupled with the recent trend of contracting out much of their restoration work. Those that can demonstrate familiarity with the Rosgen system are often selected over those with a more robust understanding of hydrogeomorphic processes, but that lack experience with classification systems (Malakoff 2004; Simon et al. 2007). The system itself has inherent problems when used as a tool to predict future behavior based upon form, especially when forms in unstable channels tend to be transient. At the heart of this issue is the fact that descriptive data is selected at the expense of the processes that create the forms or in other words, “form leads to function”, rather than “form follows function” (Bradshaw 1996; Juracek & Fitzpatrick 2003).

All failings and criticisms aside, the RCS has been rigorously tested over many years and does serve as a useful tool with correct application. Recommendations given by Goodwin (1999), include the advice that riparian classification systems should enhance communication with a common, approachable nomenclature and that classifications should be treated as hypotheses rather than inviolate archetypes. Every time a classification system is used, it essentially tests the system's explanatory capability. Adjustments can be made to improve this power, or the classification scheme can be dropped altogether should no longer prove to be valuable. The staying power and common language of the RCS demonstrates its utility, even though its detractors make valid points. Although a dimension of universality is desired for classification systems and models alike, managers ultimately have to make decisions that pertain to their specific area. Part of this decision-making process is to determine if the results garnered from a given land-stratification system are valid and make sense for the particular suite of environmental factors they encounter. Implementation of a scientific system or model does not free the user from responsible interpretation nor does it suggest blind adherence, irregardless of personal observation.

History and Development of Predictive Models for Rangelands

For over 100 years, the primary goals of range management have been to determine suitable indicators of rangeland health and to develop models based upon ecological processes that predict outcomes of abiotic and biotic disturbance. Inherent within these goals is the need to both prevent rangeland degradation and restore sites that have deteriorated. A growing dissatisfaction with applications of linear successional models, such as the climax model of Dyksterhuis (1949), led to much restructuring of vegetation dynamics theory through the 1970's and 1980's. The existence of non-equilibrium, alternate states, system resiliency, and thresholds between states was encapsulated by Holling (1973) in his discussion of basins and domains of attraction. Evidence of non-equilibrium, and the need for revised theory, was particularly strong in the arid regions of the world and in systems prone to wildfire (Smith & Schmutz 1975; Noble & Slatyer 1978; Cattelino et al 1979; Wickens & White 1979). With recognition of multiple steady states, came a focus on the thresholds existing between states and the mechanisms involved that move systems across a threshold (Wissel 1984; Allen 1988). Eventually, a new model rose to prominence that incorporated the ideas put forth in the aforementioned research corpus. This was the State and Transition Model (STM), as first proposed by Westoby et al. (1989). Diverse interpretations of the terms used in the model led to the formation and adoption of precise definitions, with particular emphasis on ecosystem resistance and resilience (Bengtsson et al. 2003; Stringham et al. 2003).

This model has been used extensively by managers, researchers, and governmental agencies to monitor rangeland health, develop management practices, implement restoration activities, and numerous other applications (Ratcliff et al. 2015). Its success led to its adoption by the Natural Resource Conservation Service (NRCS) in the late 1990's and in 2010, an Ecological Site Manual was created that standardized ecological site descriptions and STMs among the NRCS, US Forest Service (USFS), and Bureau of Land Management (BLM) (Briske et al. 2017). The application of the model, and development of ecological sites in riparian settings, evolved from questions as to whether models, classification systems, and inventory methods used in upland settings were appropriate for riparian areas (Winward 1984). These questions led to a collaborative effort between the BLM and Soil Conservation Service (SCS) that produced a technical report presenting key concepts in riparian functioning and compared 11 different riparian and wetland classification systems in use at the time (Gebhardt et al. 1990). In this report, the term "site progression" was introduced to help elucidate the highly dynamic nature of riparian systems. The idea being that a site progresses through a variety of changes that can lead to differing potential natural communities. The concept was demonstrated in a prototype riparian STM that incorporated processes such as natural dams (e.g. beaver, landslide), aggradation, degradation, stream migration, etc. Each process was linked to possible states and associated stream types described by Rosgen (1986). This model allows for the presence of different plant communities both spatially and temporally within a given riparian site as affected by soil and water conditions. Changes in stream dynamics are reflected in the loss, expansion, or formation of new communities. The linking of processes to successional communities greatly aided cross-referencing to elements in the 11 classification systems examined in the report. A follow-up study by the authors found that the application of ecological site descriptions to riparian areas was practical, but must incorporate surface and groundwater hydrology, geomorphology, fluvial processes, stream characteristics, and similar factors (Leonard et al. 1992). A concurrent study in the mountains of central Oregon arrived at a similar conclusion after relating riparian condition to a series of hierarchical levels starting with physiographic area, then scaling down to watershed, riparian landform, and finally fluvial surface/plant association (Kovalchick & Chitwood 1990).

Work in this arena began in earnest towards the end of the 1990s. Pioneer studies researching the relationship of soil moisture and depth to groundwater in meadow plant communities were conducted in central Nevada (Weixelman et al. 1997) and east-central Oregon (Stringham et al. 2001). In the Nevada study, a STM relating hydrology and grazing disturbance was developed for a dry meadow community adjacent to the riparian zone, but many of the proposed soil and hydrological variables were incorporated to delineate ecological types within the riparian system. An important aspect of the STM work was the finding of quantifiable measures (e.g. basal area of vegetative cover, production of key grass species, infiltration rate) that correlated with the degree of degradation. In Oregon, there existed a distinct soil moisture/depth to groundwater gradient from wet meadows next to the stream channel to dry bluegrass sites found along the lower slopes of the valley bottom. It was found that in these semi-arid settings, typical to the western U.S., even minor drops in the water table could lead to conversions of wet meadows to drier plant communities. This was a significant finding given that downcutting of associated stream channels and gully formation has been dewatering a large proportion of western meadows for decades (Cottam & Steward 1940; Benedict 1982; Chambers & Miller 2011; Oles et al. 2017). Between 2013 and 2016, the utility of applying ecological site theory and STM models was tested along five creeks in southwestern California (Ratcliff et al. 2018). The creeks are located within rangeland with a history of grazing which allowed for the examination of relationships between grazing management, abiotic factors, and plant community composition. Abiotic data was collected over fourteen variables including geology, soil texture, channel sinuosity, and various stream measurements. Biotic data consisted of vegetation composition collected from greenline transects. Cluster analysis was then used to separate out significant vegetation communities and define ecological sites based upon abiotic factors. Observed "clusters" of vegetation defined individual states within a site and final CART analysis was used to determine the role of abiotic factors and grazing in forming different vegetation clusters. Analysis resulted in the formation of two distinct ecological sites, several states, and numerous phases within a state. Channel slope, elevation, soil texture, geology, and watershed size explained the greatest amount of variation between vegetation assemblages. Though suspected to be prime explanatory variables, grazing exclusion and precipitation, proved to be insignificant. This was attributed to low variability in rainfall amounts over the study period; the grazing exclosures were only in place for two years; and that indicator plant species were primarily woody perennials and less apt to respond in the short term to changes in grazing practices. None-the-less, this study further demonstrated the utility of applying the ecological site description and STM format to riparian areas.

The manifest intention of understanding the linkages between fluvial dynamics, stream geomorphology, vegetation community types, and riparian ecosystem resiliency has culminated in the Riparian Complex Ecological Site Description (RCESD) as proposed by Stringham & Repp (2010). This adapted description and STM marries the concept of a riparian complex with the stream/valley classification developed by Rosgen (1994). The riparian complex concept of Winward (2000) embodies the realization that channel geomorphology, hydrology, and resulting fluvial surfaces are major determinants of plant communities at given point in time. As channel shape and location respond to shifts in hydrology, riparian vegetation naturally adjusts to these changes. Thus, with Rosgen's stream classification system, it is possible to identify current channel/valley configurations and predict plant community assemblages as a function of valley-imposed restrictions and channel evolution. Unlike the standard ESD, the STM of an RCESD recognizes the transitory nature and mobility of plant communities within riparian systems and subdivides vegetation into plant community components. These units represent proportional communities that exist within a given phase in an ecological state.

For example, a given state may contain several phases, each with hallmark components (e.g. sedge/willow, moist meadow, Baltic rush/sedge) broken out by percent composition. The phases themselves are identified by channel type with the overall state representing condition (e.g. potential natural channels, stable analogue, entrenched/degraded) (Stringham & Repp 2010). The ability to link states and predict transitions based upon channel evolution is an important step forward in riparian management and has fostered much interest by the BLM and NRCS in constructing RCESDs at a regional level, especially for dry areas where the importance of riparian corridors is notably magnified. Contemporary riparian STM work has been, and continues to be, applied in portions of Utah. Quistberg et al. (2013) collected data from ten stream reaches in northeastern Utah with the goal of developing four RCESDs to aid in BLM allotment management. Two of these site descriptions have since been published in provisional format and are available for public viewing through the Ecosystem Dynamics Interpretive Tool (EDIT) online platform co-managed by the NRCS, Agricultural Research Service (ARS), Jornada Experimental Range, and New Mexico State University (Bestelmeyer et al. 2016). There are also several provisional RCESDs available for northeastern Oregon and in Maine, though the STMs for the Maine examples are not yet linked to channel type and plant community component.

In North Dakota, a desire to have a consistent framework from which to guide effective riparian management sparked interest in the burgeoning protocols used for creating RCESDs. An initial study focused on testing and validating the use of the Rosgen Stream Classification System for prairie streams and finding suitable geomorphic parameters for dividing streams into stable and unstable reaches (Meehan & O’Brian 2019). Analysis of stream channel data (e.g. entrenchment ratio, width-to-depth ratio, slope, sinuosity) found that significant breaks between different channel types matched well the Rosgen classifications, thus this system serves as a reasonable foundation for building RCESDs for prairie systems. However, this study echoes the findings of Stringham & Repp (2010) in that channel classification needs to be linked with vegetation and management practices to form a more complete picture of riparian health and trajectory.

Because geomorphological factors, such as channel shape and depth to water table, affect vegetation development while vegetation simultaneously exerts an influence on geomorphology, it is necessary to develop a fine-scale understanding of these relationships before expanding up to greater spatial scales. In a follow up study by Meehan & O’Brian (2020), prairie vegetation communities were mapped on foot along eight streams in an attempt to solidify these relationships and to tease out specific thresholds between state and phase transitions. They identified three plant community components, each fairly well associated with a discrete fluvial feature (streambank, floodplain, and terrace). Valley-width cross-sections and longitudinal profiles were surveyed at each study reach in order to correlate geomorphic parameters with vegetation data. As with the 2019 study, reaches were also classified as “stable” or “unstable” and assigned a Rosgen channel type. In general, stable reaches were associated with larger riparian vegetation complexes, containing all plant community components. Unstable reaches were typified by narrow strips of the streambank component with little to no development of floodplain and terrace components. It was surmised that unstable reaches have a hydrologic disturbance regime capable of supporting only the streambank component, which due to water availability and plant adaptations, can readily reestablish following flood events. This indicates transitional pathways linked to bed incision, bank failure, channel widening, and aggradation. Actual thresholds can be determined from the channel metrics (e.g. meander width ratio, entrenchment ratio, bank height ratio) and presence, absence, and proportion of plant community components. For instance, it was determined that the terrace community is not present along reaches with an entrenchment ratio of less than 1.4. These fine-scale relationships between vegetation composition/proportion and channel types can now be tested in larger, landscape contexts.

The pathway leading from early succession-based models in uplands to the RCESDs now being applied in complex, riparian settings, demonstrates the ever-evolving nature of ecological research and application as new understandings are brought to light. As this process continues, key challenges to face include correctly applying learned principles to specific areas, while simultaneously upscaling models to cover a broader region. Greater regional coverage is also needed in order bring national riparian management inline with up-to-date, current knowledge. In the context of RCESDs, there is also ample room within the STM framework to incorporate numerous, previously unexplored elements such as wildlife metrics, carbon sequestration potential, and stream temperature.

Riparian Ecosystem Service Elements

Bird Monitoring

The desirability of riparian habitat to a wide array of wildlife makes the monitoring of wildlife species an integral part of assessing riparian health and functionality. With limited time, funding, and specialized knowledge, birds offer a type of umbrella species that encompass many other elements of biodiversity (Bradford et al. 1998). They are also relatively easy to monitor given their high visual and aural presence. Additionally, bird abundance and diversity are exemplary metrics as they tend to respond quickly to habitat declines and improvement (Bryce et al. 2002; RHJV 2004). Different assemblages, defined by behavior such as foraging and nesting strategy, also make use of a variety of riparian habitat ranging from open water/herbaceous vegetation mosaics to dense stands of mature woody plants. This tendency can help define current riparian status based upon species presence and absence, abundance, and diversity (RHJV 2004; Lussier et al. 2006). Breeding birds are the usual target for general surveys as habitat suitable for a variety of breeding birds is thought to be sufficient to support transient migrants as well as winter residents (Verner 1984). A very common and relatively easy method of monitoring bird populations is through bird point counts (Ralph et al. 1995). Points counts originated from the standardization of a common, naturalistic tendency, namely, standing quietly in place and gathering information through simple observation and listening. They also evolved out of the need for an increased level of flexibility necessary for sampling patchy habitat that previous transect methods could not provide. In their earliest manifestation, the goal was to record all detectable birds at any distance. Averages by species were then calculated across the sample points to derive an index of abundance (Blondel et al. 1981). The ease of use, a practitioner need only be able to identify birds by sight and sound, and ability to cover large areas at a relatively rapid pace pushed point counts into the forefront of techniques used for monitoring bird populations. This trend was further motivated by a need to quantify and understand noted declines in migrant songbirds across North America (Matsuoka et al. 2014). However, as usage increased, certain drawbacks made themselves known.

First, despite the relatively simple design, there was sizable variation in how counts were being conducted. This prompted an avian research workshop in 1991 that sought to develop a set of common standards that would; 1) allow for comparisons of different studies over time and space; and 2) be useful for predicting population responses to habitat alteration (Ralph et al 1995). This effort served well for the continuation of determining relative abundance, but as interest in density and absolute abundance grew, it became necessary to institute modifications. A primary concern of relative abundance indices is that they rely upon the assumption that all birds at every point will be noticed by an observer. In reality, variation occurs in true population size and in detectability (Johnson 2008). Recognition of this spurred development of various analytical approaches that sought to account for individuals that were not detected during a survey. These models and statistical methods require additional information including distance of detection from the observer, length of observation period, number of observers, and number of visits per year. The goal of such measures is to account for the main factors that influence detection; singing rates and detection distance (Matsuoka et al. 2014). Whether used singularly or in combination, they all work to factor in the probability of detection which is a function of availability and perceptibility. The removal model is an example geared towards determining availability. This method operates under the assumption that as individuals are removed from a population, the total population is reduced, but the probability of capture remains the same. The rate of decreased captures following each removal serves as an estimate of the total population, or absolute abundance (Farnsworth et al. 2005). Adapted to bird points counts, the observation period is divided into several increments with observations in the first interval being “removed”, so that the next interval only records additional detections. This process is repeated for each ensuing time interval. Post hoc analysis can then tease out patterns such that birds recorded most frequently within the initial time increment are deemed more detectable and vice versa. Probabilities can then be applied to adjust numbers up for less detectable birds, thus accounting for present birds that were not seen or heard by the surveyor (Farnsworth et al. 2005; Sólymos et al. 2018). Distance sampling is used to adjust for perceptibility and works on a similar principle. For instance, population is equivalent to area and the number of individuals occupying the area. The number of individuals actually detected is correlated to distance from observer, observer effectiveness, environmental conditions, and, with wildlife in particular, target behavior (e.g. moving in response to the observer’s presence). Incorporating distance measurements into surveys aids in estimating abundance or density closer to the true population size by adjusting for individuals that were present, but not detected as result of these confounding factors. Although exact distances can be measured for each object of interest, it is more common to divide the search area into categories, reserving more exact measurements for distances closer to the observer. A key assumption to distance sampling is that detection is perfect at distance zero, thus 100 percent or 1 in mathematical terms. From this assumption the percent of undetected individuals as distance increases can be formulated and applied to the count data as a correction factor. Models used for determining detection probabilities are designed to be robust in terms of data pooling, such that variance in detection probabilities over numerous subsets does not affect overall density estimates (Buckland et al. 1993; Rosenstock et al. 2002).

The addition of distance measurements and time intervals greatly adds to the utility of avian point counts. However, even with the standardization efforts documented in Ralph et al. (1995), consistency across studies has not been phenomenal (Matsuoka et al. 2014). Academic studies often use a limited observation distance and a variety of time durations per point location. The North American Breeding Bird Survey is an important, volunteer-driven, annual survey that limits time on point to three minutes and utilizes an unlimited distance for detection (O'Connor et al. 2000). This multitudinous array of methodology makes comparisons across studies difficult, thus limiting the potential knowledge that could be gained regarding birds at landscape levels. As each study has valid reasons for implementing a given study design, a recommended approach to facilitate data comparisons is to incorporate several distance and time intervals. To wit, a study with numerous sampling points in small-scale, discrete habitat, may limit distance to 25 meters and time per point to 5 minutes. Separating out detections into 0-3 minutes, 3-5 minutes, 0-25 meters, 25-50 meters, 50-100 meters, 100+ meters, would cover the needs of the study while simultaneously allowing for direct comparisons of portions of the data to other studies (Matsuoka et al. 2014). Though more time consuming, this ability to provide a source of useable data for the larger research community while still having the flexibility to adjust study parameters to specific needs is important. This is especially true for riparian studies which often require adaptive protocols to account for unique circumstances and habitat variability which ranges from open water to dense, closed-canopy stands of vegetation. This variation provides habitat not just for riparian bird species, but upland, aquatic, and semi-aquatic species alike (Sullivan & Vierling 2009). To illustrate, point count studies typically include only breeding birds. However, studies seeking to relate bird populations to riparian health are best served by also including migratory species that utilize riparia as transitory stop-over habitat. The availability of water, foraging opportunities, and cover serve as attractive inducements for these species and their presence, absence, and abundance serve as indicators of riparian condition (Finch & Yong 2000; Naiman et al. 2005). Correlating bird persistence/decline and level of disturbance with habitat characteristics offers an additional, useful tool for assessing riparian functionality (Bryce et al. 2002).

Survey timing is usually restricted to a 3 to 4-hour window immediately following sunrise as this is regarded as stable time-frame when bird calling and singing is at a maximum (Strong & Bock 1990; Ralph et al. 1995; Knutson et al. 2016). Evening surveys are sometimes used for detecting species, such as many marsh birds, that are fairly inconspicuous and tend to be more vocal at night (Krzys et al. 2002; Nadeau et al. 2008). However, Harms and Dinsmore (2014) found time of season to be more important that time of day for common marsh birds in Iowa. Establishing monitoring goals prior to field efforts is necessary for many reasons, including temporal survey considerations. Overall, species diversity appears to trend higher during morning surveys, but assemblage composition does change based upon time of day. Generalist species, such as granivores are often present in equal numbers during the morning and evening and tend to utilize both upland and riparian habitat. Piscivores, like the belted kingfisher (Ceryle alcyon) and osprey (Pandion haliaetus), will be more active during peak daylight hours when visibility is highest. Many aerial insectivore species, such as swallows, are observed feeding on insects above water most frequently during the evening (Sullivan & Vierling 2009). Related to time of day are seasonality and frequency of visits. For riparia, the breeding season is most commonly selected as these areas not only provide breeding habitat, but also serve as stop-over habitat for migrants moving on to other locations, such as habitat at higher elevations and higher zones of latitude (Finch & Yong 2000; Kirkpatrick et al. 2009). The breeding period is also the peak period of vocalization for birds in general, especially early in the season when males are competing for mates (Simons et al. 2007). Visitation is largely dependent upon the nature of the study, but whereas one visit per season is often sufficient, repeat visits throughout the period of study will improve precision and often necessary where detection probability is low (e.g. inconspicuous target species, dense vegetation, difficult terrain) (Ralph et al. 1995; Knutson et al. 2016).

Stream Water Temperature

Atypical changes in stream temperature can have a large impact on overall lotic system health. The array of aquatic species that have adapted to a given area tend to decline when temperatures exceed or fall below an acceptable range and many process rates, such as aeration and organic matter decay are also impacted by non-typical fluctuations (Bashar & Stefan 1993; Naiman et al. 2005; Burkholder et al. 2008; Zhu & Piotrowski 2020). Though shifts towards colder than average water temperature can be concerning, such as in the management of warm water fisheries, elevated water temperatures, especially as related to current climate trends and continued riparian degradation, have become a paramount concern of water resource managers (Poole & Berman 2001; O’Briain et al. 2020). Fundamental to the question of water temperature dynamics, is heat energy. All water contains heat energy and the greater the concentration, the warmer the water (Poole & Berman 2001). Likewise, heat energy moves from higher to lower concentrations, thus stream temperature is dictated in part by heat exchange between the water, air, channel bottom, and channel banks (Larson & Larson 1996). For example, water entering the alluvial aquifer at the upstream end of a gravel bar will come into direct contact with the underlying substrate and exchange energy. In many cases, the water will encounter cooler material and heat will move from the water into the substrate and reenter the stream at the lower end of the bar with reduced temperature (Hester et al. 2009). However, where flow paths are shallow, it has been found that water can gain heat energy, likely due to solar heating of surface sediments (Arrigoni et al. 2008). The temperature of stream water ultimately results from the interaction of internal (e.g. surface/groundwater exchange, flow rate) processes with external environmental drivers, such as ambient air temperature, elevation, solar angle, and anthropogenic disturbances (Weber et al. 2017). Understanding this interaction requires a fundamental grasp of the functions and interrelated nature of 1) channel characteristics; 2) the underlying aquifer and 3) the floodplain/riparian zone.

Channel Characteristics

The width of a channel relates directly to surface area with greater surface area translating to greater heat conduction and radiation. Thus, a wide, shallow channel will tend to have warmer water than a narrow, deep channel with all climatic factors held equal. Though conversely, shallower and wider channels will also lose heat more quickly during periods of cool weather such as overnight and during winter months (O’Briain et al. 2020). Pools too, though typically cooler at depth, can eventually become sources of warmer water due to greater surface area and prolonged water residency time. Channel gradient also plays a role by influencing flow rate, which factors into how readily solar energy is absorbed by water. Flow rate varies by stream size and flood event, but in general, the higher the volume and velocity of stream discharge, the less responsive it is to atmospheric conditions (Ward 1985; Hawkins et al. 1997; Poole & Berman 2001; O'Briain et al. 2017; Qie et al. 2020). Harvey and Bencala (1993) studied the effects of channel topography on surface-subsurface water exchange along a mountain stream with step-bed morphology in central Colorado. It was found that water followed subsurface flow paths that begin at the downstream ends of pools, detour around steps or cascades, and re-enter surface flow at the head of the next pool. Ground water entering from adjacent hill slopes was limited to areas between these sub-surface flow paths. This topological control on water exchange is consistent with other studies that have documented similar scenarios with log dams, riffles, and gravel bars which increases hydraulic head, causing downward, sub-surface flow upstream of the structure and upwellings in the downstream position (Kasahara & Wondzell 2003; Gooseff et al. 2005; Hester et al. 2009). Regarding stream temperature, induced water exchange (hyporheic flow) influenced by topographical features can have at least localized effects, usually taking the form of cooler water re-entering surface flows due to conductive loss of heat to the substrate (Fernald et al. 2006; Poole et al. 2008; Hester et al. 2009).

Alluvial Aquifer

The alluvial aquifer consists of groundwater held by stream-deposited sediments within the channel bed and floodplain. This water is termed "hyporheic groundwater" and is connected to channels through hyporheic flow, which is the exchange of water between the channel and alluvium. A larger aquifer, the catchment aquifer, contains “phreatic groundwater” which feeds into the alluvial aquifer from beneath and eventually mixes with hyporheic flow as it approaches the stream (Brunke & Gonser 1997; Poole & Berman 2001; Kasahara & Wondzell 2003). Hyporheic flow is significant for many reasons, but in context of stream temperature, it helps buffer streams against diurnal and seasonal temperature changes (Poole et al. 2006; Westhoff et al. 2011; Muntz et al. 2017; O’Briain et al. 2020). This is accomplished by redistributing heat from the channel into the alluvial aquifer where it is retained for a period of time before reemerging into the stream. In the aquifer, heat exchange occurs with sediments and groundwater through conduction and dispersion and water is also less affected by air temperature and solar radiation. Though the mixing of water through hyporheic flow may not be sufficient to change annual, mean stream temperatures, this process does often serve to ameliorate the high and low ends of the temperature gradient experienced on daily and seasonal time frames (Burkholder et al. 2008; Hester et al. 2009; Wu et al. 2020). The connectivity of floodplains to channels, porosity of channel beds, type of channel substrate, and depth of hyporheic zone all regulate the effectiveness of this process (Brunke & Gonser 1997; Johnson 2004).

Floodplain/Riparian Zone

The riparian zone is composed largely of terrestrial vegetation that has the ability to capitalize on shallow groundwater tables and the movement of nutrients and substrate to and from the channel and floodplain during flood events. Hence, the lateral extent of riparian vegetation is largely confined to the active floodplain on a given reach of stream or river. How the arrangement and composition of this vegetation factors into stream temperature is a source of contention among the riparian management and research communities (Johnson 2004). At the heart of the argument is direct solar radiation and its contribution to the heating of water. Proponents of stream shading, through maintenance of streambank vegetation, recognize that shade does not cool water, but rather prevents additional heating by blocking direct solar radiation. Additionally, vegetation can facilitate the trapping of cooler air next to the stream surface by reducing wind speed and increasing humidity (Ward 1985; Beschta 1997; Poole & Berman 2001; Roth et al. 2010). Other researches maintain that though vegetation can reduce the interception of direct solar radiation by water, this form of heat energy only makes up roughly 20 percent of contributing sources. Furthermore, if air is warmer than the stream surface, it will transfer heat to the water even through shaded reaches (Larson & Larson 1996; Larson et al. 2002). The width of a stream, height of vegetation, sun angle, and stream orientation are also important considerations when considering the nature of riparian vegetation in ameliorating stream temperature (Larson & Larson 1996; Poole & Berman 2001). Despite differing opinions of the insulating value of riparian vegetation, it has other attributes worth consideration. For example, the root development and additions to soil organic matter through decaying plant matter increase soil porosity, allowing for greater hyporheic exchange through stream banks (Beschta 1997; Poole & Berman 2001). Vegetation also influences hyporheic exchange through evapotranspiration which creates flux in groundwater gradients. Peak periods of evapotranspiration, usually summer afternoons, coincide with the lowest groundwater levels. This change in the water table facilitates greater surface water infiltration which leads to higher rates of hyporheic exchange (Magliozzi et al. 2018). An exploration of this link between hydrogeomorphological processes, riparian cover and restoration was explored in O’Briain et al. 2020 where it was found that vegetation helped buffer stream temperatures more in catchments that were experiencing the greatest loss of hydrogeomorphological functioning. In other words, cooler water in healthy systems relied less on shading and more on hydrological functions, such as hyporheic exchange. Only in systems where these processes were damaged or absent, did stream shading play a direct role in moderating stream temperatures. In addition to enhanced hyporheic flow through stream banks, plant root structure also increases bank stability, thus reducing erosion. Excessive bank erosion tends to lead to shallower, wider channels which are more susceptible to daytime heating (Bohn 1986; Beschta 1997). Finally, woody debris are an important source of in-channel structures that, much like topographical features, can alter hydrology leading to discrete points of water downwelling and upwelling (D’Angelo et al. 1993; Abbe & Montgomery 1996; Wondzell 2006; Hester 2009).

From a management standpoint, maintaining or attempting to change stream temperature relies mostly on working with these mechanisms of internal control within riparian systems. External drivers, such as climate, are largely beyond the scope of managerial possibility, though it is sometimes feasible to reverse or improve human-related disturbances. This includes such actions as curtailing the release of heated effluent from adjacent industrial sites, lessening of surface runoff to streams by reducing coverage of impervious surfaces, development of legislation that provides for riparian buffer zones in areas subject to forest harvesting, or restoring surrounding uplands (Davidson & Bradshaw 1967; Poole & Berman 2001; Moore et al. 2005; Kaushal et al. 2010). In terms of forest harvesting however, it should be recognized that a consensus regarding a direct link between this activity and increased stream temperature is not evident in the literature. A study in western Oregon compared temperatures in reaches adjacent to clear-cut units; un-harvested units; partial, one-bank buffers and full, two-sided buffers. They found that temperatures typically increased in clear-cut reaches, but returned to normal conditions in downstream, unharvested reaches. Temperatures along partial buffer sites increased slightly or remained the same, while full buffer sites ranged from no change to significantly increased temperatures, depending upon gradient and stream velocity (Cole & Newton 2013). Also in Oregon, Zweiniecki & Newton (1999), examined 14 low-elevation, forested streams with clear-cutting occurring on one or both sides and each unit having a stream buffer between 8.6-30.5 meters. An attempt was made to separate out naturally occurring warming trends associated with discharge and distance from headwater. Warming curves were developed for the purposes of comparing expected rates for undisturbed forest to harvested units. Overall results showed that only slight temperature elevations occurred adjacent to harvested sites and that they were very localized as temperatures associated with undisturbed canopy resumed within short distances downstream.

A critical component of stream temperature management is conducting a watershed analysis. Like any ecosystem or ecosystem component, much variation exists at the landscape scale and accordingly, the importance of external drivers and internal functioning will differ based upon historical patterns of use, geographic location, flood patterns, etc. (Montgomery et al. 1995; Whisenant 1999; Poole & Berman 2001). Ignoring a site’s history and/or only examining a small area out of context of the greater watershed has resulted in many failed restoration efforts (Wissmar et al. 1994). In altered systems, having an idea of pre-disturbance conditions can help direct restoration efforts, especially if records of stream temperatures are available and can be correlated with changes in stream characteristics over time. For example, a creek that has widened due to bank instability may be receptive to corrective measures that focus on reestablishment of stabilizing plant communities. Similarly, channel morphology that has been simplified through straightening efforts can benefit from active or passive restoration that seeks to add channel complexity (Ward 1985; Moore et al. 2005; O'Briain et al. 2017).

Identifying primary thermal controls is also important. Ward (1985) called out flow rate, ground water inputs, and water source as the three, key variables associated with a given thermal regime, but that the importance of these hydrological factors is relative to stream size as smaller streams are more easily influenced by atmospheric conditions. None-the-less, there is reasonably consistent documentation of temperature buffering occurring at locations where groundwater enters a channel (Ward 1985; Bogan et al. 2003). Less documented but still of interest to managers, is the how irrigation may affect or tie into this phenomenon.

Over the years, there has been general concern over how irrigation practices may contribute to water quality declines through streamflow decreases and changes to natural hydrology associated with groundwater withdrawal (Bredehoeft & Kendy 2008; Essaid & Caldwell 2017). However, there has also been a growing awareness that relatively inefficient flood irrigation, that diverts surface flows to adjacent pastures and hay fields, can both sustain flows during low-flow periods and contribute to cooler stream temperatures. This occurs as a result of surface water seeping back into the ground both in the target pastures/fields and along the way courtesy of leaky irrigation ditches (Essaid

& Caldwell 2017). This water percolates downward and recharges the alluvial aquifer. Over time, the water makes its way towards and re-enters the channel. The time lag between diversion and re-entry allows for substantial heat exchange to occur, yielding cooler water once it re-emerges. An example of this occurrence is presented in Stringham et al (1998) where stream temperature data was collected from a small stream in Eastern Oregon. Data loggers were placed above a diversion, within the main irrigation ditch, within the groundwater, and in the channel reach coinciding with an irrigated meadow. This study found reductions in channel flow just below the diversion coupled with a significant increase several hundred meters downstream, indicating the aforementioned return of diverted water by way of groundwater flow. Throughout the warmest months, daily maximum temperatures were several degrees cooler below the diversion and 1-2 degrees warmer than the daily lows. The slow, yet persistent discharge of groundwater to a stream also tends to maintain flows during the hot summer months, and often on into the fall and winter, well after irrigation diversions have ceased for the year (Kendy & Bredehoeft 2006; Fernald et al. 2012; Fleming et al. 2014). The same mechanisms observed with typical flood irrigation practices are often recorded for streams occupied by beaver. Water backed up behind beaver dams effectively irrigates the floodplain and filters down into the alluvial aquifer, augmenting both surface and groundwater storage (Gibson & Olden 2014; Johnson-Bice et al. 2018; Ronnquist & Westbrook 2021). However, the full spectrum of beaver activity and its effect on stream temperatures is more complicated.

By virtue of beaver dam size and numbers, far more surface water is stockpiled in comparison to a standard flood irrigation system. As previously discussed, the cycle of water flowing across the floodplain, percolating into the aquifer, and ultimately emerging back into the channel can have many positive effects in terms of stream temperature buffering and longevity of flow. Hyporheic flow is also induced by forcing water below and through the banks where a dam is anchored (Westbrook et al. 2006; Johnson-Bice et al. 2018). Warming temperatures having also been documented, primarily within the impounded sections of a beaver complex due to greater surface area and reduction of canopy cover resulting from flood-related mortality and beaver harvesting (Gibson & Olden 2014). Rosell et al. (2005) cited several studies with varying results. For instance, in Wisconsin, beaver ponds were found to collect heat in the summer and lose heat in the winter. Dams in some Wisconsin tributaries were also found to raise temperatures, causing them to be sources of heat to their recipient waters. Other studies found temperatures to be warmer downstream of beaver dams and that large ponds successfully dampened daily temperature fluctuations. A follow up review of the Wisconsin studies was performed by Johnson-Bice (2018) with the conclusion that previous results would have been more accurate had water temperatures been obtained within the water column, rather than at the surface, and downstream of dams, rather than just below. A general consensus is that temperature in occupied streams is related to numerous factors, such as the presence/absence of phreatic groundwater inputs, size and number of dams, stream flow volume, and overall vegetative and topographic shading (Rosell et al. 2005; Weber et al. 2017; Dittbrenner et al. 2022).

Carbon Sequestration

Understanding how riparian systems function requires managers to have a thorough understanding of water, soil and plant relations. For example, soil saturation and water table dynamics have a definitive effect on plant community composition and can be used to predict changes in plant communities. Incorporating this knowledge into a riparian STM offers some potential for assessing, at least roughly, levels of riparian soil carbon sequestration based upon expected plant communities and their extent in the system.

Carbon, in various forms, occupies four main reservoirs; the oceans, fossil carbon, the terrestrial biosphere and the atmosphere. The oceans hold the largest amount, followed by terrestrial systems (Schimel 1995). The balance of atmospheric carbon fluctuates as amounts are sequestered on the planet in carbon sinks and released back to the atmosphere from carbon sources. An example of this cycle is the use of CO2 by plants during photosynthesis. Carbon is stored in vegetation during photosynthesis. From this point portions are returned to the atmosphere as CO2 during the respiration process. Additional losses occur when biomass is consumed by other life forms or abiotic factors, such as wildfire. Remaining carbon is incorporated into organic matter as roots and aboveground portions die and accumulate in the soil. A progression of decomposition begins courtesy of soil microorganism that release further amounts of carbon during their respiration process (Post et al 1990; Brady & Weil 2010).

The ratio of stored soil carbon to released carbon varies considerably across a given landscape and is governed by factors like temperature, soil moisture, and rate of organic matter input (Brady & Weil 2010; Schmidt et al. 2011). From a land management perspective, there is great interest in carbon dynamics in terms of plants and the soil. Soil clearly plays a major role in ecosystem functioning and plant growth and is in turn, affected by vegetation it supports. Soil structure and stability are functions of both physical and biotic processes, in which vegetation plays a key role in the latter (Oades 1993).

A sizable component of plant-related soil structure and stability comes from the production and subsequent incorporation of organic matter into the soil profile. Soil organic matter (SOM), which consists primarily of plant material, is comprised of roughly 50 percent carbon and thus, often used interchangeably with soil carbon (Brady & Weil 2010). A crucial aspect of SOM is increased soil water retention, especially in coarse-textured soils (Rawls et al. 2003). This is achieved due to a range of factors including increased aggregation and water infiltration; high porosity, absorption, and surface area; and greater surface area (Lal 2020). Many of these factors also enhance other soil properties and many are interrelated. For instance, soil aggregate stability often increases soil macroporosity, which is related to infiltration and soil microbial activity. Similarly, the aggregation and stability of soil tends to have a positive correlation with retention of SOM, thus forming a positive feedback loop (Blanco-Canqui et al. 2013; Lorenz et al. 2017). In addition to soil health, the accumulation and retention of SOM has become part of a larger conversation pertaining to the increasing concentration of atmospheric CO2 and what role soils play both in ameliorating and contributing to this load (Lal 2002; Koch et al. 2013). Collectively, soils sequester more carbon than forests and the atmosphere combined and represent the largest reservoir of carbon and nutrients in terrestrial ecosystems (Stockmann et al. 2013; Han et al. 2016; Li et al. 2020). The important role of carbon in the soil is tempered in modern times by an unfortunate global trend in soil degradation which reduces its ability to sequester carbon and can change former sinks into sources of atmospheric carbon (Lehmann & Kleber 2015; Green et al.

2019). A review of soil deterioration in the United Kingdom, found numerous degradation threats with soil erosion and reductions in SOM being among the top two culprits. Loss of carbon to water erosion alone was estimated to be between 200,000-760,000 metric tons per year in England and Wales (Gregory et al. 2015). In Chappell et al. (2015), data from long-term plots across Australia was fitted into a modified version of the standard carbon cycling model (RothC) used in that country. By including erosion in the model, it was revealed that much of the carbon loss estimates were incorrectly being attributed to soil respiration, rather than loss of soil organic carbon through erosion. The study's findings further strengthen the importance of recognizing soil degradation as potentially large contributor to carbon flux. However, even with a growing awareness of the role of soils in carbon flux/sequestration dynamics, there still remains a significant challenge in determining point-in-time measurements of soil carbon, much less predicting gains and losses in the face of land use change, climate trends, deterioration, and conservation/restoration efforts. This is particularly true on rangelands where the mosaic of differing management, disturbance, climate, plant communities, topography, and soil types all confound ready appraisals (Schuman et al. 2002).

In the U.S. most rangeland systems are classified as arid and semi-arid, with the eastern edge of the Great Plains falling under a sub-humid clime regime (EPA 2016). Virtually synonymous with these areas is the management-related practice of grazing. The effects of grazing and carbon sequestration has long been studied and, outside of the sub-humid region, found to typically increase soil carbon storage (Derner et al. 1997; Schuman et al. 1999; Stavi et al. 2008; Escribano et al. 2020). However, there is also evidence that even with increased carbon in grazed areas, carbon cycling may be enhanced, leading to greater respiration rates during the warmer months of the year (Welker et al. 2004). Regardless, the gains appear to be a factor of stimulated production in forage grasses, mixing of organic matter into the soil through hoof action, increased tillering and root production, nutrient inputs from defecation and urination, and elevated root exudation rates (Schuman et al 2002; Milchunas & Lauenroth 2016). These effects are primarily the result of well-managed grazing. Clearly, abusive grazing leads to reduced biomass production, loss of cover, and eventual declines in soil health and ability to hold carbon (Abril & Bucher 2001; Pei et al. 2008). The relevance of rangelands in storing atmospheric carbon, paired with concerns over soil loss, places emphasis on land management practices that both maintain current SOC levels and help mitigate rising atmospheric CO2 concentrations (Derner & Schuman 2007; Dean et al. 2015). Previous focus, arising out of the United Nations Framework Convention on Climate Change (United Nations 1992), was on controlling emissions related to industry, energy consumption and transformation, and transportation. This was a logical step and these human-related activities are responsible for the vast majority of increased greenhouse gas in the atmosphere since the Industrial Revolution. However, changes within this sector of society often take long periods of time and are costly to implement (Bodansky 1993; McCarl 2007). Tackling the problem from the perspective of land management arose from seeking out alternative, lower-cost alternatives centered around changes in tillage practices and land use, production of biofuels, altering crop compositions and fertilizer use, and grazing management (McCarl & Schneider 2000). On rangelands, grazing management represents a key way to support mitigation either through practices that protect current SOC stocks or those that work to increase the capacity of formerly degraded soils (Liebig et al. 2010; Booker et al. 2013; Behtari et al. 2019). A crucial aspect of participation in active carbon mitigation practices is the provision of financial incentives, which fosters a reciprocal process where reward for improved practices leads to improvements in landscape functioning (Follet & Reed 2010; Baumber et al. 2020). This practice has thus far manifested primarily as carbon trading markets.

In the U.S., experiments in trading carbon credits to help offset carbon emissions has remained in the realm of voluntary markets, with the closest approximation of a mandatory market being the Carbon Climate Exchange (CCE) established in 2003 (Gans & Hintermann 2013). Once facet of this market was a program that permitted the trading of carbon credits associated with carbon sequestered in rangeland soils. Upfront costs and requirements, including a minimum of 10,000 tons of CO2 for trading rights and a variety of registration and verification fees, together with uncertain market prices reduced the desirability of producers to gain and retain membership (Ritten et al. 2009). The CCE as a whole was plagued with problems and eventually closed in 2010. Key issues with any carbon market are the importance and difficulty of accounting for temporal variability in climate, which plays a key role in the movement of carbon into plants/soil and back into the atmosphere, and from a trade standpoint, the need to demonstrate additional soil carbon gains beyond what currently exists at a given location (Tennigkeit & Wilkes 2008). In the latter case, this can take the form of increasing existing gains or implementing management that shifts a carbon source to a carbon sink. In non-range settings, this can be achieved by switching to no-till methods in cultivation or planting trees following timber harvesting (Lal 2002; Bayer et al. 2006; Wang et al. 2009; Krause et al. 2020). On rangeland, especially those deemed to be arid, carbon sequestration is controlled primarily by abiotic factors, such as precipitation, and thus the level of uncontrollable factors, coupled with high annual variability in these factors, work to impede determination of credible carbon gains. This ties in with the notion of alternate steady states that are recognized as states not likely to transition following removal of disturbance. Without substantial restorative inputs, there is little management can do to significantly alter the rate of carbon storage and can at best, only work to protect the carbon pools that are already present. Furthermore, even where elevated soil carbon can be achieved, the overall flux of carbon into the soil in dry regions is small, thus yielding only marginal financial gain in good years and potentially loss during drought conditions with policies that hold landowners responsible for negative carbon flux. Additionally, gains are further reduced, and losses magnified, by the costs of carbon measurement and monitoring (Lipper et al. 2010).

These problems are highlighted in Booker et al. (2013), in a paper that argues against encouraging inclusion of rangelands into carbon trading schemes, at least until many of the issues have been satisfactorily resolved. A main tenet of this article is that the extent and potential for carbon storage generally increases along a gradient of dry to more mesic sites and that most rangeland, at least within the U.S., falls along the more arid side of the scale. In short, vegetation production and soil organic matter accumulation is low in areas subject to low rainfall and high temperatures (Follet & Reed 2010). Thus, even with land aggradation by multiple landowners, financial gains seldom warrant the effort, especially with the current capacity of U.S. carbon exchange policies. This reality illuminates the need to restructure carbon trading in terms of grazing lands and to further advance understandings of how to better tie variability in rangeland carbon flux to carbon markets.

In the meantime, it is worth further exploring renumeration for rangeland management practices that restore and maintain riparian zones, which, though they represent a small proportion of rangelands, are known to have disproportionately high levels of carbon sequestration and the ability to act as long-term carbon sinks, particularly in more arid regions (Garrastazú et al. 2015; Sutfin et al. 2016; DiGioia 2021).

There are many factors that lead to elevated carbon storage in riparian systems. Access to shallow groundwater, periodic inundation of floodplains, higher nutrient availability, and low gradient, all foster higher vegetation production, which leads to greater carbon acquisition in plants as a whole (Gorham 1991; Hazlett et al. 2005; Mendez-Estrella et al. 2017). Storage of carbon takes place within the aboveground and belowground portions of vegetation, as well as in downed woody debris, sediment, and instream biomass (e.g. fish, algae, particulate organic matter, benthic invertebrates) (Sutfin et al. 2016). Once production outstrips decomposition, organic matter begins to accumulate and litter accumulation in general tends to be higher due to both onsite production and offsite inputs from adjacent uplands (Kayranli et al. 2010). Furthermore, the rate of organic matter decomposition is slowed where anaerobic soil conditions exist, such as in saturated soils. This can be a near permanent condition in some wetland settings, or relatively brief as on the outer floodplain margins (Bunn et al. 2003; Kayranli et al. 2010; Mineau et al 2012). There are also physical constraints, with soil depth being a major limiting factor in the amount of possible carbon storage. Geomorphology dictates that valleys will receive soil inputs, while the surrounding uplands actively supply soil materials. This results in riparian valley soils typically being deeper, thus possessing more storage capacity.

There are exceptions to riparian acting as carbon sinks. Periodic floods will often redistribute organic matter, exposing it to oxygen and subsequent decomposition if not reburied in flood-related sediments (Jochner et al. 2015; Sutfin et al. 2016). Whether carbon is utilized/stored or transported in a given portion of a lotic system depends primarily upon the ratio of production to respiration. Net ecosystem production is determined by subtracting respiration from production such that a negative value represents systems that are dependent upon upstream and external sources of carbon and a positive value suggests that excess carbon will likely be exported downstream. Retention is also based upon fluvial surface composition, channel structure and channel complexity. Elements such as regular inputs of floodwaters onto the floodplain, contributions of large woody debris, and well-established vegetation all work to capture and hold organic matter held in suspension (Allan & Castillo 2007; Harms & Grimm 2008; Sutfin et al. 2016). It follows then that riparian system degradation tends to lead to less organic matter production and storage. Channels that have become cutoff from their floodplains through incision are less able to facilitate carbon distribution and become linear conduits of organic matter and water as lateral movement becomes restricted. This leads to less primary production within the riparian zone and less internal production of carbon. Uniformity, often imposed upon waterways for flood control and navigation purposes, has a homogenizing effect on form and processes, which can also reduce carbon cycling and storage (Dybala et al. 2018; Wohl et al. 2018). Pools of standing water, such as behind beaver dams, have also been documented as sources of methane emissions and to lesser extent, CO2 emissions (Nummi et al. 2018). Mean annual temperature has been found to have a positive correlation with CO2 emissions, but differing precipitation conditions, age of pond, and changes in water levels all play a role in whether or when a given site, or even portion of a site, will act as a CO2 source or sink (Whiting & Chanton 2001; Nummi et al 2018). With methane, formation is restricted to anaerobic conditions, and typically acts as a countervailing effect to CO2 sequestration in deep water wetlands (Whiting & Chanton 2001). However, these same sources become substantial sinks once the succession from standing water to meadow has occurred. In the case of beavers, this is a common process once resources dwindle and beavers move on to other locations or if dams become breached and are not subsequently repaired. Impoundment related flooding often puts particulate carbon into solution, where it can be translocated and potentially returned to the atmosphere. However, a portion often settles out and is incorporated into the sediments captured behind beaver dams or other structures responsible for the impoundment (Johnston 2014; Nummi et al 2018).

It has been well-established that functioning riparian systems provide a wide diversity of ecosystem services such as flood attenuation, buffering of water pollutants from up-gradient sources, increased pollinator abundance and diversity, improved soil stability, and amelioration of hotter, drier climate trends (Langendoen et al. 2009; Cole et al. 2015; Perry et al. 2015; Matzek et al. 2020). What is sometimes overlooked is the interdependent role of carbon in maintaining and advancing these services. Passive contributions include coarse woody debris that not only hold large amounts of carbon, but also provide channel complexity, can increase lateral water flow, and provide streambank stability. Carbon-bearing litter mats provide similar functions at smaller scales (Tabacchi et al. 2000). Denitrification, an important process in removing excess nitrogen from agricultural runoff and other sources, is facilitated by carbon substrates associated with decaying organic matter. This process occurs in the upper soil profile and generally does not extend to nitrates in groundwater. However, a slower process occurs where nitrates are extracted from groundwater by plants and eventually added back to the zone of denitrification through litter and subsequent decomposition (Naiman & Décamps 1997). Plants also act as a source of energy and matter for riparian systems and are themselves partially sustained by nutrient-related reactions in the soil that are fueled by inputs of vegetative organic matter (Tabacchi et al. 1998).

With the importance of carbon to riparian systems and the value of riparia in storing and cycling carbon, comes an ever-growing need to attune management so that goals can be met, but not at the expense of riparian health and biodiversity. The demonstrable capacity of riparian area carbon storage also points at the real potential of rewarding good stewardship with payment as related to carbon credit. Despite the failings of past market attempts, many valuable lessons have been learned in the exchange of financial gain for environmental responsibility (Gans & Hintermann 2013). In terms of rangeland/riparian management, it is evident that many facets need further investigation, including the social aspects of carbon market participation, the feasibility of tracking carbon sequestration over time, variability in annual SOC on rangelands, and the economic viability of carbon trading, especially among the agricultural community where financial solvency is an ever-present challenge (Wong & Mabee 2003; Ritten et al. 2009; Baumber et al. 2020). One step that may help in advancing solutions to these challenges is linking soil carbon data to states/phases within a riparian STM. Though it would be at a coarse scale, this linkage would allow for rough soil carbon estimates without having to incur costs of repeated soil sampling and analysis.

The Role of Beaver in Riparian Functioning and Management

When contemplating riparian resiliency, habitat heterogeneity, hydrological dynamics, and carbon storage, no rumination is complete without considering beaver (*Castor canadensis*), both in terms of their direct presence in streams and rivers, and former occupancy. This is especially true in arid landscapes, such as the Great Basin, where available water is a scarce resource (Fesenmyer et al 2018). An excellent survey of the myriad effects beavers can have upon the form and functioning of riparian ecosystems is provided by Gibson and Olden (2014). Among their observations is that beavers fit the classic definition of ecosystem engineers by altering lotic hydrology, creating pockets of lentic habit within the greater riparian zone, affecting sediment transportation and nutrient cycling, and modifying riparian vegetation, both directly and indirectly. Their overt capabilities pose both challenges and exciting prospects for maintaining and restoring riparian resiliency and capacity. One obvious benefit is that beaver dams facilitate water storage and can contribute to longer duration channel flows through the summer through hyporheic outflow and delaying peak flows downstream (Woo & Waddington 1990; Meentemeyer & Butler 1999; Baker 2003; Westbrook et al. 2006; Nash et al. 2021). The dams themselves are constructed to create pools of water that effectively hide entrances to lodges and bank burrows, allow for relatively safe and easy access to food, and provide for winter food storage below the ice. This expansion of open water increases both surface and groundwater storage (Ronnquist & Westbrook 2021). Impounded water is further spread through the riparian corridor by a series of channels excavated by beaver to allow them safe access to more distance resources. This effect is greatly increased during flood events where the dams add to hydraulic head which typically results in overland flows. The network of channels offers flow paths to floodwaters, effectively irrigating large portions of the riparian zone away from the main channel and creating wetland complexes (Westbrook et al. 2006). Linked to water storage and water spreading, are further benefits related to groundwater and sediment.

In terms of groundwater, beaver-related impoundments raise the water table by establishing a new gradient maintained by the extended contact between soil and stream water away from the main channel (Baker 2003; Gibson & Olden 2014). This is achieved through overbank flooding when peak stream flows exceed bank capacity. In comparison to waterways without beaver dams, or similar obstructions, overflow is usually limited to the immediate floodplain, especially in small, less complex stream systems. The presence of dams tends to greatly increase lateral connectivity and waters a greater portion of the riparian corridor (Westbrook et al 2006). Naturally, this effect is accentuated or diminished by a variety of factors including valley confinement, hydraulic permeability of soils, bank height to beaver dam height ratio, and the intensity of flood stage fluctuations (Chen & Chen 2003). This augmentation of groundwater through bank and aquifer recharge works to both attenuate flood intensity and to supplement channel flow through the gradual release of stored water following flooding (Karran et al. 2017; Fouty 2018; Ronnquist & Westbrook 2021). A raised water table also favors wetland vegetation over upland vegetation. Thus, it is often the case that establishment of beaver colonies in areas where they have been absent promotes expansion of wetland/riparian vegetation due to the prevalence of saturated soils (Baker 2003). For example, much of the Great Basin is occupied by big sagebrush (*Artemisia* spp.) and, especially in disturbed areas, rabbitbrush (*Ericameria* spp.) Where water tables have lowered due to channel incision, these species often occupy space right up to the edge of streambanks, effectively converting former floodplain communities into upland communities (Blank et al. 2003). Neither species is tolerant of oxygen deficient, saturated soils and will be replaced by facultative and obligate wetland species where these conditions persist (Ganskopp 1986; Donovan & Ehleringer 1994; Lautz 2008; Naumburg et al. 2005). Anticipated increases in wetland/riparian vegetation is one reason why beaver reintroduction is often employed in arid and semi-arid stream restoration projects (Pilliod et al. 2018).

Sediment capture and redistribution behind beaver dams has numerous ramifications. In general, by obstructing flows, especially during flood events, beaver dams effectively cause sediments and organic matter to drop out of the water column and settle to the pond bottom (Bigler et al. 2001). The amount of sediments captured depend upon age and size of a dam, upstream sediment inputs, geology, upstream vegetation and a host of other factors (Bigler et al. 2001; Butler & Malanson 2005). Sediment depths in related literature have been recorded as ranging from as low as 0 meters to as high as 20 meters, with averages usually falling between 0.5 and 2 meters (Meentemeyer & Butler 1999; Butler & Malanson 2005; McCullough et al. 2005; Pollock et al. 2007). One implication of sediment aggradation within incised channels is the restoration of a measure of floodplain connection by elevating the streambed. This often occurs on a decadal timescale, or longer, but numerous studies have documented this phenomenon. In Oregon, Pollock et al. (2007) measured sediment depths behind 18 dams and constructed accumulation rates as well as changes in bed slope. The highest annual aggradation occurred in the first year of dam construction and tapered off rapidly with an eventual accumulation rate of about 0.075 meters by year six. Bed slope diminished significantly upstream of beaver dams. These data, and findings from similar studies, allowed for the projection of riparian area growth 90 years into the future. This exercise predicted a two-fold to six-fold increase beyond the current extent (25-50 meters). The use of beaver dam analogues in the Colorado Front range found that 1-year post-installation increased sediment retention to levels exceeding unrestored, reference reaches (Scamardo & Wohl 2020). In the eastern U.S., Curran and Cannatelli (2014) studied channel morphology and sedimentation following the removal of an earthen dam and subsequent occupation by beavers. Beaver dams were in place for two years before flooding removed them. The channel that formed in the former reservoir (behind the earthen dam) was initially unstable and prone to migration and bank failure. With the arrival of beaver, their dams worked to stabilize the channel through additional sediments that narrowed and deepened the channel. There was however, no noticeable increase in bed height or reduction in channel slope given the short duration of the dams. This brings the fate of stored sediments following dam failure in to the discussion.

Average beaver dam longevity is about two years, but can range from less than a year to several decades (Curran & Cannatelli 2014; Hafen et al. 2020). While in place, sediments first form as wedge against the base of the dam, then gradually move back to cover larger portions of the channel bed and backwater regions covered by the pond and associated beaver canals (Larsen et al 2021). Though these latter locations may contain far less sediment than directly behind the dam, they tend to be rich in organic matter and located on the floodplain above the main channel. Once a dam fails, these “further afield” sediments provide an ideal substrate for the development and expansion of emergent herbaceous and woody wetland species. The repeated cycle of dam and pond formation, followed by water level recession with dam failure, is recognized as a strong influence on riparian community development and persistence (Baker 2003; Demmer & Beschta 2008; Gibson & Olden 2014). This includes habitat heterogeneity by imposing varying states of succession at various locations along a stream system (McComb et al 1990; Naiman et al. 2005). Some portion of sediments released from behind a dam are transported downstream. In many beaver-inhabited streams, this journey can be relatively short if, as is often the case, a series of dams is in place and suspended sediments are simply recaptured (Larson et al. 2021). Where no dam exists below a breached dam, or flows are sufficient to take out multiple dams, suspended sediment can reduce water quality and potentially lead to channel downcutting following the reduction in water levels and flood dampening effect of dams (Kondolf et al. 1991; Marston 1994). Still, there is evidence that much of the trapped sediment often remains in place once the dam is removed and tends to be quickly stabilized by vegetation (Butler & Malanson 2005). There is also speculation that incisions downstream of a failed dam may actually be mitigated by movement of the sediment behind the dam. Ultimately, much remains to be discovered about sediment fate and channel reactions to breached beaver dams (Larsen et al. 2021). With the many proven benefits derived from the presence of beaver, it is not surprising that they have become a large part of riparian restoration plans, either through direct reintroduction or by mimicking their effects by constructing instream structures, like beaver dam analogues (Fouty 2003; Pollock et al. 2007; Abrams et al. 2019; Scamardo & Wohl 2020; Wade et al. 2020). These artificial dams are often implemented to also accelerate development of woody riparian species in the hopes that beavers will return to an area of their own accord. As a majority of riparian areas on western rangelands are exposed to some degree of livestock use, a need exists to balance livestock use with beaver occupancy in order to capitalize on the resulting benefits, while still providing for the beaver themselves. Though wild ungulates, such as elk, can dramatically browse willows at the expense of beaver, it is typically livestock that comes to mind with large-scale habitat loss. In addition to forage, riparian areas also hold great appeal to livestock and wildlife for the availability of water and shade, an appeal greatly increased during the summer months (Baker 2003; Beschta et al. 2013). With continuous, largely unmanaged grazing, cattle in particular will happily spend a large portion of a day and grazing season in riparian zones with a predictable end result being vast reductions of willows due to browsing and trampling. Willows themselves tend to be able to handle herbivory and/or beaver utilization, but the combination of both, especially at high levels, reduces annual recovery with an eventual removal from the ecosystem (Baker 2003; Fouty 2003; Gibson & Olden 2014; Small et al. 2016). In essence, excessive grazing can effectively eliminate key woody species and beaver which can lead to channelization and downcutting of a stream (Kay 2003). To address this issue, the practice of conservation-orientated grazing is becoming more prominent and includes tactics like shifting riparian use to the spring and/or fall, periodically resting riparian pastures, and creating water gaps that allow livestock to access water only at select points (Baker 2003; Wyman et al. 2006; Swanson et al. 2015; Small et al. 2016; Fesenmeyer et al. 2018). In several cases, changes in grazing practices have led to recolonization of streams and expansion of existing populations (Dieter & McCabe 1989; Campsey 1991; Pollock et al. 2014; Swanson et al. 2015; Charnley 2019). Resistance towards beaver re-establishment is still present given potential interference with irrigation and other infrastructures; water consumption by willows; and potential downstream deficits in water and sediment (Albert & Trimble 2000; Abrams et al. 2019; Nash et al. 2021). Overall though, active beaver colonization is thought to be a very useful tool in the field of passive riparian restoration and production of forage for livestock. Facing a warming climate, their presence is particularly welcome from the standpoint of buffering against drought (Fouty 2018). As with all things ecological, consistently successful management requires an understanding of underlying processes through observation and research. Therefore, even with the obvious benefits of beaver in riparian systems, especially in the contemporary settings where at least some degree of degradation has occurred, it is important to recognize that they are not "fix-all" solution warranting a lapse in vigilance (Pilliod et al. 2018; Nash et al. 2021).

One potential issue with the return of beaver to sites where they were extirpated, is that current conditions seldom match historical conditions (Gibson & Olden 2014). In some cases, this may affect the ability of beavers to successfully colonize, due to increases in gradient in incised channels, lack of suitable construction and food-source vegetation, insufficient flows, or shifts to higher velocity flood events that routinely breach dams (Gibson & Olden 2014; Small et al. 2016). Natural colonization, versus deliberately releasing beavers, is a safer bet, given that the beavers themselves have decided that a site is suitable for their purposes (Kozlowski et al. 2016). With the enthusiasm for using beaver and artificial structures meant to mimic beaver dams, a situation has developed where practice has outrun our understanding of these practices (Pilliod et al. 2018). For example, translocated beaver, especially where conditions are not to their liking, will sometimes move many miles away from their release site, often causing issues with neighboring land owners. This is not helpful as it often adds to negative perceptions that private landholders may harbor towards conservation groups, federal agencies, and beavers themselves. Beavers do not always build dams either, which may run contrary to restoration goals (Pilliod et al. 2018; Nash et al. 2021). Use of beaver dam analogues, as well as natural dams, have alarmed fishery biologists concerned about fish passage and water temperature (Weber et al. 2017). In the western U.S. there is general consensus that stream temperatures are cold enough to buffer impoundment related rises in temperature and that impacts are highly dependent upon existing site conditions (Collen & Gibson 2001). Similarly, fish passage has been shown to be impeded at varying levels depending upon dam characteristics such as permeability, height, and distribution. In Utah, Lokteff et al. (2013) found beaver dams did not affect the movements of two native trout species, but did restrict the range of the non-native brown trout (Salmo trutta). On the flip side, another study used a 36-year photographic record to investigate the effects of beaver dam removal on a British Columbia stream. The dams were removed in an effort to improve fish passage and flood conveyance. The result was an undesired five-fold increase in mean flow velocity, movement of over 600 cubic meters of sediment, and the change from a multi-thread stream to a single-thread form (Green & Westbrook 2009). None the less, fish biology/beaver dam dynamic is a complicated affair and warrants at least peripheral consideration when developing riparian management goals.

A final note regarding current conditions is the consequences of floodplain reconnection along degraded reaches. This is often the expected outcome of natural and artificial beaver dams, but deserves some careful thought where the former floodplain is vulnerable to erosion due to a lack of stabilizing vegetation. This is especially true on large terraces where old channel meanders and flood scouring accumulate beaver pond overflow and are subject to the same downcutting experienced by the main channel. The literature is largely devoid of commentary regarding this scenario, though John and Klein (2004) mention that on a stream in Germany, dam-related diversions followed relict ditches and that dams can serve as triggers for avulsions. Larsen et al. (2021) mentions the possibility of channel incision following the drying up of meadows following beaver abandonment of a system. Erosion channels on the floodplain were recorded by Levine and Meyer (2014), when observing beaver dam dynamics on a creek in southern Montana. In this case, the erosion contributed to overall channel complexity and dam remnants were found to induce channel meandering due to water being forced into the opposing bank. However, the adherence of overflow to old channels, ditches, etc., can also promote incision as water seeks the current base level of the incised stream, leading to the dewatering of adjacent meadows (personal observation 2021). Whether contributing to channel and floodplain heterogeneity or causing further deterioration, an awareness of the consequences of water being added to older, poor condition floodplains adjacent to an incised channel is necessary for managers and further documentation of results should be contributed to body of riparian ecology going forward. Additional research would also be handy regarding the effects of remnant beaver dams and activity as mentioned in Levine and Meyer (2014). For instance, it has been reported that sediments held in place following dam breaching can become colonized by vegetation, thus helping to continue the channel aggradation process (Gibson & Olden; Pollock et al. 2014). In Oregon, Demmer and Beschta (2008) recorded increased channel sinuosity and roughness from debris, downstream sediment pockets, and an increase in pool and riffle complexity, suggesting a prominent legacy effect. Further exploration of this facet of beaver occupation could substantially increase the utility of predictive models. At the end of the day, beavers are their own masters and do what they do to promote their own survival. The best we as humans can do, is to keep furthering our understandings of their efforts and work, as best we can, with them and the systems they inhabit. Several of the cited works in this review stand out as exemplary summaries of what we currently know and areas in need of further study (e.g. Gibson & Olden 2014; Pilliod et al. 2018; Larsen et al. 2021; Nash et al. 2021).

Remote Sensing Applications in Riparian Ecology and Management

Despite occupying a comparatively small part of rangeland ecosystems, riparian areas are often difficult to access and traverse, and still make up a large enough component of the landscape to stymie field-based monitoring and survey efforts. This reduces the number of observations for a given location which can be problematic when studying stream and river continuums. These systems tend to have a large spatial extent and processes at one location are affected by a variety of factors at the entire watershed scale (Huylenbroek et al. 2020). One solution to this dilemma is remote sensing, which has recently seen advancements in sensors, techniques, and pixel resolution that lend themselves, at least more than previously, to studying the complexities and linear features present in riparian settings. This technology makes use of cameras mounted to either airborne craft (e.g. airplane, blimp, kite, drone) or spaceborne satellites (Vierling et al. 2006). The goal, and challenge, of remote sensing in natural resources is to use an overhead view to detect patterns in the landscape (i.e. vegetation types, bare ground, rock, human infrastructure) and use machine learning to apply or map these patterns over large scales (Campbell & Wynne 2011). Two primary methods for working with remotely sensed data in vegetation studies are pixel-based and object-orientated approaches. Pixel-based methods analyze only the spectral properties of an individual pixel, which are then separated into classes and pooled together into meaningful groups (e.g. fuel type, vegetation community). This method is suitable for low and moderate resolution images, such as those produced from the Landsat satellites. Applied to high resolution images, the extra reflectance variability makes classification difficult, leading to increased classification error (Gergel et al. 2007). In object-orientated classification, classification is achieved by aggregating pixels into “objects” based upon spectral signature and the spatial context surrounding a pixel, and provides away to utilize high resolution images (Burnett & Blaschke 2003). Additionally, objects can be grouped into hierarchical schemes where small objects comprise larger ones, thus relating more easily to traditional hierarchy classifications utilized in ecological theory (Arroyo et al. 2006; Johansen et al. 2010; Weih & Riggan 2010).

In the past 10-15 years, riparian applications of both classification approaches have included mapping vegetation composition and structure, mapping fuels, identifying populations of invasive species, determining vegetative temporal patterns, and determining carbons stocks (Arroyo et al. 2006; Hestir et al. 2008; Bellon-Maurel & McBratney 2011; Suchenwirth et al. 2012; Petrakis et al. 2017; Dufour et al. 2019; Silverman et al. 2019). Yet, with the promise of unprecedented monitoring capability, comes a list of shortcomings unique to interpretations of riparian-related imagery, especially in arid and semi-arid regions. One prominent issue is that in these drier regions, riparian corridors tend to be narrower than the resolution offered by readily accessible land cover data sets (Hollenhorst et al. 2006). From a mapping perspective, it can be difficult to correctly classify riparian vegetation as it is common to have pixels containing a mix of riparian, adjacent upland, and non-vegetated attributes, such as water and bare ground. Coarse resolution can even miss small riparian corridors, such as those associated with first and second order streams (Gergel et al. 2007). Remote sensing with low-level aircraft equipped with aerial cameras can photograph areas with resolution down to 0.1 meter, but this technique comes with a much higher price tag, though the use of drones has lowered this cost (Klemas 2014; Kedia et al. 2021). In remote sensing, drones are referred to as Unmanned Aerial Vehicle (UAV) technology and have been used in various riparian studies. Dunford et al. (2009) flew a paraglider-type drone over 174 hectares of riparian habitat to test out a variety of imagery analysis techniques for classifying vegetation and identifying standing dead trees. An assessment of the impacts of livestock in relation to riparian woody vegetation was instigated by Karl et al. (2020) using a drone to quantify willow (Salix spp.) canopy volume. A project objective was to compare the efficacy of drone estimates to field derived values in an attempt to see if drone use was warranted for such exercises. Tradeoffs were evident with canopy volume being underestimated by the drone, but the entire process from data collection to final processing took less than half the time of field data collection. In the geomorphological realm, Koutalakis et al. (2020), anticipating increased flash flood events related to climate change weather patterns, tested the ability of drone imagery to detect flood debris and changes in channel bed and banks. Test flights were successful and changes in riparian vegetation were also evident. This study has implications for predicting flood impacts and monitoring the post-flood environment through the use of drones.

The use of high-resolution imagery and drones offer numerous avenues of new research. However, for most practical applications operating on small budgets, freely available imagery from satellites such as Landsat, Modis, and Sentinel, are still the most attractive sources of data for land managers. This desirability is increased by platforms, such as the Google Climate Engine, that not only offer an array of gratis data sets, but provides them with much of the initial processing, such as atmospheric correction, already completed (Huntington et al. 2017; Huylenbroek et al. 2020). Landsat imagery is useful for trend analysis as the sensors, pass intervals, and measured wavelengths have remained essentially the same between the original launch in 1972 and Landsat 9, the most recent satellite, which launched in 2021 (Campbell & Wynne 2011; Nitze et al. 2017). Unfortunately, more specific usages, such as mapping riparian vegetation assemblages, are still hindered by Landsat's level of resolution, which is about 30 meters per pixel. A study comparing vegetation maps made using traditional aerial photo interpretation and Landsat imagery found the higher spatial resolution of the photos increased accuracy while satellite imagery was only in agreement 25-30 percent of the time (Congalton et al. 2002). In more recent times, free and publicly available satellite data sets are available at 10-meter resolution from the Sentinel satellites. Additionally, both Landsat and Sentinel craft have sensors that register infrared, along with the standard red, blue, and green bands, which allows for the calculation of vegetation indices, such as NDVI. These indices are another way to distinguish land surface cover types and monitor trends in vegetation health. The NDVI takes advantage of the fact that plants actively absorb solar radiation for photosynthesis, particularly at wavelengths associated with red light, and reflect near-infrared (NIR) light to disperse heat (Baihua & Burgher 2015). Normalizing this aspect, by dividing the difference between NIR and RED light by their respective sum, yields a value between -1.0 and 1.0. This provides a metric of "greenness" where healthier, more productive vegetation tends to have values closer to 1, and small values often indicate unhealthy or sparse vegetation (GISGeography 2021). Water availability plays a major part in determining plant vigor, biomass, and productivity. Accordingly, NDVI has been used to depict vegetation response to rainfall patterns, flood events, and groundwater (Groenveld & Baugh 2007; Aguilar et al. 2012; Baihua & Burgher 2015). The link to water availability and vegetation along stream and river corridors makes NDVI an important management tool within these systems. A primary aim of riparian management is monitoring and assessing riparian health for predictive purposes and to track the effects of land use and restoration efforts.

Riparian status and trend were assessed by Jones et al. (2008) using historical Landsat data to calculate NDVI along a river corridor in southeastern Arizona. Here, the contrast between upland desert shrubs and riparian plant communities lends itself to assessments of greenness using NDVI. This was done by comparing values in a conservation area that excludes grazing and mining activity, to less protected sections of the river. Time since formation of the conservation area showed an overall increase in NDVI values due to increases in riparian vegetation, resulting from reduced herbivory, lack of mining disturbance as well as various flood events that facilitated cottonwood establishment. Negative trends in the unprotected areas were due in part to the natural variability expressed along free-flowing rivers, but also related to continuous, unrestricted livestock use, increases in adjacent urban areas, and a general lowering of the water table from urban and agricultural groundwater wells.

Another study in Arizona, used vegetation indices to track hydrological and meteorological changes along a section of the San Pedro River (Nguyen et al. 2014). Intermittent reaches in the study area showed a 20 percent decrease in NDVI values over the period between 1984-2012. This change was linked to shifts in monsoonal moisture patterns, reduced flow rates and increased temperatures. Demonstration of hydrologic condition in these studies supports the continued use of NDVI, or similar indices, for tracking changes in hydrological regimes and climate change. However, much of its application on rangeland has been restricted to larger rivers and few applications have taken place within the Great Basin region, though exceptions do exist. Huntington et al. (2016) evaluated the proficiency of a 30-year Landsat data set in monitoring groundwater dependent ecosystems in the Great Basin based upon various land and water management scenarios. The study included Maggie Creek and the neighboring Susie Creek drainages in northeastern Nevada and it was found that NDVI correlated well with changes in groundwater levels related to riparian restoration. Susie Creek was also the subject of riparian condition assessment in Fesenmyer et al. (2015), where NDVI was used in concert with water and textural classifications to monitor changes in condition related to precipitation, recovery from wildfire, and a shift to conservation-orientated grazing. The emerging picture of using moderate-resolution satellite imagery and vegetation indices like NDVI to monitor and assess riparian functioning is one of compromise. There is definitely utility in these tools, and the price is right, but limitations do exist in accurately classifying fluvial features and vegetation in small systems and narrow channels. This drawback is partly alleviated by the now freely available 10-meter resolution imagery from the Sentinel 2 satellite. There are also other indices, such as the

Normalized Difference Water Index (NDWI), that uses the NIR and GREEN spectral bands to enhance and better account for standing water (Ashraf & Nawaz 2015). The bottom line of remote sensing is that regardless of the details, its function is to provide large temporal and spatial data sets in order to compliment and expand the reach of traditional field-based approaches. Correlations between images, indices, and actual processes will never be exact, but remotely sensed extrapolations can greatly aid in picking out landscape trends, ascertain areas that are failing or appear to be in good health, and provide starting points and guidance for on-the-ground surveys and restoration activities.

Chapter 2: Riparian Complex State and Transition Model Development

Abstract

Riparian state and transition models are important tools for guiding management within the ever-increasing importance of healthy riparian ecosystems. Their development so far has primarily been relegated to government agencies capable of providing a team of interdisciplinary scientists. This is a reasonable approach, but one that is usually left within the realm of expert opinion and qualitative assessments. In recent times, more thorough, quantitative efforts have been successfully implemented and borne to fruition. This study sought to follow this latter example for the purposes of increasing geographic coverage of riparian STMs, fostering accelerated progress in creating additional STMS and introducing previously unexplored elements within the context of riparian process

modeling. To achieve this aim, a low-gradient, perennial stream was selected in the northern Great Basin region, an area that was currently lacking any substantial STM work, but replete with broad-valley riparian areas in various stages of channel evolution and riparian function. Channel and vegetation measurements were obtained from 14 reaches within the study area and examined using hierarchical clustering and ordination analyses. This resulted in the identification of three distinct states, and numerous phases, each bracketed by distinct plant community components and channel metrics. The primary schema encapsulated by the model is a potential natural channel with a connected floodplain, high entrenchment ratio, low width to depth ratio, and prevalence of stabilizing riparian vegetation (State 1 – Potential Natural Channel). Disturbance factors reducing vegetation (e.g. flooding, grazing, fire) increases channel width and can ultimately trigger vertical incision transitioning stream reaches into a degraded channel state (State 2 – Degraded Channel). Through a process of further deepening and widening, room is eventually created, and stream power reduced, to the point that a new floodplain is created, marking the movement into the final, stabilizing analogue state (State 3 – Stabilizing/Stable Analogue Channel). Without undue disturbance, progression is then marked by a series of channel evolutions culminating in reduced extent version of the original potential natural state. Additional elements considered in the model were bird diversity, stream temperature and carbon sequestration potential. Though direct correlations with individual states and phases were not evident, there was a detectable pattern of increasing bird diversity and soil carbon in association with higher functioning, more complex reaches within the study area. Stream temperature data was most connected to inputs of subsurface, cold water, some of which appear to be tied to below ground, irrigation returns from irrigated hay meadows.

Introduction

Natural and anthropogenic disturbances tend to work together in shaping vegetation dynamics. Susceptibility of organisms to disturbance is often enhanced with the addition of human-caused stress and rates of change are typically accelerated (Archer & Stokes 2000; Morris & Rowe 2014). In much of the western United States, including the Great Basin, the past two million years (Quaternary Period) were marked by numerous glaciations interspersed with relatively brief, variable, warm interglacials. As such, many of the grasslands first encountered by settlers in the mid-1800s are believed to be relics of a cooler, wetter climate and therefore had a diminished capacity to persist under a changing climate and high use by introduced livestock (Cottam 1947; Tausch et al 1993; Morris & Rowe 2014). The combination of climatic changes and human activities in the past 150 years have led to ecosystem conversions, fragmentation, and loss (Archer & Stokes 2000; Morris & Rowe 2014). This is particularly true amongst the dry grasslands and shrublands of western rangelands, and even more pronounced in the relatively scarce riparian zones found within these systems.

Since the time of western settlement, Great Basin riparian corridors have represented key areas for agricultural development due to water availability and increased soil fertility. This trait is pronounced at the lower ends where increased water accumulation and relatively low-relief topography facilitated land development. These low-gradient systems are generally rich in hydrophytic plant species, capable of stabilizing banks and providing habitat for a wide range of wildlife species. Associated impacts from increased post-settlement use include dam construction, groundwater pumping, wetland drainage, flood control, livestock grazing, and replacement of native vegetation with introduced crops and livestock forage (Warkentin & Reed 1999; Sada 2008; Dilts et al. 2012). All of these activities have led to various degrees of loss of riparian species, which has contributed to bank erosion, channel incisement and / or widening and, in many cases, disconnection between the channel and floodplain, greatly reducing the extent of the riparian zone and leading to a shift in dominance from wetland-type species to more upland-preferring plants (Coles-Ritchie et al. 2007). These type conversions and degradation are of particular concern as riparian areas, despite making up only about one percent of the region, offer substantial benefits including wildlife habitat, flood amelioration, livestock water, and wildfire buffering (Thomas et al. 1979; Kauffman & Krueger 1984; Warkentin & Reed 1999; Naiman et al. 2005; Stringham & Repp 2010; Oles et al. 2017; Vidon et al. 2018). Over the past few decades, awareness of the vital role riparian areas play in ecosystem functioning has fostered a wide array of research and restoration efforts (Naiman et al. 2005). This work required several initial steps including coming to consensus on the very definition of “riparian”, developing classification systems, and identifying and understanding key processes involved with the formation and maintenance of riparian areas. Much like the dynamic nature of riparian systems, the theoretical underpinnings of riparian ecology remain in flux and much work is still needed to better understand the processes involved. However, given the degraded nature of many of these systems, coupled with a trend of hotter and drier climatic conditions, application of contemporary knowledge must proceed out of necessity (Norman et al. 2019). This places an ever-growing importance upon the continued development of models that can predict change based upon historical use, current use, natural disturbance, climatic factors, and restoration inputs (Karl & Talbot 2016).

A good example is the state and transition model (STM), put forth by Westoby et al. (1989). By this time, it was widely recognized, especially in dry regions of Australia and Africa, that counter to the traditional successional climax model (Clements 1936), there exist numerous instances of non-conformities, such as alternate steady states; thresholds beyond which recovery is not possible by simply altering or removing grazing pressures; and that not all communities exist in equilibrium (e.g. Hodgkinson & Harrington 1985; O'Connor 1985; Walker 1988). As originally imagined, the STM was organized with a strong bias towards management. Individual states were recognized and included in the model only if they were determined to strongly influence or affect management. Subsequent contributions and criticisms led to the restructuring of the model such that states and transitions were developed based upon ecological principles rather than management objectives (Friedel 1991; Archer & Stokes 2000; Stringham et al. 2003, Bestelmeyer et al. 2004; Herrick et al. 2006). The STM has since been incorporated into the ecological site descriptions used by a wide array of land managers, ranging from private ranches to government agencies. Typically, an ecological site is tied to mapped soil units and encompasses a unique plant community that is shaped by environmental conditions and expected to respond in predictable ways following disturbance. The accompanying STM gives an overview of states that can exist within a given ecological site, the various community phases within a given state, a description of thresholds between each state, and potential pathways of restoration (Westoby et al. 1989; Stringham et al. 2003; Stringham & Repp 2010).

Until recent years, this model was applied almost exclusively to upland systems. The use of the model in non-upland sites evolved from questions as to whether models, classification systems, and inventory methods used in upland settings were appropriate for riparian/wetland areas (Winward 1984). Further investigation found that the STM/ecological site concept was applicable, but must incorporate surface and groundwater hydrology, geomorphology, fluvial processes, stream characteristics, and similar factors (Leonard et al. 1992). Building upon this foundation, the model was further expanded in recognition that changes in plant communities in these dynamic systems do not necessarily indicate a change in ecological site or state and that groundwater connectivity plays a pivotal role in plant community development and composition (Baker & Walford 1995; Stringham et al. 2001; Zweig & Kitchens 2009; Stringham & Repp 2010; Ratcliff et al. 2018, Meehan & O’Brian 2019, Meehan & O’Brian 2020). This work eventually led to the formation of the Riparian Complex Ecological Site Description (RCESD), that not only incorporates hydrology, multiple plant communities, and channel geomorphology, but also successfully assimilates the Rosgen Stream Classification System (RSCS) into the state and transition model (Stringham & Repp 2010). At its heart, the RSCS is a channel evolution model. Channel evolution models have been in common usage since the 1980s and form a strong basis for organizing and thinking about riparian dynamics. By inputting a range of quantitative and qualitative data, these models can greatly aid in predicting how a given channel will react to disturbance or combination of disturbances (Van Dyke 2013). The ability to link states and predict transitions based upon channel evolution was an important step forward in riparian management. The Rosgen system does not lack for criticism (e.g. Miller & Ritter

1996; Smith & Prestegaard 2005; Simon et al. 2007) however, it has been widely adapted by federal agencies and thus serves as a familiar scheme that facilitates communication and understanding. The addition of these riparian-based elements into the STM framework has so far led to satisfactory results where trials have been implemented (Quistberg & Repp 2013; Meehan & O'Brian 2019, Meehan & O'Brian 2020; Meehan et al. 2021).

A key aspect of contemporary ecosystem management is resiliency. In state and transition modeling, each described, steady state has an inherent level of resiliency structured around feedback loops that keep a state from transitioning into an alternate state (Colloff & Baldwin 2010). A significant management step then, is determining which state(s) is desired and what ecosystem processes lead to maintenance of, loss, and return to this state. The long tenure and flexibility of this model is encouraging. The need for periodic updates and continued regional development will persist given the size, complexity, and diversity of landscapes across the western United States, but the model and site description format prove to be vastly useful for land managers. Therefore, the goal of this study is to contribute to the growth of this tool by further bolstering its application and capacity. This will be achieved in part by developing a riparian STM for low-gradient, perennial stream systems in the northern Great Basin, which will help satisfy the "continued regional development" need. To date, construction of these models has taken place primarily in portions of Oregon, Utah and North Dakota. Looping in a large portion of the Great Basin will be advantageous, especially in northern Nevada where riparian ecological site descriptions are entirely absent. Given the general aridity of the Great Basin region, riparian areas have also experienced substantial direct and indirect degradation, both from human use and climate trends, which emphasizes the need to better understand how management related disturbance, including non-use, affects these systems. A second focus of this research is placed on model augmentation. Little research has been conducted regarding the inclusion into STMs of functional metrics such as wildlife diversity/abundance, soil carbon sequestration potential, and stream temperature. Successful incorporation of these ecosystem services would enhance the utility and breadth of the model by offering additional indicators of a given state/phase. Similarly, the large role of beaver in affecting fluvial dynamics in these low-gradient systems needs more comprehensive representation. When included in existing RCESDs, their role is typically mentioned only in the Ecological Dynamics portion of the site description and if present in the STM, given only brief consideration. The complexities encountered with developing these types of models for riparian or upland settings are legion. And utility and applicability come with the inevitable sacrifice of nuance and detail. Still, it is believed that the resulting work will be of benefit to both land managers and the riparian systems falling within their purview.

This study is focused upon the following research questions:

1) Do different riparian states and phases occur within the study area and can they be determined from quantitative channel and plant community data?
2) If so, can a reasonable model be assembled that correlates states, phases and transitions with land use?
3) Can such a model be applied to stream systems beyond the study area that share similar topography and environmental factors?

Materials and Methods

Study Area and Description

The study took place within the Maggie Creek Valley which spans portions of Eureka and Elko Counties in northeastern Nevada. Running through the valley is Maggie Creek, a perennial stream that drains into the Humboldt River near Carlin, NV (Figure A.1, Appendix A). The regional climate is Mediterranean Continental typified by hot summers, cold winters and a mean annual precipitation of 10-12 inches (EDIT 2022). Elevations in the study area range from 5,000-5,500 feet above mean sea level. Gradients are low, seldom exceeding more than 0.1 percent for the majority of the study area. The stream is alluvial, passing through unconsolidated material derived from a mix of extrusive/intrusive igneous rock, consolidated ash, and sedimentary rock rich in calcium carbonate and silica (Plume 1995; Crafford 2007). Much of the lower valley, including the study area, formerly produced large stands of basin wildrye (*Leymus cinereus*) intermixed with wet meadows (Horton 2000). These communities are still present in a few locations, though much of the terrace above the current channel is now dominated by basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) and rubber rabbitbrush (*Ericameria nauseosa*). The active floodplain is occupied by coyote willow (*Salix exigua*) and yellow willow (*S. lutea*) of variable age and density depending upon location. Common species within streambank communities include the aforementioned willows, woolly sedge (*Carex pellita*), Nebraska sedge (*C. nebrascensis*), creeping bentgrass (*Agrostis stolonifera*), bulrush (*Schoenoplectus* spp.), and various weedy forb species. Irrigated hay meadows, located at the north end of the study area, are populated with a mix of basin wildrye and creeping meadow foxtail (*Alopecurus arundinaceus*).

Maggie Creek has undergone an array of management practices in the past 150 years. Of particular note is the legacy of unmanaged, year-round grazing that took place in the Maggie Creek Basin in the late 1800's through the early 1900's following the establishment of a railhead at Carlin, NV (NDWP 2000). The resulting rangeland deterioration was further exasperated by periodic large-scale floods and mining activity associated with the discovery and subsequent development of the Carlin Trend, leading to severe degradation along the majority of the creek by the 1980s (NDWP 2000; Kozlowski et al. 2016). A shift in management was enacted along approximately 20 miles of the middle portion of Maggie Creek in 1993 as part of the Maggie Creek Watershed Restoration Project (MCWRP). This project arose out of a mitigation plan designed to offset the impacts of mining operations, including dewatering, by improving riparian conditions and enhancing habitat for Lahontan cutthroat trout (*Oncorhyncus clarkii henshawi*), a federally listed threatened species (Evans 2009). The MCWRP was enacted on land then owned by Newmont Gold, later taken over by Nevada Gold Mines - Barrick Corporation, and Maggie Creek Ranch. Changes included indefinite grazing exclusion for a portion of the creek (Middle Maggie Pasture, Figure 2.1) and a switch from continuous summer grazing to periodic spring/fall grazing where grazing was not restricted. However, management of the lower five miles of creek, now on Nevada Gold did not change (Hadley Place Pasture, Figure 2.1). At this location, cattle are brought in from the summer range towards the end of summer and allowed to work their way downstream to the home ranch for much of the fall. Reaches within this stream segment remain in a degraded condition. Conversely, where management changed, significant improvements have taken place within the riparian zone. Altering grazing timing led to the reestablishment of willows and herbaceous riparian species along the creek, which increased bank stability. Willow growth also fostered the expansion of beaver populations, both numerically and spatially. The resulting proliferation of beaver dams and wetland complexes have increased sediment retention, raised the water table, and increased lateral hyporheic flow (Kozlowski et al. 2016). Overflow water from the larger beaver dam wetland complexes is also serving to irrigate adjacent meadows. The study area is within Major Land Resource Area (MLRA) 25 and comprises approximately 20 miles of stream/riparian corridor. This includes most of the MCWRP area as well as portions of the Hadley Place. Along Maggie Creek, the spatial arrangement of degraded, recovering, and functioning reaches offers a useful "space-for-time" substitution when determining possible riparian states, phases within states, and potential thresholds between states. To this end, field data was collected pertaining to channel measurements and vegetation composition during the summer/fall seasons of 2021-2023. In alignment with current development of Riparian Complex Ecological Site Descriptions (RCESD), an emphasis was placed upon geomorphological characteristics and classification of channel types was accomplished using the Rosgen Stream Classification System (RCS) (Stringham & Repp 2010; Meehan & O'Brien 2019).

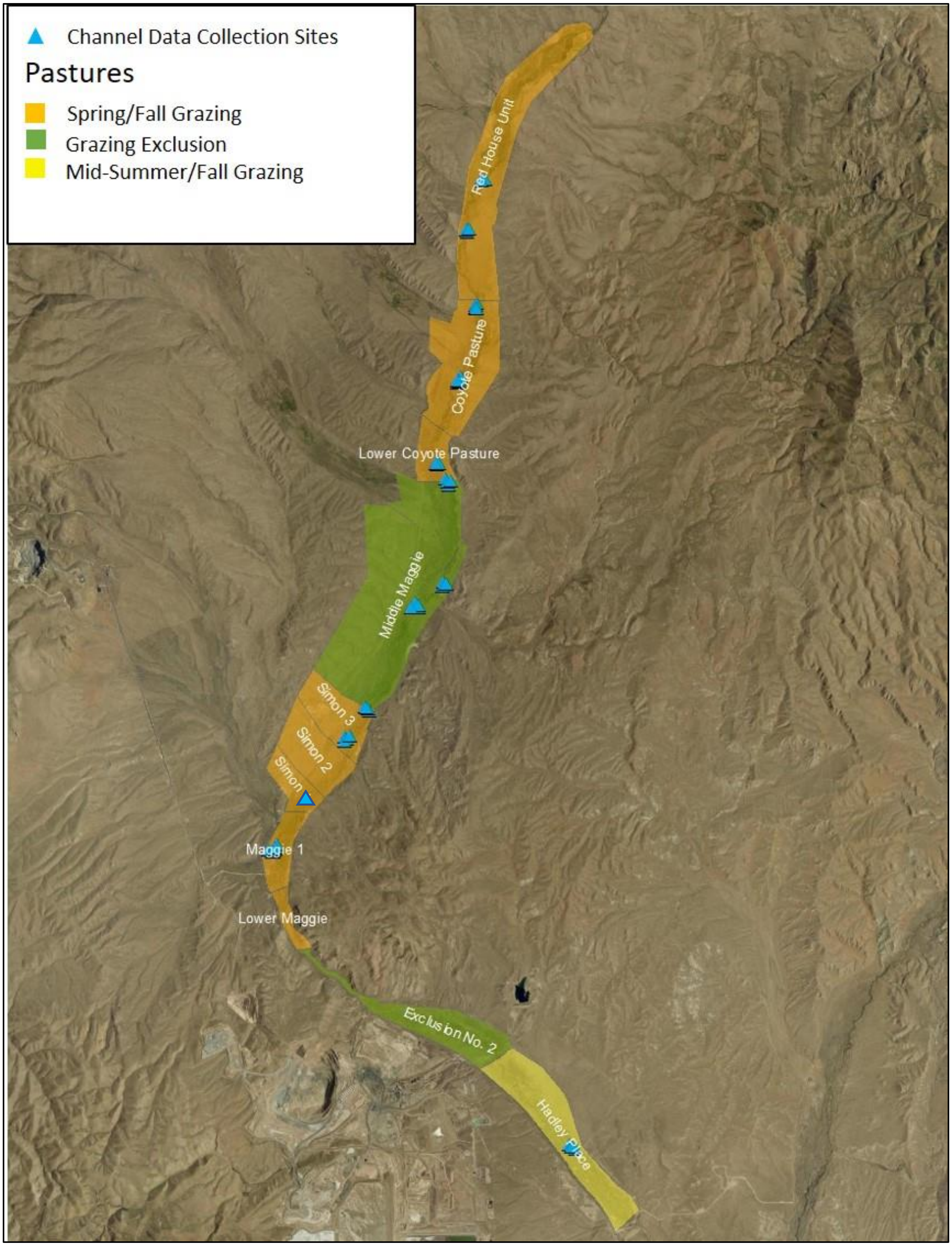


Figure 2.1. *Pastures and Channel Data Collection Sites*

Stream Channel Data Collection

A combination of channel cross-sections and longitudinal profiles were established along 14 stream reaches within the study area and are labeled as CH1-CH14 on the Figure 2.2 map. Collected information was used to determine the Rosgen stream classification for each reach and incorporated into the environmental factors analyzed when determining potential states for the STM (Table 2.1). Factors included Entrenchment Ratio (ER), Width/Depth Ratio (WDR), Sinuosity (SIN), Bankfull to Bankfull Width (BBW), Bankfull Cross-sectional Area (BXA), Maximum Depth (MD), Flood Prone Width (FPW), Bank Height Ratio (BHR), and Mean Sediment Size (D50). Sampling sites spanned a variety of channel types and pastures with different grazing practices including spring/fall grazing (CH2-CH6, CH10-14), grazing exclusion (CH7 & CH9), a water gap with summer access (CH8), and late summer/fall grazing (CH1) (Figure 2.1). Cross-sections were placed near the midpoint of each longitudinal profile. A reach was considered to consist of at least one full meander (two bends) or, in straight sections, at least 20x the average bankfull-to-bankfull width (Harrelson et al. 1994; Rosgen 2006). Most reaches were ~200 meters in length. The spring/summer of 2021 was spent in a thorough reconnaissance of the study area in order to assemble a rough idea of existing channel types and associated plant communities. This information guided the placement of each channel measurement location which sought to capture different potential states and phases within areas of differing land use (e.g. degraded/functioning channel; seasonal grazing, grazing exclusion, flood irrigation). Along each longitudinal profile and cross-section, elevations were obtained using a Topcon RL-H4C self-leveling laser and metric stadia rod.

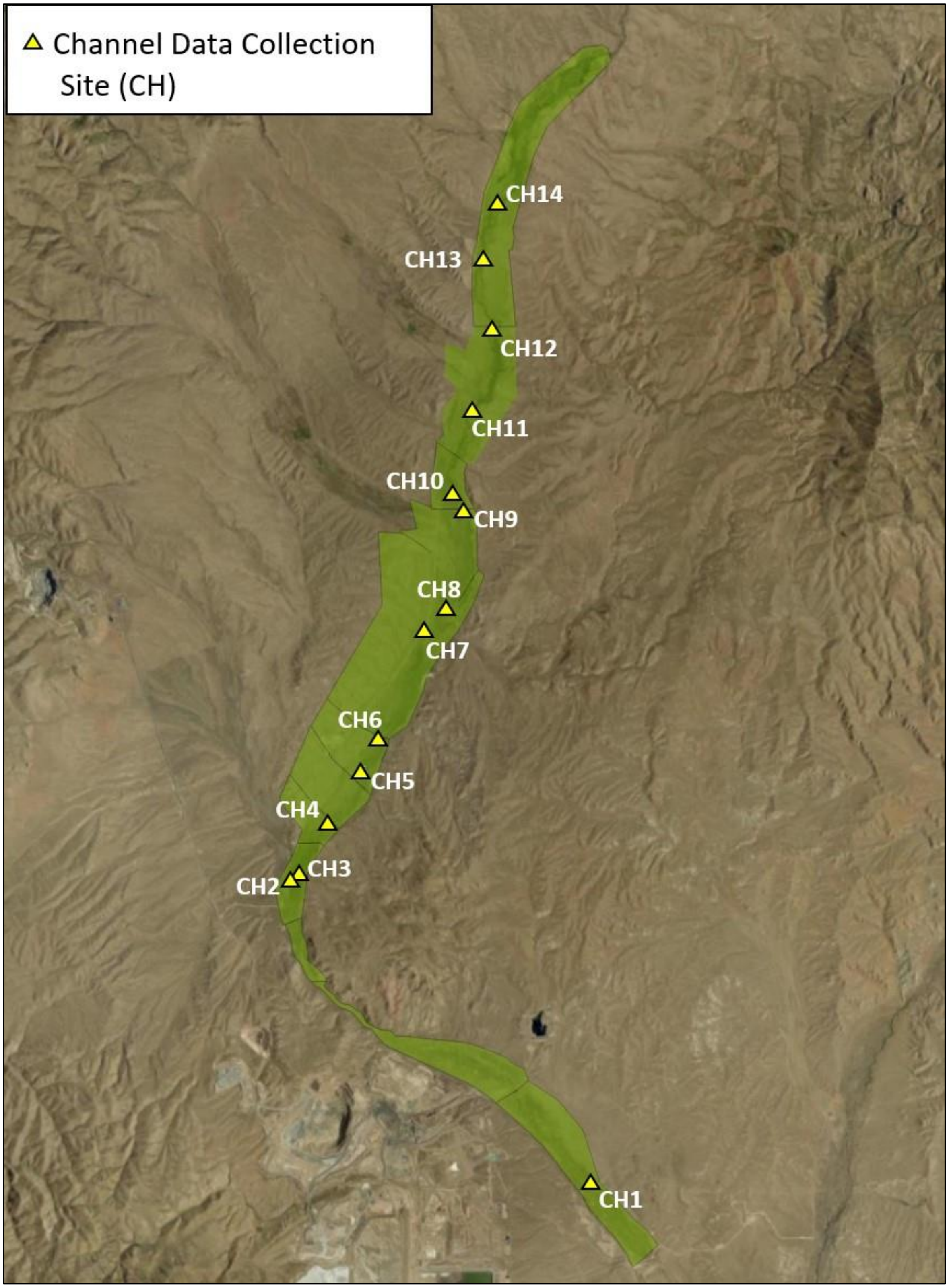


Figure 2.2. *Channel Data Collection Sites*

Measured features included thalweg, top of water, bankfull location, top of low bank and top of terrace. Open reel, metric tapes were used to mark the transect line for profiles and cross-sections and to facilitate the recording of feature positions and subsequent graphing efforts. Channel material size was obtained by collecting 100 samples in a zigzag pattern near a given cross-section and passing each sample through a Wildco® Gravelometer. The D50 size was then determined and used, along with the other metrics, to determine the Rosgen channel type present at each sampled reach.

Plant Community Data Collection

Riparian plant communities are more fluid than their upland counterparts as fluctuating water levels, sediment movement, and changes to channel morphology are common, and often frequent, events that all play a part in shaping species composition and structure (Winward 2000; Stringham & Repp 2010). Yet, even if short-lived, these patterns provide valuable information regarding active processes and disturbance regimes. This is particularly true of plant community composition and the distance of communities from the wetted channel. A prevalence of early seral, facultative and facultative upland species at the water's edge could suggest a lowered water table and loss of riparian function. Conversely, it may be a natural response in reaches subject to repeat flood and deposition disturbance, such as on gravel bars in systems with regular inputs of sediment from the surrounding uplands. Another example would be the presence of species adapted to wet soils being found throughout the entire floodplain suggesting a shallow water table and good connectivity between the channel and floodplain.

In riparian settings, the composition of dominant species is due in part to proximity to the water table. A steep gradient typically exists with depth to groundwater increasing as one moves away from the edge of the water, especially in semi-arid regions (Coles-Ritchie et al. 2007). Shallow groundwater, typical of streambanks and low points within the active floodplain, primarily supports obligate and facultative wetland species (Martin & Chambers 2001; Stringham et al. 2001). Higher points in the floodplain often have a mix of facultative upland, facultative, and facultative wetland species; with facultative upland and upland species being common along the fringes of the riparian corridor (Crowe & Clausnitzer 1997; Baird et al. 2005). Identifying plant communities as related to fluvial landform (e.g. gravel bar, stream bank, floodplain) is helpful way to organize vegetation with direct applications to developing riparian STM states and phases, as well as potential thresholds between these elements. This is particularly true when layering in channel characteristics, which can greatly affect the extent of fluvial surfaces as well as connectivity to the floodplain (Crowe & Clausnitzer 1997; Stringham et al. 2001; Stringham & Repp 2010).

Vegetation Quantification: Fluvial Surfaces

To capture this aspect of riparian dynamics, the channel cross-sections were put to the dual purpose of collecting both channel and plant community data. For vegetation, this was accomplished by denoting along the tape the position and width of all encountered plant communities based upon changes in dominant species, thereby linking communities to their respective fluvial surfaces. The types of surfaces included in this project were edge of stream, stream bank, floodplain, overflow channel, terrace slope, and terrace.

Vegetation Quantification: Greenline Community

A more detailed breakdown of plant composition, as well as cover, was performed along the greenline found at each channel data collection point. The greenline community represents the first contiguous line of perennial vegetation encountered away from the edge of the channel (Winward 2000). The proximity to water tends to produce communities of deep-rooted plants that serve to stabilize banks and slow water velocity during periods of high flows (Hecker et al. 2019). In a system that can vary greatly in soil moisture, it also serves as a convenient location for monitoring, given the relative consistency of soil conditions (Cagney 1993). However, it is also the point of comparatively rapid change, being nearest to the active forces of moving water. This makes it ideal for detecting/predicting alterations in streambanks, channel morphology, and overall riparian condition (Burton et al. 2011). The width between greenline communities on opposing banks is also valuable information and can quickly inform managers of trends in riparian condition based upon whether this distance is increasing or decreasing over time. Disturbance, especially types that remove or weaken bank vegetation, typically leads to over-widened channels, while recovering or stable systems tend to see narrowing or relative static widths, respectively, over time (Clary 1999; Erskine et al. 2012). Data collection followed the guidelines presented in; *Multiple Indicator Monitoring (MIM) of Stream Channels and Streamside Vegetation* (Burton et al. 2011). In brief, this consisted of determining relative cover of herbaceous and woody species within 80 Daubenmire frames (40 on each bank) for a distance of 110 meters along each bank. At each of the first 40 plots, the distance between the near greenline and greenline on the opposite bank, was measured using a range finder. The Wetland Indicator Status (WIS) was determined for all recorded species (e.g. Obligate, Facultative Wetland, Facultative, Facultative Upland, Upland) in accordance with the Arid Region Wetland Plant List (USACE 2020).

From this data, an additional three factors were calculated and included with the list of analyzed environmental variables. These were Greenline to Greenline Width (GGW), Wetland Rating (WR) and Adjusted Species Score (ASS). These factors incorporate biotic elements, but are intrinsically tied to channel morphology. The GGW is simply the average distance between the greenline locations on each bank at each sampled location. The WR is a weighted average based upon the wetland indicator status of species from greenline plant community data (Upland = 1, Facultative Upland = 25, Facultative = 50, Facultative Wetland = 75, Obligate = 100). Weight was derived by multiplying the number of plots a species occurred within by the average percent cover and dividing by 100. The weighted average was calculated for each species by first multiplying the wetland indicator value by the weight. The sum of this product was then divided by the summed weights of each species. The ASS was calculated by first deriving an importance value by multiplying the percent frequency of each species by its average cover. These values were relativized by dividing them by the sum of all species importance values and then multiplied by the mean percent chance of the species occurring in a wetland (Obligate = 100, Facultative Wetland = 75, Facultative = 50, Facultative Upland = 25, Upland = 1) (Reed 1996). The resulting species scores were then summed to provide a wetland index score for each site. Using this system, a “1” would indicate that all species were classified as Upland and a score of “100”, the highest possible value, would be the result if all species were considered Obligate (Coles-Ritchie et al. 2007). As a final step, the species scores were adjusted by dividing a species’ Winward Stability Rating by 10 and then multiplying the species score by the quotient. The Windward Stability Rating is a rating between 1-10 and is indicative of a plant’s rooting structure ((0-3.9 = Low, 4-6 =

Moderate, >6 = High) (Windward 2000). Greenline data was not collected at sites CH2 and CH10, therefore these sites were excluded from any analyses where these three factors were incorporated.

Plant Community Components

Greenline and cross-sectional data were also used to develop plant community components (PCC). Riparian plant community composition is directly related to hydrologic regime, geomorphic setting, and the traits of the plants themselves (Ralston et al. 2014). In light of this, current riparian STMs have adopted the approach of delineating PCCs, based upon their fluvial location in relation to the channel and composition, in order to add a visual, relatively easy to measure level of detail to defined states and phases (Stringham & Repp 2010, Ratcliff et al. 2018; Meehan and O'Brien 2020). Additionally, monitoring of plant communities can provide a quick assessment of state transition. Thresholds between states are often more intuited than quantified given the inherent levels of complexity at the interface of vegetation, hydrology, and geomorphology. However, it is possible to recognize vegetative hallmarks exhibited in plant community components that signal an impending transition across a threshold. The initial PCCs were further refined by placing them within the context the different states determined during the model development portion of this study.

Riparian Ecosystem Services Brought into the STM Framework

Out of interest, it was decided to also collect and analyze information pertaining to bird diversity, stream temperature and soil carbon sequestration. The STM framework has inherent flexibility such that recognition of the processes at work in a given state (e.g. hydrology, flood plain development, plant community dynamics) can be applied to a broad range of associated ecosystem services. Quantifying these services and relating them to a particular state in the model provides important, additional information to managers leading to a greater understanding of how changing conditions can also affect elements like wildlife habitat and soil characteristics. Bird diversity was selected as a wildlife metric as data could be collected with a minimum of specialized equipment; birds are relatively easy to detect; and most riparian species are sensitive to habitat condition (Bradford et al. 1998; Bryce et al. 2002). An interest in stream temperature was driven by two factors. The first is that several tributaries of Maggie Creek support populations of a threatened status fish species, the Lahontan cutthroat trout. This trout has also been known to use the main stem of Maggie Creek to migrate between tributaries (Neville et al. 2016). Being able to determine if a given STM state was more conducive to maintaining desirable stream temperatures for this species would aid in its management. One portion of the studied stream passes through flood-irrigated hay meadows. Here, it was also decided to test whether differences in stream temperature could be detected resulting from inputs of cool, subsurface flows as irrigation water infiltrates into the soil and eventually returns to the stream. This information would prove useful in terms of land use and how it relates to aquatic habitat.

Soil is an important feature of carbon sequestration, which is of great interest to land managers and policy makers in the light of the increasing concentration of atmospheric CO2 (Lal 2002; Koch et al. 2013). The extent and potential for carbon storage generally increases along a gradient of dry to more mesic sites (Booker et al. 2013). This increases the importance of maintaining riparian areas in arid regions, where these systems often store a disproportionally high amount in relation to their geographical extent (Garrastazú et al. 2015). Being able to, at least loosely, tie potential carbon storage to states within the model could assist with the difficult process of assessing soil carbon amounts and remunerating land owners that are employing practices that facilitate healthy riparian systems. A co-occurring soil carbon study within the study area of this project provided a means to obtain data in exchange for aid in study setup and data collection (Morra et al. 2023).

Bird Point Counts

A growing body of research has sought to connect riparian condition to wildlife species diversity, density, and abundance (Warkentin & Reed 1999; McKinstry et al. 2001; Bryce et al. 2002; Nelson 2007; Kirkpatrick et al. 2009). Successful efforts have even been made to incorporate wildlife abundance and density into state and transition models, but so far, only in upland settings (Holmes & Miller 2010; Morzillo et al. 2014; Timmer et al. 2021). Given the vast array of wildlife species that utilize riparian/wetland habitat for at least a portion of their life cycle, there remains a strong need to include wildlife considerations into riparian-based management and tie them in with easily observable trends. To help fill this void, but simultaneously respecting time constraints, bird points counts were selected in part, due to their flexibility, ease of use, and inexpensive nature. They are also the most commonly used survey method for land birds and are also well-suited for linear features, such as riparian corridors (Strong & Bock 1990; Farnsworth et al. 2005; Matsuoka et al. 2014). Initial site visits to Maggie Creek (summer 2021), revealed an observationally high species richness at developed wetland locations, especially in beaver dam complex areas. Conversely, a paucity of bird species was

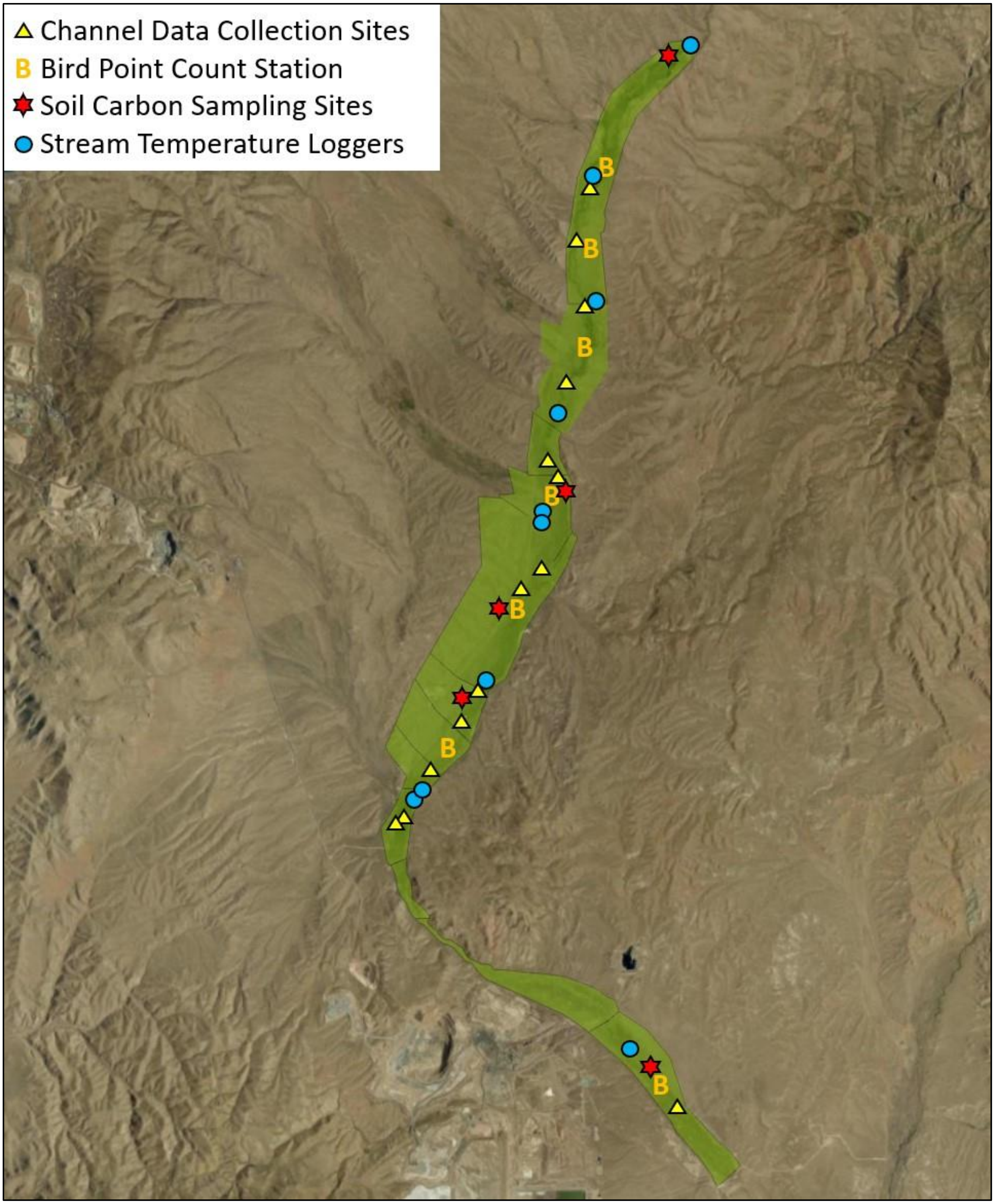


Figure 2.3. *Location of Sampling Locations for Channels, Birds, Stream Temperature and Soil Carbon*

noted in areas with little riparian development. To investigate this apparent pattern, and potentially link the data to individual STM states, bird point count stations were installed and monitored during the 2022-23 breeding seasons (May – July). Seven locations were selected that captured a range of different conditions and management within the study area (Figure 2.3). At each location, ten stations were installed within the riparian corridor for a total of 70 points. Each riparian station was paired with an upland station, installed 450 meters parallel to the respective riparian point within the adjacent sagebrush-dominated terrace. However, the terrace at the northernmost site consists of irrigated hay meadows. In order to avoid this different vegetation type, and disturbances associated with irrigation and hay mowing, this set of upland points were located along the toeslope of the valley wall. Sampling protocols followed the standardized guidelines developed for monitoring birds using points and subsequently adopted by the U.S. Fish and Wildlife Service (USFWS) (Ralph et al. 1995; Knutson et al. 2016). However, some deviation in sampling design was necessary. Double counting of individuals is a valid concern, and mediation usually involves separating sampling points by at least 250 meters, especially when the sampling format is inclusive of detections beyond 100 meters. This is not sufficient to capture the level of detail, such as changes in channel morphology and isolated beaver ponds, deemed necessary for ecological state correlation. A finer-scale sampling approach is documented in Kirkpatrick et al. (2009) where abundance, species richness, and reproductive success of riparian bird communities were assessed in light of ground water depletions and subsequent reductions in surface water. The study focused upon small perennial and intermittent streams and utilized a spacing of 100 meters. A GIS analysis of the roughly 20-mile long stretch of interest in Maggie Creek revealed the

100-meter distance to be suitable for covering specific locations (e.g. beaver wetland complexes, incised reaches, sites of active irrigation).

Ideally, the count radius would extend at least 100 meters both to increase total detections and to facilitate comparisons between other studies. However, when focusing on specific habitats, it is better to reduce edge effects by keeping the maximum radius within the desired habitat (Ralph et al. 1995). As riparian habitat in the study area seldom exceeds 50 meters in width, it was decided to restrict the count radius to 50 meters. A rangefinder was used to increase distance estimation accuracy. This arrangement also lowered the risk of double counting given that points were restricted to 100-meter spacing. Another benefit is that, in general, birds within a 50-meter radius are considered to be highly detectable, thus increasing the probability of detection (Ralph et al. 1995). Precedence for this alteration is documented in Lussier et al. (2006), a study that monitored bird populations along small streams in Rhode Island, and a riparian biodiversity study conducted along the Consumnes River in California (Dybala et al. 2018). The count radius was extended to 100 meters in the upland areas.

Evidence suggests that morning surveys alone capture a majority of species (Ralph et al. 1995). However, the wide diversity of habitat usually present in riparian areas lends itself to varied periods of peak activity by different bird assemblages. Including evening surveys with the more common morning surveys can lead to a truer, more complete picture of bird diversity (Sullivan & Vierling 2009). For this reason, and to reduce time investments by covering more areas per day, surveys took place during a 3-4-hour span beginning at sunrise and a 3-4-hour period concluding within 30 minutes of sunset. Data was collected approximately 10 meters distant from any running water to mitigate water-related noise. Birds flying over a station were noted, but unless they landed within the count radius, were not included as detections.

Originally, all points were to be sampled at least twice, swapping the starting and ending points of routes so that the beginning of one visit would be the ending point of the next visit. Likewise, a morning survey at one location would have shifted to an evening survey when next sampled. Unfortunately, time only permitted a full data set from all points during one period (Summer 2023). A partial set was collected in the summer of 2022.

Stream Temperature

Temperature maintenance is an important process for aquatic vertebrates and invertebrates, especially fish. The literature devoted to stream temperature details a variety of contributing factors to observed fluctuations and trends, including solar radiation, substrate material, flow volume, canopy cover, groundwater inputs, channel morphology, and channel bed structure. (Ward 1985; Poole & Berman 2001; Weber et al. 2017). To gain a sense of stream temperature dynamics in the study area, Onset HOBO® Pendant temperature loggers were installed at key locations during the spring of 2022 (Figure 2.3). One target of particular interest was the hayfield area at the north end of the study area. One study (Stringham et al. 1998), along with various anecdotal observations, suggest that flood irrigation systems can buffer summer stream temperatures. Water spreading across fields percolates downward, recharging the alluvial aquifer. The time lag between water diversion/spreading and eventual re-entry into the channel as ground water, allows for substantial heat exchange to occur yielding cooler water upon reemergence. Loggers were installed prior to the start of irrigation (mid-April) and allowed to collect data at hourly intervals on into the fall. One logger was placed ~100 meters above the irrigation diversion and one located near the bottom of the hayfield furthest downstream. Studies suggest similar mechanisms associated with beaver dams where ponded water spreads across adjoining floodplains and terraces (Westbrook et al. 2006; Karran et al. 2017). Hyporheic flow through the banks flanking a dam can also provide a measure of temperature amelioration (Westbrook et al. 2006; Johnson-Bice et al. 2018). In the study area, loggers were placed above and below beaver dams within large complexes and at small dams in newly established colonies as well as in abandoned complexes that still retain beaver-related features and structures (e.g. partial dams, old channels). To test the influence of tributaries, two primary side streams, Coyote and Jack Creeks, were bracketed with an upstream and downstream logger to determine if temperature differences exist as a result of surface flow inputs. At Coyote Creek, the loggers were separated by approximately one mile. This was done to capture the full spectrum of water inputs as Coyote Creek is a multithread channel at its lower end and has two primary channels that enter into Maggie Creek. This placement also covers the top and bottom of the Coyote Creek Pasture, allowing for an examination of stream temperatures in the context of grazing use. Additional loggers were installed in areas that coincided with potential states/phases to bolster the associated STM narrative (i.e. degraded F/G channel, D_A channel, C to E transition).

There is also a hope that this data will more broadly link stream condition, hyporheic flow dynamics, and land use to aquatic habitat quality. This is particularly salient in Maggie Creek which was historically part of the range of the federally-listed Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*). This species is now restricted to three of its larger tributaries, but fish make at least temporary use of Maggie Creek to migrate between tributaries (Neville et al. 2016). However, there is still concern over habitat shrinkage due to altered flow regimes and higher water temperature resulting from climate change (Wenger et al. 2011). Primary thermal refugia during warmer summer months is provided to cold water species, including Lahontan Cutthroat, by means of hyporheic inputs to the channel from subsurface sources (Boxall et al. 2007). Tracking temperature changes and buffering within the studied portion of Maggie Creek, which receives water from two of the current cutthroat-holding streams, may prove insightful in forecasting stream temperature trends.

Soil Carbon Sequestration

In the Morra et al. (2023) study, carbon stock information was gathered at five locations along Maggie Creek (Figure 2.3). Sampled features at each site included the active floodplain, terrace and corresponding upland. Four Locations correlated closely to CH1, CH6, CH7 and CH9 from this study and represent conditions under partial summer grazing, periodic spring/fall grazing and grazing exclusion (Figures 2.1 & 2.2). The fifth location was at the uppermost end of the hay meadows. The northernmost channel sampling location, CH14, is at the lower end of the hay meadows, but possesses similar characteristics in channel morphology and vegetation.

Statistical Analysis

All statistical analyses were performed using R version 4.3.2 (R Core Team 2023). The “vegan” package (Oksanen et al. 2022) was used for all ordinations, cluster analyses, Mantel tests, and Shannon Diversity Index. Hutcheson t-tests were computed courtesy of the “ecolTest” package (Salinas & Ramirez-Delgado 2021). The multipatt function from the “indicspecies” package (De Cáceres & Legendre 2009) handled all indicator species analysis. Elbow plots used to assess the optimal number of clusters for the hierarchical clustering were generated using the "factoextra" package (Kassambara & Mundt 2020). A correlation matrix for the PCA analysis was generated using the corrplot function from package "corrplot" (Wei & Simko 2021) and correlation significance levels were displayed via the rcorr function from the "Hmisc" package (Harrell 2024). A review of multivariate statistical approaches and considerations is presented in Appendix C.

Model Development – States and Phases

For initial detection of different states, hierarchical cluster analysis was used to group reaches based upon environmental factors and vegetation assemblages (See Table A.1, Appendix A for a list and description of environmental factors). Euclidean was selected as the distance measure for environmental variables, while Bray-Curtis was used for the species abundance data. To achieve tight clusters, the Ward algorithm was used in both cases. This method produces compact clusters, but does require distance measures to be metric. This necessitated applying a square root transformation to the species distances calculated with the Bray-Curtis coefficient (Zelený 2023). A Mantel test (Spearman correlation, 9,999 permutations) was used to determine if species composition patterns were correlated to sites grouped by environmental similarity.

Following methodology from Ratcliff et al. (2018), Mantel tests were also used to "prune" the hierarchical dendrograms. This was achieved by extracting the cluster assignment for each site from the analysis result and creating a new distance matrix. This matrix was then compared to the original distance matrix used by the clustering algorithm to calculate a Mantel test statistic. The Mantel test statistic represents the amount of correlation between two matrices and ranges between -1 and 1, with negative values indicating a negative correlation. The closer to -1 or 1, the stronger the correlation. The statistic is calculated by first “unfolding” one half of a distance matrix and coercing it into a column vector. Correlations are then derived between the vectors of each matrix (Buttigieg & Ramette 2014). By running an analysis for every possible number of clusters (i.e. maximum number is equal to the number of sites/samples), it was possible to test each result and identify the number of clusters with the highest statistic. For both cluster analysis and ordinations, the environmental matrix was limited to physical, channel metrics when analyzing data from all 14 sites. The three biotic-related factors (GGW, WR, and ASS) were added to the matrix when analyzing the 12 sites where greenline community data was collected.

Ordination consisted of an unconstrained Principle Component Analysis (PCA) to assess patterns in environmental variables and sites. Raw data was standardized to z-scores to allow for comparison of variables with different units and scales (Urban et al. 2002). The ordination diagram used a scaling of “2” in order to shift the focus to correlations between environmental variables. Axis evaluation was performed on resulting eigen values to determine the number of axes worth considering using a custom function from David Zelený’s “Analysis of community ecology data in R” website (Zelený 2023). Importance and relationship of environmental variables was assessed using Pearson’s correlations and significance level between variable pairs. A second, unconstrained ordination was performed on the species matrix associated with the greenline composition data to detect any plant community patterns in isolation from environmental variables. This was done using the defaults associated with the metaMDS function from the “vegan” package, which runs a Non-Metric Multidimensional Scaling (NMDS) technique using the Bray-Curtis dissimilarity coefficient, two dimensions, and automatically transforms the data set. To view the plant species from a functionality standpoint, the same ordination was also run substituting species names for their wetland indicator status. A final constrained ordination was performed, Canonical Correspondence Analysis (CCA), that incorporated both environmental and species data in order to assess which factors may be having a strong influence on species composition.

Bird Point Count Data

To test the hypothesis that bird diversity would be most similar between riparian and upland sites in degraded reaches, diversity values were calculated for each habitat type at each of the seven sampling locations. Diversity takes species richness and evenness into account and of the two common indices available, the Shannon Index emphasizes richness, whereas the Simpson Index is weighted more towards abundance of common species (DeJong 1975; Zelený 2023). For this project piece, the focus is more on which species are present between riparian and upland sites, than actual numbers. Thus, Shannon was selected as the index of choice. Index values were calculated for all riparian and upland sites and compared using the Hutcheson t-test. Comparisons of diversity were also made amongst the various riparian sites. Further exploration was done using indicator species analysis to check if any species seemed to preferentially prefer a specific habitat type, both generally and at specific sampling sites. In the western United States, over 60 percent of Neotropical migrants utilize riparia for breeding or stopover habitat, emphasizing the need to protect and enhance riparian areas and strengthens the suitability of incorporating bird census data into condition assessments (Rich 2002; Young et al. 2013). Identifying indicator species of functioning versus non-functioning riparian habitat, if any, would be a useful addition to the overall STM. To round out the avian analysis, Welch's t-test was used to test for differences in means of abundance between sites and habitat types. This test is preferred over the Student's t-test when variance is unequal (Ruxton 2006).

Stream Temperature

Graphical representations of stream temperature were created from logger data using Microsoft Excel spreadsheets. Maximum daily air temperatures were obtained from the National Weather Service and plotted along with water temperatures (NOAA 2024). The base R "stats" package was then employed to run Wilcoxon rank sum tests for data from paired loggers at the hay meadows areas and those bracketing the mouth of Coyote Creek (R Core Team 2023). The Wilcoxon method is a non-parametric way to test for significant differences between the means of two groups, which is appropriate as the data violated assumptions of normality and homoscedasticity.

Soil Carbon Sequestration

Statistical analyses were not performed for this portion of the study. General observations were made, and inferences drawn, placing the results of the carbon study within the context of land management and STM states. Attempts were made to explain the results, especially where study yielded unexpected sequestration values.

Results

Stream Classification

Initial data gathering and field reconnaissance suggested that the portion of Maggie Creek within the study area fit well with Rosgen's Valley Type VIII (Rosgen 2011). One exception is approximately two miles of stream that passes through a narrow canyon and was therefore excluded from the project. This area represents a second grazing exclosure implemented during the MCWRP and is labeled as “Exclusion No. 2” on Figure 2.1. VIII valleys are alluvial with a gentle gradient; well-developed floodplain; presence of stream terraces; and bound by colluvial slopes. They can be narrow in places, but tend to have ample room for stream movement and migration. Under functioning conditions, expected channel types are “C” (low gradient, well-defined meanders, slightly entrenched, moderate to high width/depth ratio, moderate to high sinuosity) and “E” (low gradient, meandering, slightly entrenched, very low width/depth ratio, high sinuosity, stable banks). It is also possible to have “D” (braided) or “D_A” (anastomosing) type channels which are marked by multiple thread channels and variable sinuosity and width/depth ratios. With deterioration, channels can incise forming gullies or “G” type channels with a low entrenchment ratio. An “F” type channel forms as the entrenched “G” channels begin to widen. Both “F” and “G” channels are considered to be unstable (Rosgen 1994; Rosgen 2011).

An analysis of cross-sectional and profile data revealed that “C” channels are currently the dominant type in the study area. Within this type, much variation exists (Table 2.1). Less stable reaches are more properly “C_F” types indicating that they still maintain many characteristics of an “F” channel (high width/depth ratio, moderate sinuosity), but have evolved towards higher entrenchment ratios and have more extensive and developed floodplains within the original incision banks. These types are typical for reaches that are recovering functionality, but are still experiencing active erosion/deposition and have yet to develop and sustain the type of stabilizing vegetation that promotes channel narrowing

Table 2.1. *Rosgen Classification of Sampled Stream Reaches and Channel Metrics*

Site	Channel Type[1]	ER[2]	WDR	SIN	BBW	BXA	MD	FPW	BHR	D50	GGW	WR	ASS
CH1	CF4	1.8	50.1	1.1	13.4	3.6	0.5	23.9	1.1	37	13.5	23.9	10.8
CH2	E6/C6	49.7	5.9	1.1	2.8	1.4	0.7	141	1.6	1.5	N/A	N/A	N/A
CH3	E6	109.3	3.8	1.1	1.8	0.9	0.6	200	1.1	1.5	3.0	96.5	70.8
CH4	C4	4.0	14.0	1.1	8.0	4.6	0.8	32	1.5	20	5.4	81.8	50.7
CH5	C4	8.7	35.3	1.3	16.4	7.6	0.8	142	1.1	20	9.7	79.4	43.5
CH6	C4	6.1	19.7	1.4	9.8	4.9	0.9	60	1.5	52	10.2	76.8	39.7
CH7	DA4	13.7	13.2	1.0	9.6	7.0	1.2	131	1.0	9	7.0	98.1	61.7
CH8	C4	3.8	32.0	1.51	5.4	0.9	0.3	21	1.3	25	4.4	82.7	28.7
CH9	E4	9.0	10.0	1.56	6.6	4.4	1.1	60	1.0	22	7.2	84.1	56.4
CH10	DA4	15.8	7.7	1.31	3.6	1.7	0.5	57	1.3	20	N/A	N/A	N/A
CH11	CE4	2.9	15.5	1.78	14.2	13.0	1.4	41	1.1	19	7.2	80.2	45.5
CH12	CE4	2.1	10.6	1.61	7.3	5.1	1.0	16	2.1	3	6.4	76.6	44.5
CH13	CE4	2.1	17.9	2.19	6.2	2.1	0.6	13	1.3	17	4.0	88.4	50.0
CH14	CE4	1.7	10.5	1.38	7.6	5.5	1.2	13	1.4	18	6.5	79.0	55.0

[1] All channel types are analogue (formed within incised channel banks) except for CH2 and CH3. Rosgen classification system allows for +/- 0.2 units for Entrenchment Ratio and Sinuosity and +/- 2.0 units for Width to Depth Ratio.

[2] ER = Entrenchment Ratio, WDR = Width to Depth Ratio, SIN = Sinuosity, BBW = Bankfull to Bankfull Width, BXA = Bankfull Cross-sectional Area, MD = Maximum Depth, FPW = Floodprone Width, BHR = Bank Height Ratio, D50 = Mean Particle Size, GGW = Greenline to Greenline Width, WR = Wetland Rating, ASS = Adjusted Species Score.

and deepening. On the opposite end of the spectrum, more stable “C_E” channels are present that are well on their way towards converting completely to “E” type channels. In the parlance of stream classification, virtually all stream types in the study area are considered to be stabilizing analogues. In this context, this term refers to channels that have incised below their original floodplain, but have since widened out and are now in the process of building a new floodplain within the incised banks. A stable analogue is reached at the “end” of this process and is considered a narrower, more confined reflection of conditions prior to the disturbances that led to channel erosion (Harvey & Watson 1986; Cluer & Thorne 2014). Analogue versions of “D_A” channels also occur in reaches currently hosting beaver wetland complexes. Numerous avulsions form when impounded water finds new flow paths, often reconnecting formerly abandoned channels.

Small, beaver-created lateral channels are also common and are used to provide beavers cover while expanding into new areas of food and building material (Westbrook et al. 2006; Hood & Larson 2014). An example of analogue development can be seen in Stream Evolution Model of Cluer & Thorne (2014). Channel incision occurs in Stages 3-4, followed by a widening and aggrading phase in Stage 5. Stabilization occurs when quasi-equilibrium is reached in Stage 6 and an anastomosed, multi-thread channel is possible in Stage 8 (Figure 2.4).

True “C”, “E”, “F”, and “G” channels occur in only one portion of the studied area and are once again, directly related to beaver activity. In this case, a series of large, long-term dams have effectively shifted water out of the incised channel to form a marshy wetland to the west. This wetland is also fed by a tributary coming in from the west side of the valley. The combined waters leaving the wetland spread out across a meadow and eventually re-enter the older stable analogue C channel approximately one mile downstream. Viewing historic aerial imagery, it appears that flows became channelized at the lower end of the meadow, possibly from following scour marks left by old floods, around 30 years ago. The channelized flow dropping off of the meadow surface into the stabilized analogue channel created a knickpoint that has formed a headcut that is actively migrating up the central region of the meadow (Photo B1, Appendix B). Above the headcut, a deep, narrow “E” channel extends upstream for ~250 meters where it becomes a “D_A” reach at the downstream end of the beaver wetland area. Below the headcut, a classic gully-type “G” channel exists which transitions to an “F” and then a “C” channel as it nears the older, former channel. Two sample locations were placed within this wetland system in attempt to characterize both a functioning, non-degraded reach and one that is actively incising (CH1, CH2).

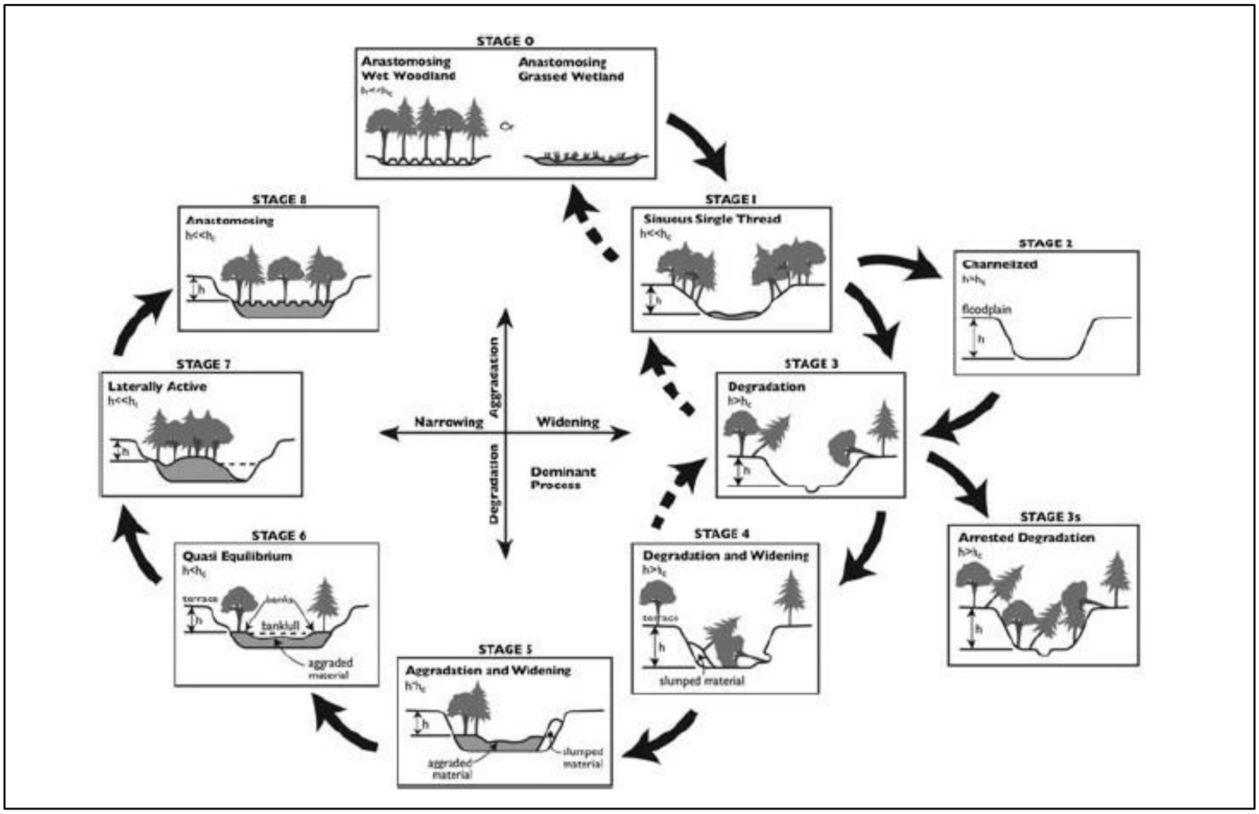


Figure 2.4. *Stream Evolution Model (Cluer & Thorne [2014]).*

The first mission was a success, the second, not so much. The longitudinal profile of the site furthest downstream (CH2) spans a distance from about 40 meters above the headcut to approximately 60 meters below this feature. The cross-section was placed 20 meters from the end of the profile where the channel appeared to have developed into a gully or "G" type channel. However, later calculations revealed a high ER (49.7), low WDR (5.9) and moderate sinuosity, which indicates a transitional stream denoted by an "E_C", or "E" channel going towards a "C" channel (Table 2.1). In other words, it was placed too far upstream to successfully capture a true "G" to "F", actively degrading scenario.

Vegetation Quantification – Fluvial Surfaces

The plant communities encountered along each cross-section, along with channel type, wetland indicator status by species, sediment size class and fluvial surface, are listed in Table 2.2.

Table 2.2. *Plant Communities by Fluvial Surface within the Study Area*

Site	**Channel[1] Type**	**Dominant Species[2]**	**Fluvial Surface**					
			Edge of Stream	**Stream Bank**	**Floodplain**	**Overflow Channel**	**Terrace Slope**	**Top of Terrace**
CH1	C_F4	*Agrostis stolonifera* (FACW)	Y					
		Equisetum arvense (FAC)	Y					
		Eleocharis palustris (OBL)	Y					
		Bare Ground		Y				
		Bromus tectorum (UPL)		Y			Y	
		Cobble			Y			
		Leymus triticoides (FAC)					Y	Y
		Iva axillaris (FACU)					Y	
		Onopordum acanthium (FACU)					Y	
		Artemisia tridentata ssp. *tridentata* (UPL)						Y
		Leymus cinereus (FAC)						Y
CH2	E_C6	*Carex nebrascensis* (OBL)		Y	Y			
		Argentina anserina (OBL)	Y					
		Lactuca seriola (FACU)	Y	Y				
		Symphyotrichum ascendens (FAC)		Y	Y			
		Juncus arcticus ssp. *littoralis* (FACW)			Y			
CH3	E6	*Veronica americana* (OBL)	Y					
		Eleocharis palustris (OBL)	Y					
		Carex nebrascensis (OBL)		Y	Y			
		Deschampsia cespitosa (FACW)			Y			
		Senecio integerrimus (FACU)		Y				
		Melilotus officinalis (FACU)		Y				
		Juncus arcticus ssp. *littoralis* (FACW)			Y			
CH4	C4	*Eleocharis palustris* (OBL)	Y					
		Salix lutea (OBL)		Y	Y			
		Carex pellita (OBL)		Y				
		Salix exigua (FACW)			Y	Y		
		Ericameria nauseosa (FACU)					Y	Y
		Bromus tectorum (UPL)					Y	
		Artemisia tridentata spp. *tridentata* (UPL)						Y
CH5	C4	*Salix exigua* (sapling) (FACW)	Y	Y				
		Argentina anserina (OBL)	Y					
		Salix lutea (sapling) (OBL)		Y	Y			
		Carex pellita (OBL)		Y	Y	Y		
		Salix exigua (mature) (FACW)			Y			
		Solidago lepida (FAC)				Y		
		Ericameria nauseosa (FACU)					Y	Y
		Artemisia tridentata spp. *tridentata* (UPL)					Y	Y
		*Leymus triticoides (*FAC)					Y	Y
		Carex praegracilis (FACW)					Y	Y
CH6	C4	Gravel	Y					
		Salix exigua (sapling) (FACW)		Y				
		Schoenoplectus americanus (OBL)		Y				
		Eleocharis palustris (OBL)		Y				

		Juncus arcticus ssp. *littoralis* (FACW)		Y				
		Salix lutea (OBL)			Y			
		Salix exigua (mature) (FACW)			Y		Y	Y
		Carex pellita (OBL)			Y			
		Leymus triticoides (FAC)					Y	Y
		Artemisia tridentata spp. *tridentata* (UPL)						Y
CH7	D_A4	*Typha latifolia* (OBL)	Y	Y	Y	Y		
		Eleocharis palustris (OBL)	Y					
		Potentilla biennis (FACW)		Y				
		Mentha arvensis (FACW)		Y				
		Carex pellita (OBL)			Y	Y		
		Cicuta douglasii (OBL)			Y	Y		
		Carex nebrascensis (OBL)			Y			
		Juncus arcticus ssp. *littoralis* (FACW)			Y	Y		
		Salix exigua (FACW)				Y		
		Ericameria nauseosa (FACU)					Y	Y
		Iva axillaris (FACU)					Y	Y
		Leymus triticoides (FAC)					Y	Y
		Poa pratensis (FAC)					Y	
		Artemisia tridentata spp. *tridentata* (UPL)						Y
CH8	C4	*Catabrosa aquatica* (OBL)	Y					
		Carex nebrascensis (OBL)		Y		Y		
		Agrostis stolonifera (FACW)		Y				
		Argentina anserina (OBL)		Y		Y		
		Schoenoplectus americanus (OBL)		Y				
		Hordeum brachyantherum (FACW)		Y				
		Artemisia cana (FACU)			Y			
		Ericameria nauseosa (FACU)			Y		Y	Y
		Carex praegracilis (FACW)			Y	Y		
		Juncus arcticus ssp. *littoralis* (FACW)				Y		
		Artemisia tridentata spp. *tridentata* (UPL)					Y	Y
		Carex douglasii (FAC)					Y	
CH9	E4	*Carex pellita* (FAC	Y	Y		Y		
		Salix lutea (OBL)	Y	Y				
		Salix exigua (FACW)	Y	Y	Y			
		Eleocharis palustris (OBL)	Y					
		Argentina anserina (OBL)		Y				
		Leymus triticoides (FAC)			Y		Y	
		Juncus arcticus (OBL)			Y			
		Mesic Forbs (early seral)			Y			
		Poa pratensis (FAC)				Y		
		Artemisia tridentata spp. tridentata (UPL)					Y	Y
		Ericameria nauseosa (FACU)					Y	Y
		Iva axillaris (FACU)						Y
		Leymus cinereus (FAC)						Y
CH10	D_A4	*Typha latifolia* (OBL)	Y		Y			
		Eleocharis palustris (OBL)	Y					
		Carex pellita (OBL)		Y		Y		
		Carex praegracilis (FACW)		Y				
		Poa pratensis (FAC)		Y				
		Juncus arcticus ssp. *littoralis* (FACW)		Y		Y		
		Agrostis stolonifera (FACW)		Y				
		Salix exigua (FACW)			Y	Y		
		Cirsium arvense (FACU)			Y	Y		
		Lactuca seriola (FACU)			Y			
		Leymus triticoides (FAC)			Y		Y	Y
		Ericameria nauseosa (FACU)					Y	Y
		Artemisia tridentata spp. *tridentata* (UPL)						Y
		Leymus cinereus (FAC)						Y
CH11	C_E4	*Agrostis stolonifera* (FACW)	Y					
		Veronica americana (OBL)	Y					
		Polypogon monospeliensis (FACW)	Y					
		Salix lutea (OBL)		Y		Y	Y	

		Salix exigua (FACW)		Y	Y	Y	Y	
		Carex pellita (OBL)		Y				
		Juncus arcticus ssp. *littoralis* (FACW)		Y				
		Bromus ciliatus (FAC)			Y	Y		
		Carex praegracilis (FACW)			Y			
		Rosa woodsii (FACU)				Y		
		Leymus cinereus (FAC)					Y	Y
		Ericameria nauseosa (FACU)						Y
		Mesic Forb (early seral)						Y
CH12	C$_E$4	*Eleocharis palustris* (OBL)	Y					
		Equisetum arvense (FAC)	Y	Y				
		Salix exigua (FACW)		Y	Y			
		Carex pellita (OBL)		Y	Y			
		Juncus arcticus ssp. *littoralis* (FACW)		Y				
		Symphyotrichum spathulatum (FAC)		Y				
		Salix exigua (saplings) (FACW)					Y	
		Rosa woodsii (FACU)			Y		Y	
		Leymus triticoides (FAC)					Y	
		Leymus cinereus (FAC)						Y
		Bromus tectorum (UPL)						Y
		Onopordum acanthium (FACU)						Y
CH13	C$_E$4	*Salix exigua* (FACW)	Y	Y	Y	Y		
		Agrostis stolonifera (FACW)	Y					
		Epilobium ciliatum (FACW)	Y					
		Glyceria grandis (OBL)	Y					
		Eleocharis palustris (OBL)	Y					
		Salix lutea (OBL)		Y	Y			
		Carex pellita (OBL)		Y				
		Juncus arcticus ssp. *littoralis* (FACW)		Y				
		Carex nebrascensis (OBL)		Y				
		Rosa woodsii (FACU)			Y	Y		
		Leymus triticoides (FAC)			Y	Y		Y
		Carex praegracilis (FACW)				Y		
		Artemisia tridentata spp. *tridentata* (UPL)					Y	Y
		Leymus cinereus (FAC)					Y	Y
		Artemisia ludoviciana (FACU)					Y	
		Onopordum acanthium (FACU)					Y	
		Agropyron cristatum (UPL)						Y
		Ericameria nauseosa (FACU)						Y
		Iva axillaris (FACU)						Y
CH14	C$_E$4	*Salix exigua* (FACW)	Y	Y	Y			Y
		Rosa woodsii (FACU)	Y	Y	Y		Y	
		Carex pellita (OBL)	Y	Y	Y			
		Equisetum hyemale (FACW)		Y			Y	
		Leymus cinereus (FAC)					Y	Y
		Symphyotrichum spathulatum (FAC)					Y	

[1] Number indicates sediment size class (4 = gravel, 6 = clay)

[2] Wetland Indicator Rating – Arid West Region (OBL = obligate, FACW = Facultative Wetland, FAC = Facultative, FACU = Facultative Upland, UPL = Upland)

All sampled reaches but two (CH2 & CH3, see Figure 3) are located within the confines of incised channel banks. The terrace above the current channel is fairly consistent in species composition, with *Artemisia tridentata* ssp. *tridentata* (basin big sagebrush) and *Ericameria nauseosa* (rubber rabbitbrush) being the dominant canopy species. Typical understory species include *Leymus cinereus* (basin wildrye), *L. triticoides* (creeping wildrye) and *Iva axillaris* (poverty sumpweed). In sections where the water table has risen due to beaver activity, facultative wetland species, such as *Carex praegracilis* (clustered field sedge) and *Salix exigua* (coyote willow) may also be well represented within the community. The terrace at the four northern-most sites (CH11-CH14) experienced wildfire in 2006 and 2017 converting the shrubland to a grassland dominated by basin wildrye and creeping wildrye. Where the terrace still represents the active floodplain, as is the case at CH2 and CH3, the community is well-watered and supports a near monoculture of *Carex nebrascensis* (Nebraska sedge). Terrace slopes are populated with communities representing a transition between the upland terrace and floodplain. They typically contain a mix of species from both features and plants adapted to drier conditions, such as basin big sagebrush, may drop out. They can also have a high proportion of grass species, particularly creeping wildrye, basin wildrye, and *Poa pratensis* (Kentucky bluegrass). Active floodplain communities range from nearly depauperate beds of gravel/cobble to dense thickets of multi-aged stands of willows. Herbaceous species are typically under-represented, except for at beaver-created wetlands found along the CH7 and CH10 reaches. These sites are marked by large populations of *Typha latifolia* (broadleaf cattail), *Schoenoplectus* ssp. (bulrush), *Carex* spp. (sedge), and various forbs capable of living in soils saturated for much of the growing season.

The stream bank, due to consistent observed differences in species composition, was divided into two portions; the “edge of stream”, from the edge of the wetted channel at low flows up to the bankfull line, and the “stream bank” consisting of that portion from bankfull up to the top of the bank. At some sites, the streambank community was present along both sides of the stream at the location of the cross-section. Other sites only had one true bank where deposition was occurring. Active erosion on the opposite side created a vertical face rising directly from the edge of the water up to the top of the terrace. The streambank communities were most often comprised of a mix of coyote and yellow willow with an understory of woolly sedge, *Juncus arcticus* ssp. *littoralis* (mountain rush) and forbs such as *Equisetum hyemale* (scouring rush horsetail), *Symphyotrichum spathulatum* (western mountain aster). Exceptions were site CH8 which is within a grazing exclosure water gap and experiences regular summer use, CH1 which also undergoes grazing during a portion of the summer, and very wet sites such as the beaver wetland complex at CH7 and the true "E" channel site at CH3. Stream edge communities were comprised primarily of early seral graminoids like *Agrostis stolonifera* (creeping bentgrass), *Eleocharis palustris* (common Spikerush), *Muhlenbergia asperifolia* (scratchgrass) and *Hordeum brachyantherum* (meadow barley); and forbs such as *Melilotus officinalis* (yellow sweetclover), *Argentina anserina* (silverweed cinquefoil) and *Veronica americana* (American speedwell). Overflow channels were not present at all sites, but where they occurred, they often supported a suite of species similar to that found closer to the channel, regardless of their distance from the channel.

Vegetation Quantification – Greenline Community

Assessment of the greenline community data showed that wetland index scores were between 76-98 for all sites except for CH1 (24.6) (Table 2.3). Applying the stability adjustment, the highest score was for CH3 (70.8), followed by CH7 (61.7). The majority of sites fell between 40-56, though CH1 fell to 10.8. Of particular note was CH8, which dropped dramatically from 82.7 to 28.7.

Table 2.3. *Greenline Wetland Index and Stability Scores*

Site	**Channel Type**	**Wetland Index Score**	**Stability Adjusted Score**
CH1	C_F4	24.6	10.8
CH2	E_C6	No Greenline Data	No Greenline Data
CH3	E6	96.5	70.8
CH4	C4	81.9	50.7
CH5	C4	79.5	43.5
CH6	C4	76.9	39.7
CH7	D_A4	98.1	61.7
CH8	C4	82.7	28.7
CH9	E4	84.2	56.4
CH10	D_A4	No Greenline Data	No Greenline Data
CH11	C_E4	80.2	45.5
CH12	C_E4	76.6	44.5
CH13	C_E4	88.4	49.7
CH14	C_E4	79.0	55.0

Model Development – States and Phases

Hierarchical Classification

Visually, the hierarchical classification of sites grouped by physical factors for all 14 sites formed four distinct clusters. An elbow plot, displaying total within-cluster sum of squares, showed that the rate of decline starts to taper off between four and five clusters. Pruning the dendrogram using the Mantel test showed the highest correlation for four clusters (R = 0.57), followed by two clusters (R = 0.49) and then three clusters (r = 0.41) (Table 2.4).

Table 2.4. *Mantel Test Correlations for Pruning Environmental Cluster Dendrogram for all sites*

# of Clusters	Mantel R Statistic	P Value
2	0.49	0.01
3	0.41	$5e^{-4}$
4	0.57	$5e^{-4}$
5	0.27	0.03
6	0.28	0.03
7	0.29	0.03
8	0.25	0.03
9	0.22	0.06
10	0.24	0.05
11	0.15	0.12
12	0.14	0.14

With four clusters, the two meadow sites (CH2 and CH3) constitute one cluster; CH5, CH7 and CH11 are a second cluster; CH1 is broken out by itself and the remaining sites fall into the third grouping. With three clusters, CH1 is added to the CH5-CH7-CH11 cluster. Five clusters yield the same as the four-cluster arrangement with an additional group formed by subdividing the largest cluster (Figure 2.5). There is little change that occurs when the two sites lacking greenline data (CH2 and CH10) are removed and the remaining sites are analyzed with the inclusion of the three biotic-related factors (GGW, WR, and ASS) (Figure 2.6, left side). The optimal number of clusters is still four according to the elbow plot. The Mantel test statistic was slightly higher for three groups (0.77) than four (0.74), but with the four clusters the associated p-value was much lower (0.0003 vs 0.02) (Table 2.5). Regardless, CH1 and CH2 remain in their own, segregated groups with either three clusters or four clusters. Scaling up from three clusters simply subdivides the large group bearing all other sites.

The greenline species data produced an elbow plot that suggested an optimum number of four clusters. Mantel correlations were also highest for four clusters (0.78), then three clusters (0.73), followed by six clusters (0.69). Cluster membership is similar to that observed with the environmental data. However, CH8 is brought out of the largest cluster of sites and placed into its own cluster when cluster size is set to four or five. As before, CH1 is its own cluster when sites are divided into either four or five groupings. Another minor difference with species data is that CH7 is now included with CH3 at three, four, or five clusters (Figure 2.6).

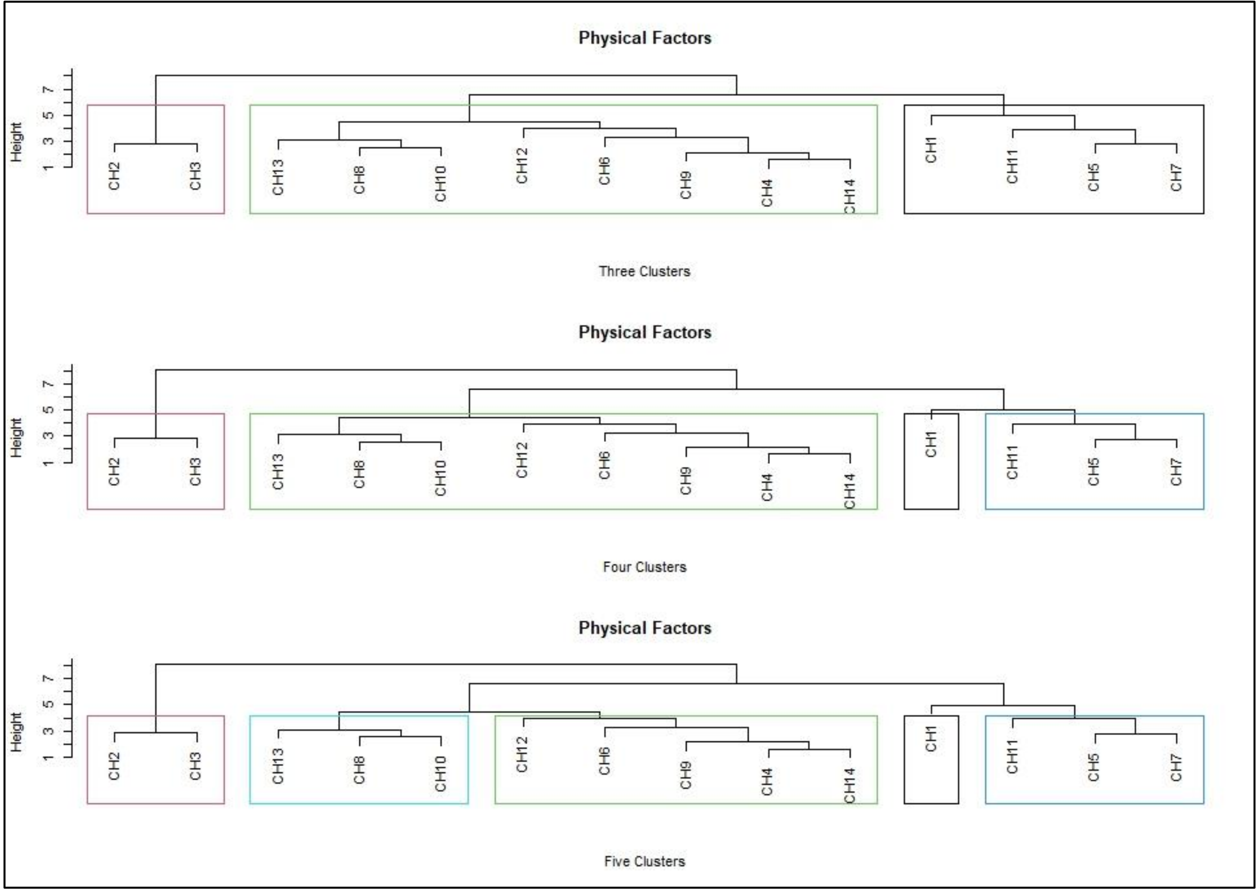


Figure 2.5. *Hierarchical Cluster Analysis of Physical Factors (14 Sites)*

Table 2.5. *Mantel Test Correlations for Pruning Environmental Cluster Dendrogram for 12 Sites*

# of Clusters	Mantel R Statistic	P Value
2	0.53	0.09
3	0.77	0.02
4	0.74	0.0003
5	0.57	0.002
6	0.55	0.01
7	0.43	0.03
8	0.24	0.09
9	0.30	0.20
10	0.20	0.05

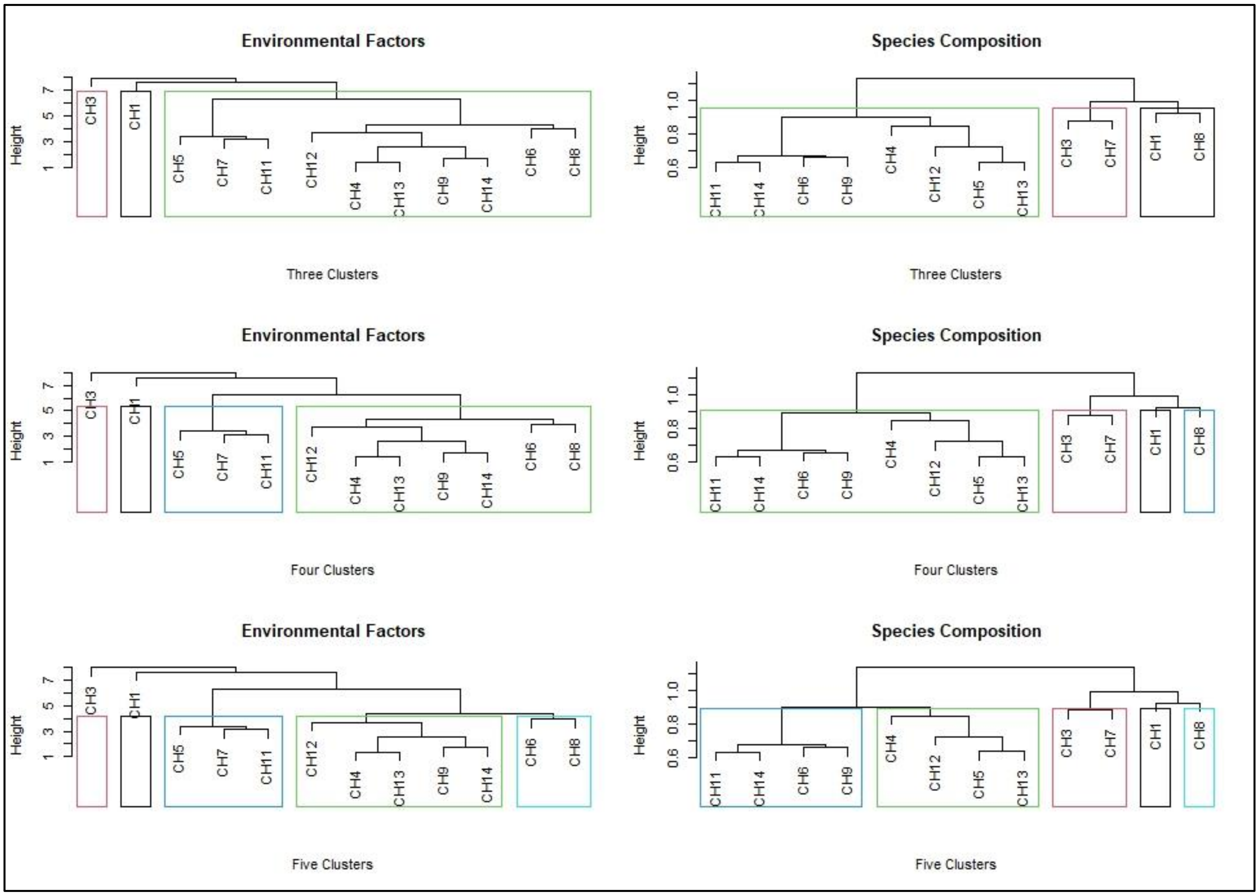
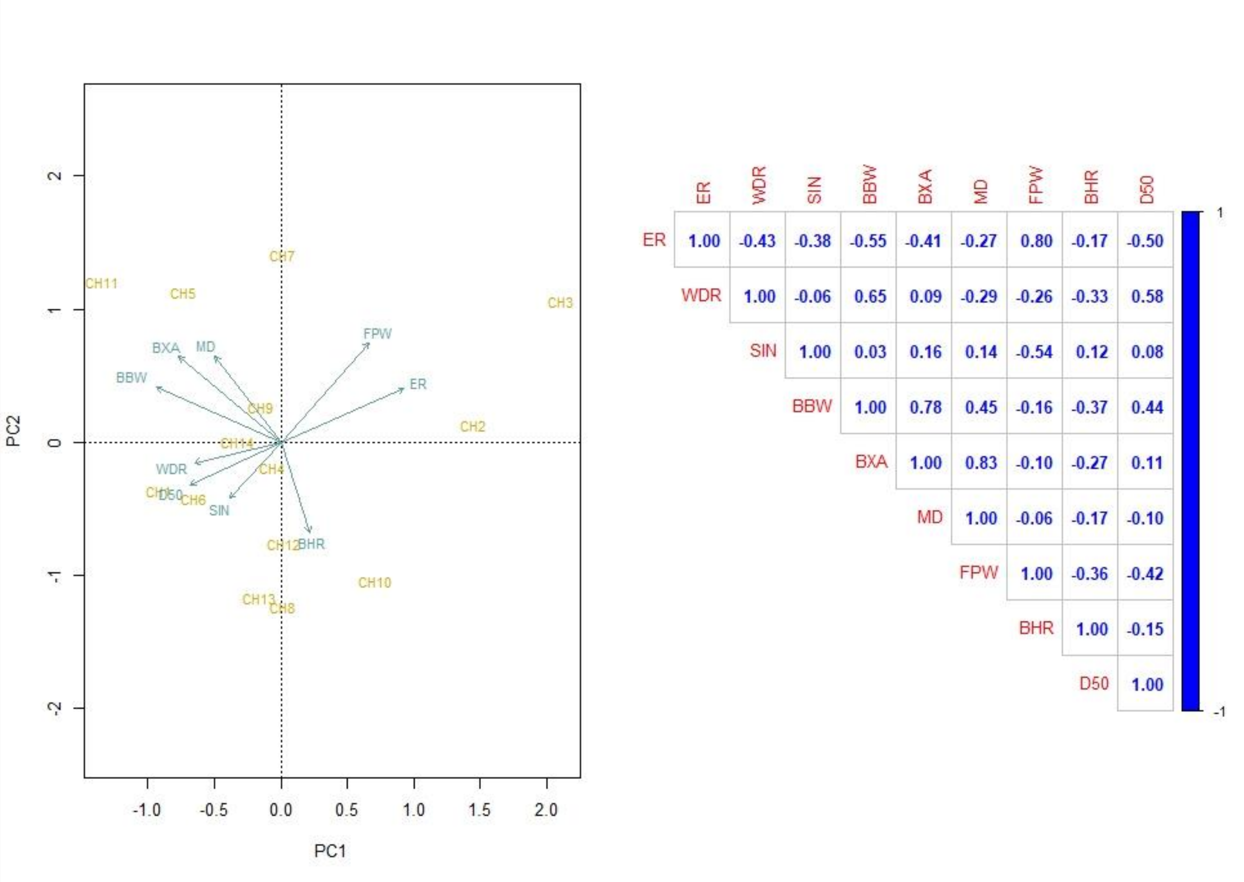


Figure 2.6*. Hierarchical Cluster Analysis of Environmental Factors and Species Composition (12 Sites)*

Based on the mantel tests and cluster analysis dendrograms, four clusters were selected as the most appropriate grouping for sampled reaches. This is not indicative of four separate states, but rather marks the entry point in pattern analysis for developing states and shows how, based on environmental and vegetation data, various sites are related to each other. The next step was to look for agreement, or additional signatures within ordination space.

Ordinations

Eigenvalue evaluation of the PCA of physical variables for all 14 sites indicated that the first three axes were important for interpretation. This was also true when including the three biotic-related factors for 12 sites. However, the default PCA diagram only displays the first two axes so as to reduce the level of visual complexity (Figures 2.7 & 2.8).



	ER	WDR	SIN	BBW	BXA	MD	FPW	BHR	D50
ER	1.00	-0.43	-0.38	-0.55	-0.41	-0.27	0.80	-0.17	-0.50
WDR		1.00	-0.06	0.65	0.09	-0.29	-0.26	-0.33	0.58
SIN			1.00	0.03	0.16	0.14	-0.54	0.12	0.08
BBW				1.00	0.78	0.45	-0.16	-0.37	0.44
BXA					1.00	0.83	-0.10	-0.27	0.11
MD						1.00	-0.06	-0.17	-0.10
FPW							1.00	-0.36	-0.42
BHR								1.00	-0.15
D50									1.00

Figure 2.7. *PCA Ordination Plots and Pearson's Correlation Matrix (14 Sites)*

Explained variance of each axis for the 14-site PCA are as follows: Axis 1 = 38%, Axis 2 = 23% and Axis 3 = 20%. The variables with the highest correlation to the first axis were ER (+95%), BBW (-91%), and BXA (-75%). Second axis correlations were strongest for

MD (+88%), BXA (+77%) and FPW (+53%) . The BHR (+0.75%), WDR (-65%) and SINUOSITY (+0.65) variables are most associated with the third axis (Table 2.6).

Table 2.6. *Variable scores from PCA Ordination (14 Sites)*

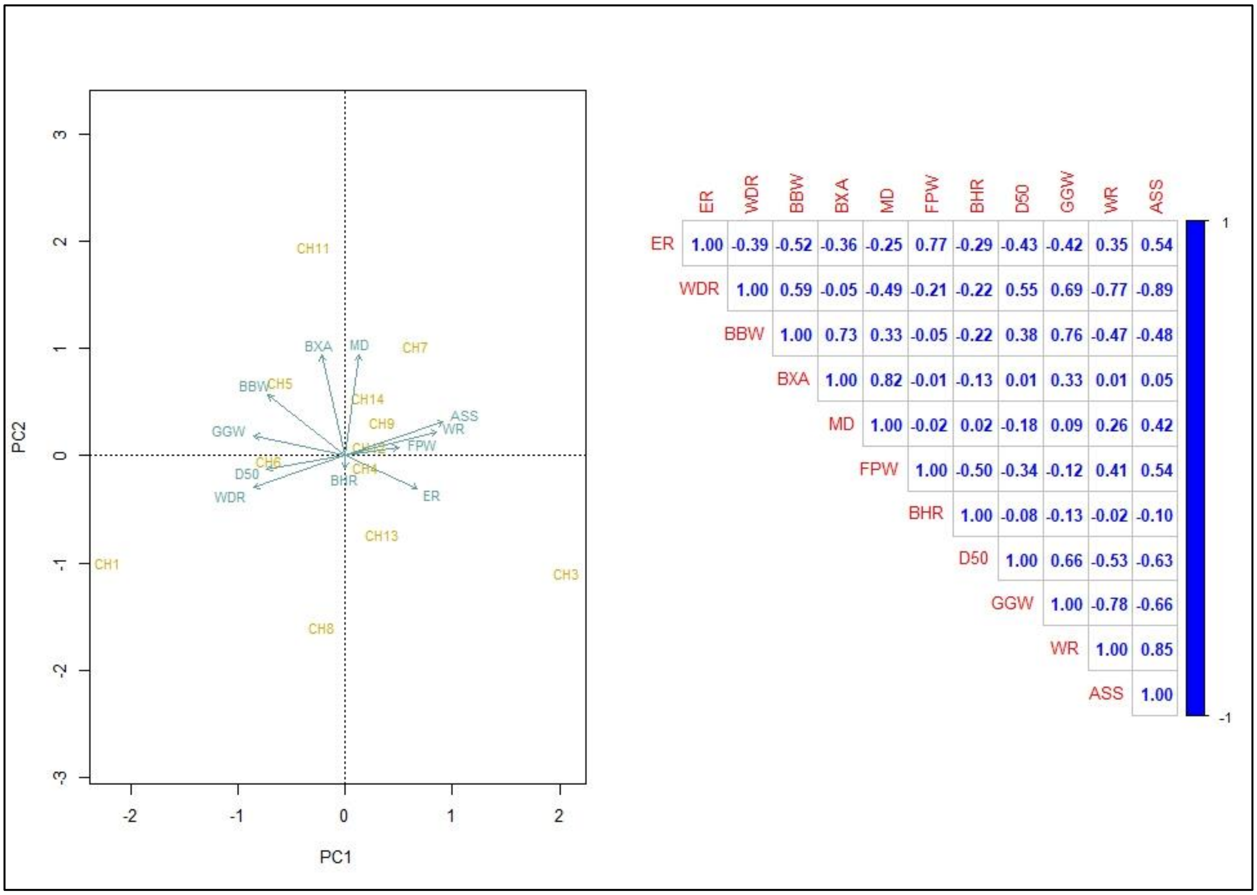
Variable	PCA Axis 1	PCA Axis 2	PCA Axis 3
ER	0.95	0.24	-0.28
WDR	-0.63	-0.48	-0.65
SINUOSITY	-0.40	-0.16	0.65
BBW	-0.91	0.30	-0.42
BXA	-0.75	0.77	0.01
MD	-0.47	0.88	0.34
FPW	0.70	0.53	-0.56
BHR	0.09	-0.26	0.75
D50	-0.68	-0.48	-0.34

High positive variable correlations were seen between BXA and MD (r = 0.83), ER and FPW (r = 0.80) and BBW and BXA (r = 0.78). Negative correlations were most evident between ER and BBW (r = -0.55) (Figure 2.7).

The following values were returned for explained variance of each axis of the 12-site PCA: Axis 1 = 44%, Axis 2 = 23% and Axis 3 = 17%. Variables most associated with the first axis were ASS (+92%), WDR (-86%), WR (+85%), and GGW (-85%). The second axis is correlated most strongly to MD (+94%) and BXA (+94%), while BHR (+82%) has a correlation best exhibited along the third axis (Table 2.7). Variable correlation showed strong, positive relationships between WR and ASS (r = 0.85); BXA and MD (r = 82%); ER and FPW (r = 0.77); BBW and GGW (r = 0.76) and BBW and BXA (r = 0.73). Strong negative relationships exist between WDR and ASS (r = -0.89); GGW and WR (r = -0.78); WDR and WR (r = -0.77) and GGW and ASS (r = -66) (Figure 2.8).

Table 2.7. *Variable scores from PCA Ordination with Additional Biotic Metrics (12 Sites)*

Variable	PCA Axis 1	PCA Axis 2	PCA Axis 3
ER	0.67	-0.30	-0.54
WDR	-0.86	-0.30	-0.30
BBW	-0.72	0.57	-0.28
BXA	-0.22	0.94	-0.05
MD	0.13	0.94	0.17
FPW	0.50	0.07	-0.80
BHR	-0.01	-0.11	0.82
D50	-0.73	-0.12	-0.05
GGW	-0.85	0.19	-0.28
WR	0.85	0.23	0.07
ASS	0.92	0.32	-0.07



	ER	WDR	BBW	BXA	MD	FPW	BHR	D50	GGW	WR	ASS
ER	1.00	-0.39	-0.52	-0.36	-0.25	0.77	-0.29	-0.43	-0.42	0.35	0.54
WDR		1.00	0.59	-0.05	-0.49	-0.21	-0.22	0.55	0.69	-0.77	-0.89
BBW			1.00	0.73	0.33	-0.05	-0.22	0.38	0.76	-0.47	-0.48
BXA				1.00	0.82	-0.01	-0.13	0.01	0.33	0.01	0.05
MD					1.00	-0.02	0.02	-0.18	0.09	0.26	0.42
FPW						1.00	-0.50	-0.34	-0.12	0.41	0.54
BHR							1.00	-0.08	-0.13	-0.02	-0.10
D50								1.00	0.66	-0.53	-0.63
GGW									1.00	-0.78	-0.66
WR										1.00	0.85
ASS											1.00

Figure 2.8*. PCA Ordination Plots and Pearson’s Correlation Matrix (12 Sites)*

Viewing the ordination diagram for all 14 sites, increases in FPW and ER are associated with the CH2 and CH3 sites. As these values decrease, the association switches to CH1 and CH6, which are also positively correlated with WDR, D50 and to a lesser degree, sinuosity. Along the second axis, CH8, CH10, CH12 and CH13 correlate with increased in BHR (Figure 2.7). This changes little when examining the ordination diagram for 12 sites. Two of the biotic-related variables (ASS and WR) are strongly correlated with increases in FPW, while the third, GGW, trends with increases in WDR and BBW. In general, the “E”, “D$_A$” and “CE” channel types found at C3 and many of the more northerly sites are tied to higher values of ASS and WR. Less developed “C”/ “CF” channels are prone to higher WDR and GGW values (Table 2.8).

Table 2.8. *List of Environmental Variables Most Associated with Sample Reach Channel Types*

Channel Type	**Sampling Location**	**Associated Environmental Variables**[1]
Analogue “C$_F$4”	CH1	-FPW, -ER, +WDR, +D50, +GGW, -ASS, -WR
Analogue “C4”	CH4-6, CH8	+WDR, +GGW
Analogue “C$_E$4”	CH11-14	+ASS, +WR, +SINUOSITY, -ER
Analogue “E4”	CH9	-BHR, +MD, +BXA
Analogue “D$_A$4”	CH7, CH10	+FPW, +WR, +ASS
“E6”	CH3	+FPW, +ER, -WDR, -D50
“EC6”	CH2	+FPW, +ER, -WDR, -D50

[1] A “+” indicates that associated variables increase at a given channel type. A “-” indicates that variables decrease

The NMDS ordination of greenline species data yielded a solution with a stress value of 0.14, indicating that the fit between the original matrix and assigned ranks is not particularly tight, but the ordination is still suitable for assessing general patterns (Figure 2.9). Immediately noticeable is a cluster of upland species associated with the CH1, “C$_F$” site and a second cluster of obligate wetland species around the “D$_A$” channel at CH7. There is also a broader trend of obligate and facultative wetlands species appearing in conjunction with “E” and “C_E” channels. Beyond this, sampled reaches tend to have a fairly heterogenous mix of species with varying degrees of affinity for wet soils.

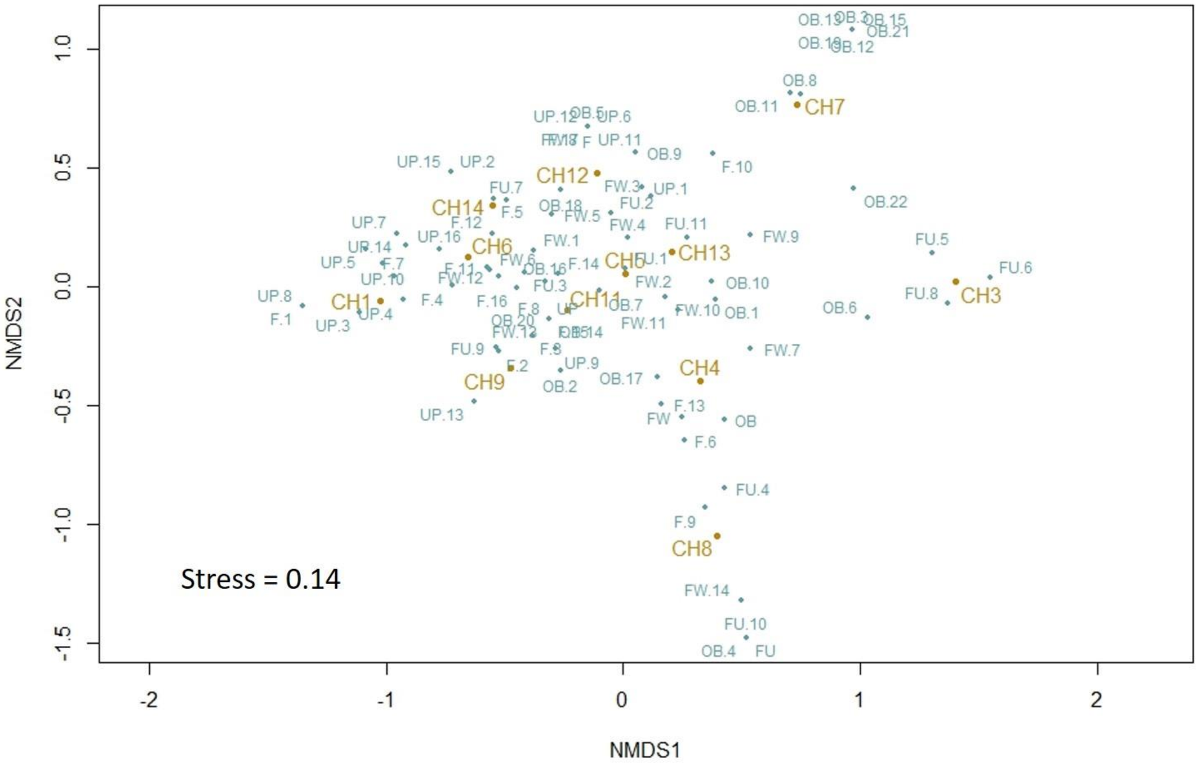


***Figure 2.9**. NMDS Ordination Plot of Species Composition*

Layering in the environmental factors, the constrained CCA ordination showed a correlation along the first axis between plant communities dominated by facultative, facultative upland, and upland species and increasing values of BBW, D50, GGW, and WDR (Figure 2.10). The site most aligned with these species is the CH1, “C_F” channel, followed by the CH8, “C” channel. Obligate wetland species responded favorably to high values of ER, ASS, FPW, and WR. Sites most associated with high values are the CH3, “E” channel at the meadow site and the “D_A” channel at CH7. The variables aligned with the second axis are MD and BXA. Species run the gamut of wetland to more upland preferring species. Many of the analogue "C"/ "C_E" channels have a similar mix of species and their plant communities appear to not be strongly driven by any particular set of environmental factors. A Mantel test comparing the environmental and species matrices produced a test statistic of 0.5 (significance = 0.02), indicating a 50 percent correlation between species composition and environmental variables.

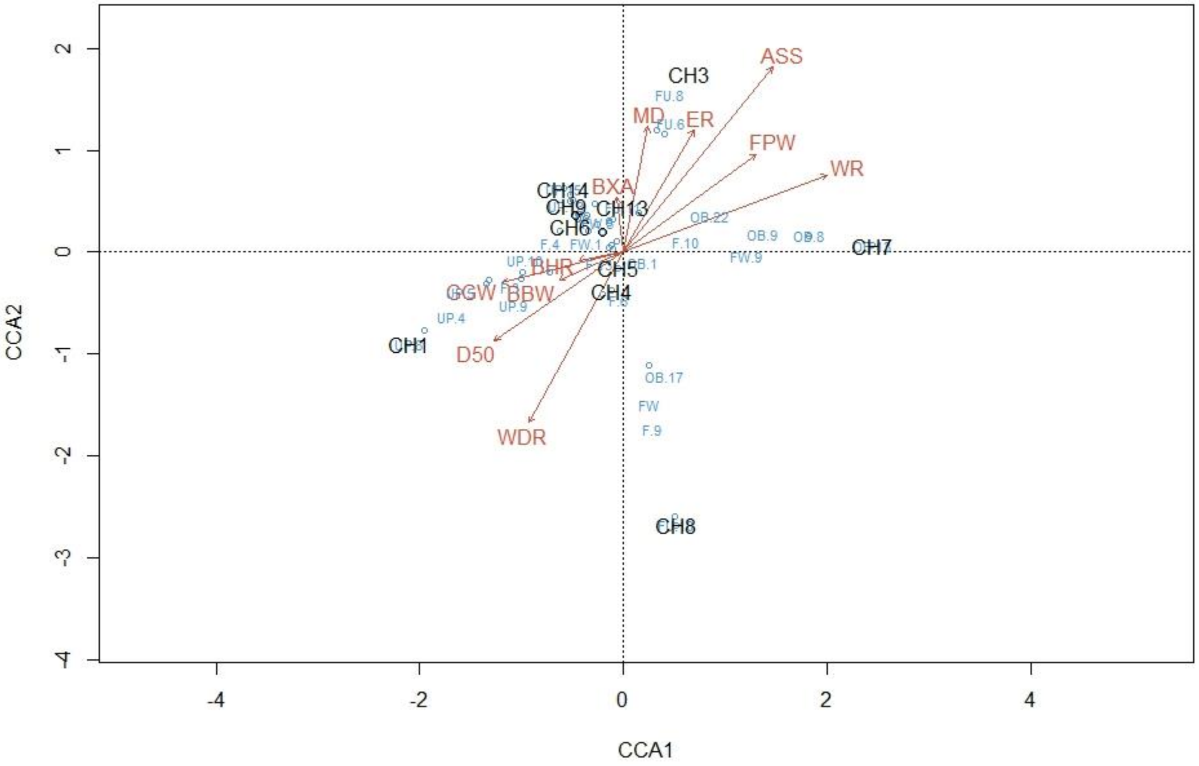


Figure 2.10. *CCA Ordination Plot of Species by Environmental Factors*

State and Phase Determination

Analysis of environmental data showed the strongest differences between sites to be occurring between the "C_E" and "E" channels located at CH2 and CH3, and all other sampled reaches. This is based upon the height of the "branch" associated with these to sites (Figures 2.5 & 2.6). In ordination space, these two channel types also showed an obvious separation (Figure 2.7). A strong positive correlation was also exhibited for two environmental factors, ER and FPW (Table 2.8). Incorporating the three biotic factors derived from the greenline data, and additional positive correlation was noted for the CH3, “E” channel for WR and ASS as well as a negative correlation for GGW (Table 2.7). Analysis of species composition data clustered together the CH3, “E” channel with the CH7, “D_A” channel (Figure 2.6). In the NMDS ordination, it can be seen that species at both reaches mostly have obligate wetland ratings (Figure 2.9). With this information, and factoring in the fact that two meadow sites are not in the incised condition of the other reaches, the CH3, “E” channel was determined to represent a functioning state, existing prior to channel incision and characterized by a high ET, low WDR, low GGW, wide, connected floodplain and prevalence of stabilizing, wetland vegetation. This is essentially the reference state for the model, but as a return to pre-settlement conditions will not realistically ever be achieved, it is instead referred to in this model as: State 1 – Potential Natural Channel, indicating the highest potential of riparian conditions within the system. With a lower ER/FPW and higher WDR, The CH2, “EC” channel is viewed as marking a transition between an “E” channel phase and “C” channel phase within the state.

The analysis of environmental data also separated out the CH1, “C_F” channel. With four clusters, its dendrogram branch is at a lower height than “CH2/CH3” branch for the 14-site, channel metric cluster analysis, the 12-site channel/biotic factor analysis and species composition analysis (Figures 2.5 & 2.6). Ordination revealed an array of primarily UPL and FACU species at this location (Figure 2.9) and positive correlations with WDR and D50 (Figures 2.7 & 2.8). However, this reach shares the characteristic of all other sampled reaches, minus the two meadow sites, of being within the old incised channel banks. Because the incision banks play such a strong role in determining floodplain and channel development, and effectively curtail the extent of the riparian corridor, it made sense use this aspect as a defining feature of a second state, and sort the channel types into phases. With this approach, the CH1, “C_F” channel marks an early evolutional phase where channel downcutting and widening have stabilized enough to allow for the formation of a rudimentary floodplain. In the model, this is referred to as, State 3 – Stabilizing/Stable Analogue Channel. The remaining incised sites are grouped much closer together when viewing the environmental cluster dendrograms, due to greater similarities between metrics, and represent different phases of channel evolution (Figures 2.5 & 2.6). This pattern is also observed on the PCA ordination diagram, with the exception of the “C” channel found within a water gap (CH8) and one “C_E” site (CH11). These both show separation along the second axis, which maximizes explained variance for BXA and MD. The “C_E” channel is high on the diagram, indicating a positive relationship with these variables, while the CH8, “C” channel, with its small cross-sectional area and shallow depth, aligns with the bottom of the diagram, indicating a negative relationship. This latter location is similar to CH1 in that, as a water gap, cattle have access to it during the summer months. However, it has a higher ER and lower WDR suggesting better floodplain connectivity. When looking at the WR value it is quite high indicating a predominance of wetland obligate and facultative wetland species. Yet, the ASS score plummets dramatically implying that stability is low. Though the reason is unclear, the BXA and MD values at the CH11, “C_E” channel are substantially higher than at any other location, hence causing the separation seen in the PCA diagram (Table 2.1).

To a lesser degree, this also true for BXA at the CH5, “C” and CH7, “D_A” channels, which seems to be why these three reaches were placed in the same clusters associated with the environmental factor analysis (Figures 2.5 & 2.6). It is difficult to separate out phases based on the environmental data alone, but some patterns are apparent. For example, “C” channels tend to have high WDR and GGW, while the “C_E” channels are more sinuous and are associated with high WR and ASS scores (Table 2.8). With species data, wetland ratings are variable, but communities along “C_E” channels have a greater prevalence of willows and higher seral graminoids. When considering all of the analyzed data there appears to be an overarching pattern of channel evolution that starts from the early, “C_F” phase with a very high WDR and GGW, coupled with a low ER, FPW, ASS and WR (Table 2.8). The next phase is marked by the formation of a “C” channel which still maintains high WDR and GGW values, but has a lower ER and vegetation metrics are higher in connection to a wider and more developed floodplain (Table 2.8). This trend continues into a “C_E” channel phase marked by higher sinuosity and further development of floodplain width and plant communities (Table 2.8). All of these phases would be considered as “stabilizing” phases. A “stable” phase is reached with the formation of an “E” channel marked by a high ER and low BHR (Table 2.8). However, anomalies exist within the data for each phase, such the high BXA/MD at CH11, as well as high BHR values associated with most of the “C_E” channels. It would be expected to see lower values in these reaches as a high BHR typically indicates low stability. An alternate phase occurs with beaver colonization, which creates a “D_A” channel marked by a high FPW, ASS, and WR (Table 2.8).

The process of disconnection from the original, State 1 floodplain and subsequent incision and channel widening is the domain of a third state, State 2 – “Degraded Channel”. A transition to State 3 then occurs when these processes have abated enough to allow for the development of a new floodplain within the incised channel banks. Regarding State 2, as discussed, direct, representative data was not adequately collected. However, anecdotal evidence advocates for its existence. From historical data, it is known that the wide, low-gradient portions of Maggie Creek once consisted of a complex of wet meadows, mesic meadows, and where beaver were established, sprawling, multi-thread channels (Horton 2000, NDWP 2000). The combination of human-induced and natural disturbance led to substantial vertical erosion and narrowing of the active floodplain, which is now contained almost exclusively within incised banks. These banks are often over 3 meters tall and flank riparian corridors spanning from 30 to 100+ meters. Although rapid change no doubt followed some of the large floods that occurred over the past 120 years, it is evident that current conditions represent a more enduring cycle of channel downcutting and widening. This cycle is best witnessed in contemporary times by travelling once more to the beaver-irrigated meadow. The reach at CH2 classified as an “E” channel going towards a “C” channel demonstrates a noticeable reduction in the floodprone area. At CH3, the floodprone area is 200 meters wide, but shrinks down to 140 meters immediately downstream at CH2. Even more dramatically, the ER is reduced by just over 50 percent (CH3 = 109, CH2 = 50). Below the CH2 reach, the channel becomes even deeper and the channel banks are beginning to slough off. The dominance of Nebraska sedge also lessens as the plant community becomes saturated with species better suited for drier soil conditions, including non-native weed species such as

Onopordum acanthium (Scotch thistle) and *Bromus tectorum* (cheatgrass). The bank sloughing is more evident further downstream and the channel becomes wider suggesting the transition to an "F" channel. Continued vertical and lateral erosion in this area was observed during a prolonged period of large-scale flooding that occurred during the spring of 2023. In the last ~900 meters of the channel, before it joins back with the older, stable analogue C / E channel, conditions are representative of the majority of the study area with sufficient widening of the channel banks to form a new, reduced floodplain and the entrenchment ratio, at least observationally, appears to be in keeping with a classic "C" channel.

Plant Community Components

Within the study area, PCCs were stratified along a depth-to-groundwater gradient. Four unique components were consistently observed and occupy the following fluvial surfaces: "Edge of Stream", "Stream Bank", "Floodplain" and "Terrace". The floodplain community can include overflow channels and the terrace community can be further subdivided into top of terrace and terrace slope. As some reaches do not contain overflow channels and the compositional differences are often minor between the top and slope of the terrace, it was decided to combine these elements when outlining the PCCs for the actual model.

The PCCs were numbered to indicate their proximity to water (PCC1 = Edge of stream, PCC2 = Stream Bank, PCC3 = Floodplain, and PCC4 = Terrace) (Figures 2.11 & 2.12). The dynamic nature of riparian systems makes it difficult to fit these communities with a blanket template, but the following composition breakdowns provide at least an approximation of expected species and mean percent frequency and are similarly associated with each state in the model. Table 2.9 relates each PCC to its corresponding fluvial surface and composition.

Figure 2.11. *Representative Plant Community Components (PCC) in State 1*

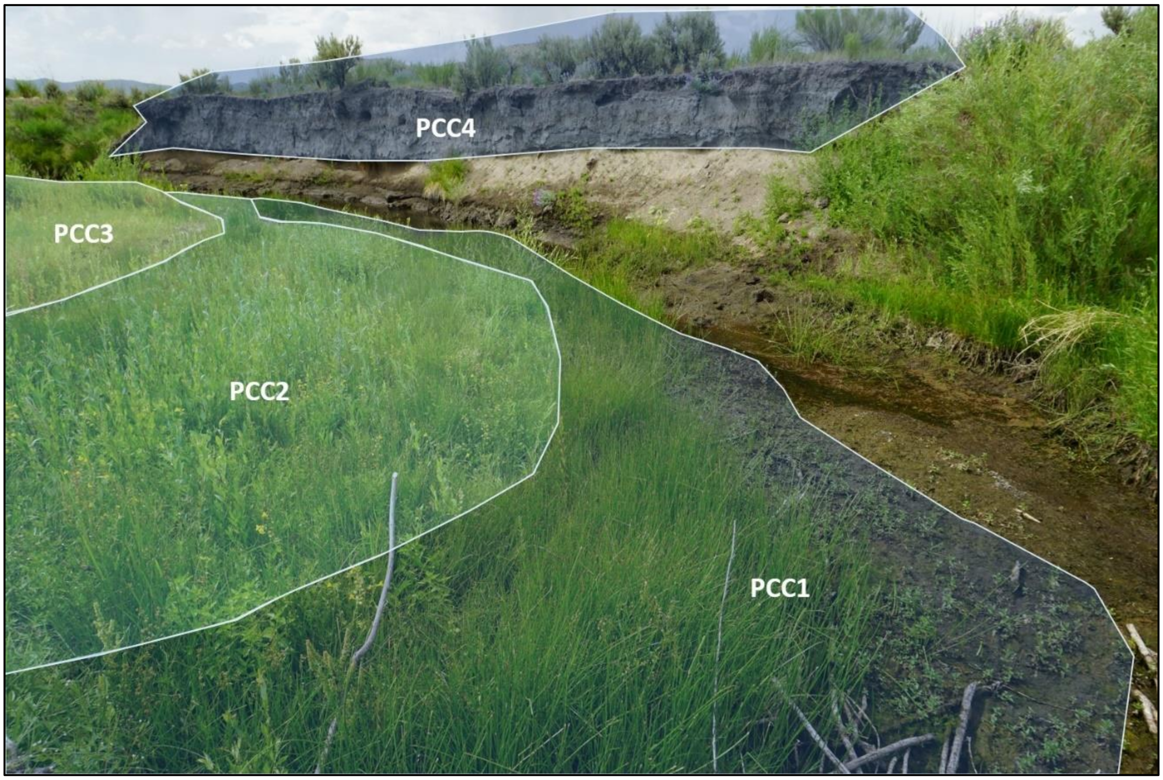


Figure 2.11. *Representative Plant Community Components (PCC) in State 3*

Table 2.9. *PCC Location, Wetland Rating Composition and Stability Rating by State*

State/PCC	**Fluvial Surface**	**% Composition by Wetland Rating**	**Winward Stability Rating**
State 1 – PCC 1	Edge of Stream	OBL (100%)	8.5-10
State 1 – PCC 2	Stream Bank	OBL (85%), FACW (10%), FAC (5%)	5-8.5
State 1 – PCC 3	Flood Plain	OBL (80%), FACW (10%), FAC (10%)	5-8.5
State 2 – PCC 1	Edge of Stream	Absent	N/A
State 2 – PCC 2	Stream Bank	FAC (80%), FACU (20%)	2
State 2 – PCC 3	Flood Plain	FACW (20%), FAC (70%), FACU (10%)	2-5
State 3 – PCC 1	Edge of Stream	OBL = 100%	5-8.5
State 3 – PCC 2	Stream Bank	OBL (25%), FACW (55%), FAC (15%), FACU (5%)	2-5
State 3 – PCC 3	Flood Plain	OBL (15%), FACW (55%), FAC (15%), FACU (5%)	5
State 3 – PCC 4	Terrace	FAC (5%), FACU (25%), UPL (70%)	2

PCC1

Of all the PCCs, this one is both ubiquitous and fairly consistent throughout the study area. At more degraded locations, such as at CH1, there is a higher percentage of bare ground and weedy, non-native species. In the more developed CE reaches, there is a tendency for early seral obligate wetland species to be replaced by mid-to-late seral obligate wetland species. General plant community: *Eleocharis palustris* (common Spikerush) (33%), *Schoenoplectus americanus* (chairmaker's bulrush) (15%), *Argentina anserina* (silverweed cinquefoil) (14%), *Agrostis stolonifera* (creeping bentgrass) (13%), *Veronica americana* (American speedwell) (8%).

PCC2

The stream bank PCC has more variation than PCC1, though stays somewhat uniform based upon channel type. Again, the "CF" channel at C1 is more prone to bare ground and weedy, non-native species, particularly *Bromus tectorum* (cheatgrass) and *Onopordum acanthium* (Scotch thistle), both of which are rated as upland species. Within the water gap at CH8, woody species are lacking and though the community is composed mostly of obligate and facultative wetland species, they tend to be shallow-rooted, early seral species. However, the other "C" type channels have a mix of woody and often, mid-to-late seral graminoids. In these areas, the woody component consists largely of willows. The "C_E" channels show an increase in woody species which include willows, *Rosa woodsii* (Woods' rose), and *Ribes aureum* (golden currant). The analogue "E" channel is similar, though diversity is higher within the herbaceous component. The community associated with the true "E" channel at the meadow site is made up almost exclusively of *Carex nebrascensis* (Nebraska sedge). General plant community: *Carex pellita* (woolly sedge) (32%), *Salix exigua* (coyote willow) (17%), Nebraska sedge (12%), *Symphyotrichum spathulatum* (western mountain aster) (12%), *Salix lutea* (yellow willow) (9%), *Equisetum hyemale* (scouringrush horsetail).

PCC3

This community component is the most varied in the study area is shaped by factors such as width of floodprone area and beaver activity. In newer beaver wetland complexes with a high percentage of standing water, such as at CH7 and CH10, *Typha latifolia* (broadleaf cattail) is a dominant floodplain species, and often found growing with *Schoenoplectus* spp. (bulrush) and wetland obligate forbs (e.g. *Bidens cernua* [nodding beggartick], *Mimulus guttatus* [seep monkeyflower]). Where they have been established for longer periods of time, or where they have been abandoned, successional processes lead to willow-dominated communities with pockets of sedge meadows at higher positions or where ponded water has since been drained. Narrow floodplains with “C_E” channels are characterized by dense stands of mature willows and rose thickets. Adjacent to the “CF” and “C” channels, floodplains typically have sparse to extensive stands of coyote willow. Where overflow channels are present, they often intersect groundwater and support fringe communities similar to PCC1. However, by virtue of not being regularly exposed to the buffeting forces of flowing water, ruderal, disturbance-adapted species are less abundant. These often mark the occurrence of yellow willow as well, especially in the floodplains of the less-stable “C” channels. General plant community: coyote willow (50%), *Leymus triticoides* (creeping wildrye) (30%), yellow willow (15%), *Carex praegracilis* (clustered field sedge) (20%), woolly sedge (15%), mountain rush (10%), and *Poa pratensis* (Kentucky bluegrass) (5%).

PCC4

With the exception of the Nebraska sedge meadow sites, this community is represented by an upland shrub community dominated by basin big sagebrush with rubber rabbitbrush occurring in at least a subdominant position, but often codominant. The understory can be depauperate, but typically maintains a mix of graminoids, like *Elymus elymoides* (bottlebrush squirreltail), creeping wildrye, basin wildrye, and clustered field sedge; along with forbs, especially *Senecio integerrimus* (lambstongue ragwort) and *Iva axillaris* (povertyweed). When burned, the shrubland typically converts to a basin wildrye/creeping wildrye grassland. General plant community: basin big sagebrush (60%), rubber rabbitbrush (40%), Creeping wildrye (10%), clustered field sedge (5%), povertyweed (5%), lambstongue ragwort (3%).

Bird Diversity

A total of 69 bird species were observed utilizing the Maggie Creek riparian area and surrounding uplands (Appendix A, Table A.2). A majority of species (43) belonged to Order *Passeriformes*, followed by *Accipitriformes* (5 species), *Anseriformes*, *Charadriiformes* and *Pelecaniformes* (4 species each). Nine other orders were represented, each with a single species. Diversity, richness and evenness values for each location (riparian and upland) are listed in Table 2.10.

Table 2.10. *Bird Diversity – Riparian vs. Upland*

Site/Channel Type	H - Riparian	H - Upland	Richness - Riparian	Richness – Upland	Evenness - Riparian	Evenness - Upland	Hutcheson's t-test (riparian ~ upland) (p value)
$1/C_F$	0.87	1.32	8	9	0.42	0.6	0.012
2/C	1.79	0.96	8	4	0.86	0.69	$2.6e^{-3}$
$3/D_A$	2.09	1.77	12	10	0.84	0.77	0.041
4/E	1.55	1.71	8	7	0.75	0.88	0.45
$5/C_E$	2.49	1.82	18	10	0.86	0.79	$3.8e^{-4}$
$6/C_E$	2.15	1.05	14	5	0.82	0.65	$1.6e^{-12}$
$7/C_E$	2.30	1.9	19	9	0.78	0.86	0.0052

With alpha set at 0.05, all sites proved to have significantly different bird assemblages between their respective riparian and upland locations, with the exception of Site 4. Diversity was lowest at Site 1 (0.87) followed by Site 4 (1.55) and Site 2 (1.79). All other sites had values above 2.0, with Site 5 having the highest (2.49). An analysis of similarity between riparian and upland areas at each sampling site, using Bray-Curtis distance, revealed Site 1 to have the shortest distance (0.31) and the remaining sites are fairly similar and have much longer distances (Site 2 = 0.96, Site 3 = 0.98, Site 4 = 0.92, Site 5 = 0.82, Site 6 = 0.85, Site 7 = 0.79). The results of pairwise comparisons of each riparian site are presented in Table 2.11. All sites were significantly different than Site 1, which is located within the State 3, "C_F" reach. Significant differences were not detected between several of the other sites, such as a between Site 2 and Site 3, Site 2 and Site 4, and Site 3 and Site 6. An Indicator Species Analysis did not reveal any significant indicators of a given location or habitat type.

Table 2.11. *Pairwise Comparison of Riparian Sites – P Values (Hutcheson's T-Test)*[1]

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Site 2	**$2.4e^{-5}$**					
Site 3	**$5.3e^{-11}$**	0.06				
Site 4	**0.003**	0.26	**0.003**			
Site 5	**$2.43e^{-16}$**	**$8.7e^{-5}$**	**$2.7e^{-4}$**	**$2.23e^{-6}$**		
Site 6	**$1.97e^{-13}$**	**0.05**	0.64	**0.027**	**0.013**	
Site 7	**$1.4e^{-13}$**	**0.003**	0.06	**$1.0e^{-4}$**	0.103	0.28

[1] Bolded values are significant at alpha = 0.05

Stream Temperature

As noted, 10 data loggers were deployed and their tenure on the creek was fraught with issues. Loss of some loggers during flood events, and periods where loggers became stranded above water as flooding receded, necessitating the removal of portions of the timeline for all loggers.

For paired, upstream/downstream loggers, the most complete data sets were from 2022 and obtained from those located at the hay meadow section and the two bracketing Coyote Creek. For the hay meadows, this included the period from May 8, 2022 to August 8, 2022, minus 11 days in June when both loggers were above water (Figure 10).

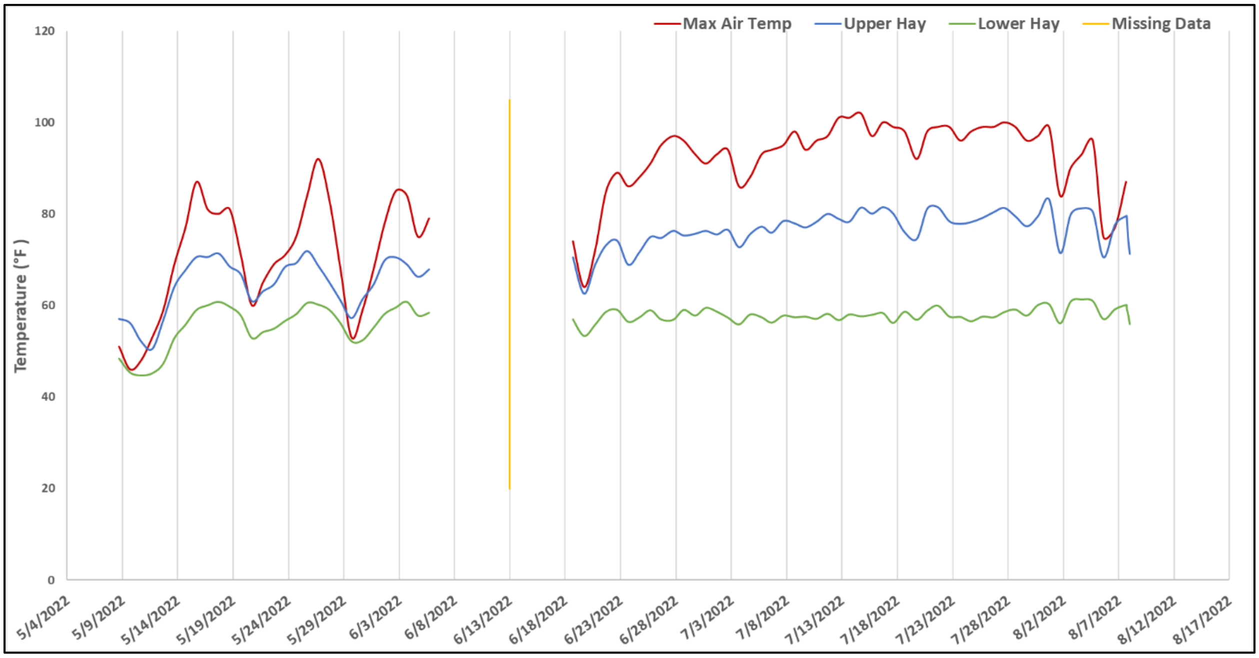


Figure 2.11. *Maximum Daily Stream and Air Temperatures at the Hay Meadows*

At the hay meadow location, irrigation began on April 15 and was shut off on July 15. During this time, flows averaged 20 CFS (USGS 2024). The corresponding graph showed water temperatures at the downstream location to be less than at the upstream site from May through early June, but temperatures fluctuated according to diurnal changes in ambient air temperature. From late June through mid-August, the average temperature at the downstream logger increased by only two degrees Fahrenheit (55.3° to 57.2°) and overall temperatures were largely independent of changes in air temperature. Conversely, the average temperature at the upstream logger increased 10 degrees Fahrenheit (64.5° to 74.6°) and daily readings tracked with increases and decreases in air temperature. The Wilcoxon test results showed a significant difference in means for each month (May: $p = 1.23e^{-05}$, June: $p = 3.19e^{-07}$, July: $p = 1.39e^{-11}$, August: $p = 1.40e^{-10}$). A significant difference was also detected when comparing the combination of all months ($2.2e^{-16}$).

Intact data from the Coyote Creek loggers spanned the period from mid-June to the end of July. A similar pattern to the hay meadows was exhibited in that temperatures were similar at both sites early in the season, then a divergence occurred where the average maximum temperature at the upper location went from 74°F for June, to 81.7°F for July (Figure 2.12).

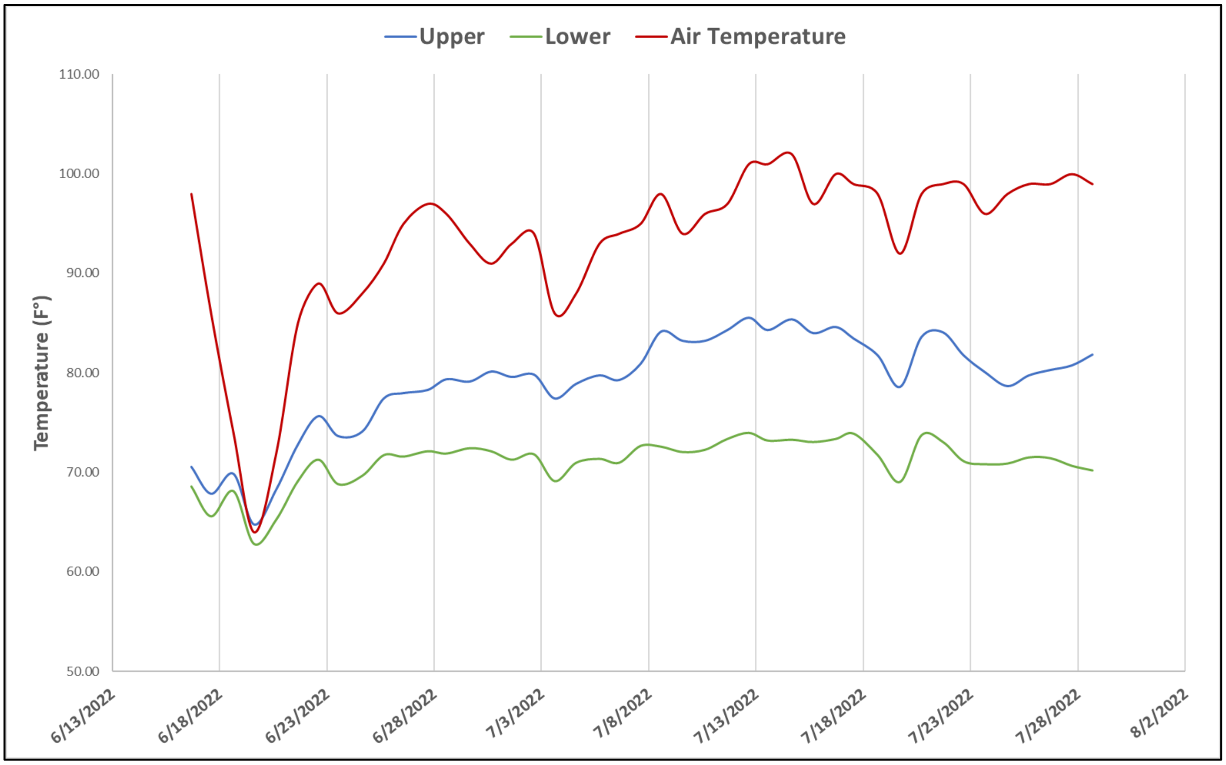


Figure 2.12. *Maximum Daily Stream and Air Temperatures Above and Below the Confluence of Coyote Creek*

Averages at the lower site were 69.4°F in June and 71.9°F in July. Wilcoxon tests were performed for June (p = 0.011), July (p = $1.4e^{-10}$), and the combination of both months (p = $3.59e^{-10}$). However, the base low temperatures at the lower site were warmer than the corresponding location at the hay meadows. The graph also showed that regardless of location, stream temperatures consistently mirrored changes in ambient air temperature throughout the duration of the analyzed time frame.

Salvageable data from the unpaired loggers showed a general trend of Maggie Creek water temperature. In the early portion of the growing season (late April to Mid-may), temperatures range from 40-50°F and gradually increase to 70-80°F during the hottest months. There is also a pattern of steeper increases during the mid-morning to early afternoon period as the stream gains solar energy. Evaporative losses slow the rate of increase during the later portions of the afternoon and early evening. A similar pattern has been documented along a stream in a semi-arid region of eastern Oregon (Benner & Beschta 2000). An overlay graph of all available data shows temperatures from each location are very similar from May through the end of June, with the exception of the lower hay meadows where stream temperature stays about 10°F below the other sites. This trend continues throughout July and August, though temperatures below the mouth of Coyote Creek run 7-10°F cooler than at other sites, barring the lower hay meadows area.

Discussion

Model Development – States and Phases

A variety of factors were considered and amalgamated in the pursuit of constructing the STM. From historic narratives, aerial imagery, and observation, it is evident that much of the study area would indeed have had an extensive riparian zone associated with a complex of “C”, “E”, and “D_A” channel-types. Just as evident is that much of the riparia within the study area has since become reduced in capacity due to channel alterations, namely channel straightening to accommodate past haying operations; a history of year-long grazing up into the early 1990s; and periodic large-scale floods. The combined effect of these disturbances has been accelerated vertical erosion that greatly lowered the channel elevation, severing it from its former floodplain. Subsequent channel widening and passive restoration has allowed for the development of a new, narrower floodplain at various levels of recovery. This process is represented in the model with three states: State 1 – Potential Natural Channel, State 2 – Degraded Channel and State 3 – Stabilizing/Stable Analogue Channel.

Analysis of physical variables and species composition demonstrated a clear separation of the two, State 1, “E” channel meadow sites from the other sites. The main drivers of this separation are FPW and ER which are positively correlated indicating that an increase in one will lead to an increase in the other. There is also the implication that a wide floodplain (i.e. wide FPW) that is well connected to the channel (i.e. high ER) will reduce the depth to groundwater thus promoting plant communities adapted to wetter soil conditions. The high WR associated with CH3 is in keeping with this line of reasoning and the high ASS score indicates that the plant community is composed primarily of stabilizing species. The active floodplain is on the same feature that represents an abandoned floodplain terrace in the incised State 3 condition. The PCC here contains Nebraska sedge as the overwhelmingly dominant species with minor occurrences of mountain rush, *Deschampsia cespitosa* (tufted hairgrass), other sedges, and various, mesic forbs. Nebraska sedge occurs in other PCCs found within State 3, though never reaches the levels seen within the meadow at CH2 and CH3. As an obligate wetland species with aerenchymous root tissue, this is certainly attributable to wetter soil conditions found throughout the floodprone area (Martin & Chambers 2001). It also demonstrates a preference for fine-textured soils, which underly the meadow, and is strongly rhizomatous giving it a competitive edge (Crowe & Clausnitzer 1997; Quistberg 2007). There is also an element of land use that may be partially responsible for the abundance of this sedge and that is a history of grazing. This species responds favorably to grazing and can withstand heavier utilization than many species that might otherwise be co-dominant (Crowe & Clausnitzer 1997). Grazing has been infrequent and light since implementation of the MCWRP in the early 1990’s, but use was much higher in the decades proceeding the MCWRP. The stream bank PCC is essentially the same as the floodplain and PCC1 is typically discontinuous and represented by only a narrow margin between water’s edge and the top of the bank. This is not surprising for this channel type. The average distance between the top of the bank and top of water is short, banks are often under cut, and the depositional features common to less-stable reaches are largely absent; thus, space and rooting substrate for PCC1 is much reduced. Quantitative data was not collected for “C” or “D_A” phases in this state, but observationally, differences were noted. The beaver dam wetland complex upstream of CH3 has numerous small channels and a broader PCC1 as the increased extent of ponded water provides much greater substrate for the development of species associated with this component. At CH2, the “E_C” channel is transitioning into a “C” channel with a wider WDR, lower ER and slight increase in the width and continuity of PCC1 as overhanging, concave banks are replaced by linear banks.

With sufficient pressure, State 1 can exceed a threshold that pushes the system into a series of channel degradations characterized by vertical and lateral erosion. Once a degree of stabilization has been reached, the systems transitions into State 3. The “C_F” channel at CH1 marks the initial phase where floodplain development begins to occur within the incised banks. At this point, channels are aligned with high WDR and GGW values, while ER and FPW are much lower than those associated with State 1. Although a low ER is somewhat counter-intuitive by name, this value is derived by dividing the floodprone width by the horizontal distance between bankfull locations. Thus, low numbers represent conditions where the FPW is narrow and channel width is wide, which is typical of incised channels that have had time to start eroding laterally. The WDR (bankfull width divided by average depth at bankfull), is a measure of energy distribution. High values indicate wide, shallow channels that push energy against the banks, increasing the amount of bank erosion. As a channel widens, so too does the distance between banks which increases the distance between green lines and bankfull locations. In these settings, flood plain connectivity is severed, or at least much reduced, and plant communities often reflect drier conditions and rooting structures that are not well suited for withstanding erosive forces. This is borne out by WR and ASS values which at CH1, are much lower as compared to any other site.

The wide, shallow channel and high terrace banks suggests that the reach has long since progressed from a State 1 condition and is maintaining or developing a low degree of channel stability within the incised banks. This site is unique from the rest of the study area in two regards; 1) grazing use and 2) water regime. As mentioned previously, this is the only area, aside from water gaps, where at least some summer grazing occurs and grazing management has remained largely unchanged for decades. It also has fairly consistent stream flow. Groundwater intercepted at an open pit gold mine on the west side of the valley is pumped into a reservoir located just upstream of this reach, on the east side of the valley. Reservoir water is conveyed to a series of cooling towers where it is then released into Maggie Creek. This creates mostly consistent flow throughout the year which supports floodplain plant communities, but also greatly dampens flood-related channel degradation/aggregation activity, effectively keeping conditions at a status quo. The annual movement of cattle through this area towards the end of the summer also affects vegetation composition which is marked by a general lack of stabilizing species, especially willows, which are particularly enticing to cattle late in the growing season when more desirable forage is dried up or lacking (Kovalchik & Elmore 1992; Batchelor et al 2015). These factors make CH1 a convenient anchor in which to compare the range of State 3 channel evolution present in the study area.

A good example of this occurs at the water gap reach at the CH8, "C" channel site. Summer grazing occurs at both CH1 and CH8, but at the latter site, cattle are managed so that they do not stack up and linger within the water gap for long periods of time. Keeping in mind that the entirety of the project area was in a conditions similar to that now seen at CH1, this lighter disturbance regime, and channel data, CH8 suggests a movement towards a narrower channel, better connected floodplain, and development of, even if early seral, a more substantial riparian plant community. With the greenline community data, it can be seen that the dominant components (e.g. *Agrostis stolonifera*, *Trifolium wormskioldii* and *Eleocharis palustris*) are mostly early seral species typical of areas undergoing regular disturbance. Vegetation trends are also reflected in the larger pattern of channel evolution. Unlike the State 1 meadow, reaches within the incised channel banks have a mostly gravelly substrate which is typical of alluvial valleys which receive abundant material from side channels and experience fairly consistent spring floods. These flooding episodes deposit finer material on top of the floodplain and behind structures (e.g. beaver dams, point bars) and/or transport them out of the system. Gravel is therefore a common component of communities found next to the wetted channel, stream banks, and portions of the active floodplain, which plays a role in species assemblage. This is evident with the proliferation of coyote willow and, to a lesser extent, yellow willow, two species that are adapted to growing in coarse substrates (Hansen et al. 2008). However, they are not limited to gravels and are often found growing initially on finer material which is an important factor when characterizing riparian development along Maggie Creek. At stabilizing "C" channels, such as CH4, CH5, and CH6, coyote willow is the dominant woody species and colonization occurs first along the tops of depositional bars. These features are stratified with sand and small gravel at the top, grading in depth to larger gravel/cobble material. As these colonies mature and fill in, recruitment begins to extend down gradient, occupying the coarser substrates between the bar tops and channel edge. In reaches with broader floodplains containing overflow channels, this pattern often repeats along bars and high points deeper into the floodplain and thus, coyote willow is a common component of PCC2 and PCC3 at stabilizing sites where channel movement and erosion/deposition events are common. Yellow willow demonstrates a similar progression of establishing on finer-textured soils and expanding onto to coarser substrates, particularly along channel margins. One difference is that yellow willow is a wetland obligate, thus even in the floodplain areas dominated by coyote willow, it will often become established in depressions and along overflow channels that intercept the water table. It also persists in flooded conditions behind beaver dams, whereas prolonged inundation will initiate mortality in populations of coyote willow. Within the study area, yellow willow is typically a later-seral species than coyote willow. It becomes more abundant in the PCC2 and PCC3 communities as the channel becomes more stable, less prone to frequent shifts and has increased floodplain connectivity.

An interesting willow-related phenomenon has occurred at the northern end of the project and is captured in the data collected at the “C_E” channels at CH19 and CH20. Here yellow willow, sometimes mixed with coyote willow, is a dominant species in PCC2 and PCC3 where it forms nearly impenetrable stands of mature individuals (Photograph B2, Appendix B). Adding to stand density are intermingling thickets of Woods’ rose and golden currant. This community type reflects a past history of sustained or substantial, short-term disturbance. The willows would have established along the bank and point bars at a time of greater floodplain connectivity and higher soil moisture. Channel incision has led to drier conditions causing a shift in the understory towards less wetland-orientated forb and grass species. These were eventually replaced by rose and currant, which tend to increase under prolonged and/or heavy livestock use (Manning & Padgett 1995). Willow persistence is attributed to a sufficient size and maturity to allow roots to still access the lowered water table and the change in grazing patterns following the MCWRP. Looking at the channel characteristics, it is evident that these reaches, though stable, suffer from very low ETs, very narrow FPWs, and high BHRs. In essence, the prevalence of deep-rooted woody species has “pickled” the channel, thus keeping it in its current narrow, incised condition. On the plus side, the plant communities offer excellent breeding habitat for many migratory bird species and good overall habitat for many other types of wildlife.

The herbaceous community in State 3 reaches can also be indicative of a specific phase and/or disturbance regime, especially when considering the compositional array of PCC2. Much as with the prevalence of coyote willow along reaches subject to frequent change, the streambank herbaceous community also reflects a system in flux. Frequent alteration of gravel point bars causes a "clock resetting" scenario expressed as a mosaic of unvegetated surfaces and patches of ruderal colonizers, such as creeping bentgrass, *Muhlenbergia asperifolia* (scratchgrass), silverweed cinquefoil, yellow sweetclover and *Trifolium* species (true clovers). These early seral species are also adapted to growing in various substrates, including gravel and cobble. Compositionally, the PCC2 within the CH8 water gap is similar to its counterpart at CH1, though appears more stable despite somewhat regular cattle use during the summer. This is probably due to it being directly downstream of an active beaver wetland complex which almost certainly dissipates flood energy. There are also numerous willow stumps scattered about with old tooth marks suggesting that the current upstream colony likely migrated from this location. Fine sediments captured behind dams would then have become exposed as dam maintenance ceased and the dams eventually failed. These types of sediments are suitable for the establishment of mid-to-late seral sedge species. Indeed, Nebraska sedge was observed occupying wetter, depressional areas within both PCC2 and PCC3.

However, it is another sedge species, woolly sedge, that serves as the most prominent symbol of "C" channel evolution. Like Nebraska sedge, it prefers finer-textured soils and is thus absent from the PCC2/PCC3 communities until sufficient deposition has occurred behind previously established vegetation, rocks, beaver dams, etc. It is also an obligate wetland plant, but on average, can be found at locations with a higher depth to the water table (Manning & Padgett 1995). It also does well on slumps that have caved off from eroding, vertical sections of bank. By falling into the creek, these slumps essentially provide well-watered blocks of fine material. The resulting widening of the channel helps dampen the velocity of subsequent floods, such that material tends to stay in place and eventually becomes stabilized by further expansions of woolly sedge dominated communities. Without undue disturbance, these communities fill in behind and adjacent to the slump blocks, forming a near continuous line along the bank toeslope (Photographs B3 & B4). As the original blocks degrade and additional sediments are trapped, stable benches form increasing the width of the floodplain. This development of woolly sedge communities tracks well with the various stages of “C” channels in the study area. The “C_F” and initial stabilization of “C” channels can be broken out by the lack of this graminoid. Slump block colonization marks the transition to more stable forms with the creation of woolly sedge benches marking the shift into a late phase, “C_E” channel form. From this point, the primary separation of analogue “E” channels from “C” channels is further narrowing and deepening of the channel and increased sinuosity. Plant community components remain similar, however there tends to be greater diversity in willow age classes and banks/floodplains start picking up additional obligate wetland graminoids such as *Glyceria grandis* (American mannagrass), *Beckmannia syzigachne* (American sloughgrass), *Scirpus microcarpus* (panicled bulrush) and *Sparganium ermersum* (unbranched bur-reed). The width of PCC2 and PCC3 also tends to increase.

Where beavers have established wetland complexes, a further dimension of nuance occurs. As mentioned at CH7, newer colonies have the trademark D_A attributes, but wetland vegetation is largely herbaceous. In beaver-created wetlands of longer tenure, a successional trend towards willow species, is evident. Where dams have failed or beavers have moved, there tends to be an initial wave of early seral, weedy species occupying areas formerly under water. The species assemblage at the CH9, “E” channel site, which range from obligate to upland, is a good representation of this process. This site, prior to this study, consisted of numerous small beaver dams with one very large one at the lower end of the wetland complex. The large dam failed in the spring of 2019 and the area behind the dam has been undergoing plant successional processes over the last few years. An example of this process further along the timeline occurs where beavers shifted Maggie Creek onto the meadow at CH2 and CH3. Adjacent areas that were previously inundated have now converted to stands of willow punctuated with open, *Carex* meadows.

Though vegetation assemblages offer a quick assessment of beaver occupation, and duration of occupation, clues are also offered from the environmental data. For instance, the two sampled locations with current beaver activity (CH7 & CH10) each had wide FPWs. The ERs are also higher than at other sites within the incised channel banks. An interesting subtlety comes to light when considering these two metrics for sites where beavers were known to be up to ten years ago, but have since moved to other locations. Beavers were active at CH9 up until 2019 and recorded measurements for FPW and ER were 60m and 9, respectively. At CH5 the metrics came in at 142m and 9. Beaver dams at this location are visible in aerial imagery up until 2014. Though ERs are certainly lower than at CH7 and CH10, they are higher than those recorded for any other reach within the old incision banks. Sign-postings of a legacy of beavers also emerge from the proliferation of deep pools, small floodplain channels, dam remnants, and numerous other features that combine to add layers of complexity and character to the system. These residual effects of beaver occupation entwine their way through the recent history of the study area and are often too fine to adequately capture within the coarse framework of a model, but are significant factors in the upward trending trajectory and current manifestation of Maggie Creek riparian dynamics.

The old terrace feature is exclusive to State 3 and marks the location of PCC4. This is a predominantly upland community component and maintains a degree of consistency throughout the study area. There are, however, a few notable differences that warrant a measure of reflection. The first has been touched upon already and concerns the incident of fire. For the most part, burned areas of PCC4 have responded very well, converting from dense sagebrush shrubland to grassland. Recovery was augmented by seeding, though it is not known how much of the current community is a result of seeding versus natural recruitment. A large portion of PCC4 within the study area has at least a moderately robust understory composed of forbs and grasses. It is also laced with an arabesque of relict channels which support mesic communities of facultative and facultative wetland grasses, rushes and sedges. Sagebrush density is higher within the exclosure which is reflected in the often-scant cover of understory plants. Fire here may produce results somewhat divergent from the grasslands further north. And yet, it is also within the exclosure that another phenomenon has been observed. Water within the reaches associated with the CH7 beaver wetland have progressively risen as sediments captured behind dams and on the floodplain have accumulated. Terrace banks around the ponded area, which is just under 1,600 m^2 in size (Morra et al. 2023), have responded to the elevated water table by producing a lush band of vegetation along the toeslope comprised of woolly sedge, clustered field sedge, and Kentucky bluegrass. This community extends up the bank slope, transitioning to rubber rabbitbrush with an understory of creeping wildrye. The tops of the terrace banks are still dominated by basin big sagebrush and rubber rabbitbrush, but the understory for a few meters back maintains a healthy population of wildrye, and sagebrush plants closest to the channel were showing signs of dieback. These observations demonstrate the ability of the surrounding terrace community to adjust to wetter soil conditions.

State and Transition Model Framework

Through data compilation, analysis, and informed theorizing, it does appear that the three developed states fit well with study area portion of Maggie Creek. Within State 1, three phases can occur with one being tied to an “E” channel (Photograph B5, Appendix B), one associated with a “C” channel (Photograph B6, Appendix B) and a beaver-occupied phase characterized by a “D_A” channel (Photograph B7, Appendix B). When external pressures reach a threshold, this state transitions into State 2, marked by a downcutting “G” channel phase (Photograph B8, Appendix B), followed by channel widening represented by an “F” channel (Photograph B9, Appendix B). It is also probable that this progression could begin with the formation of an “F” channel first, if sufficient trampling in a meadow setting caused the collapse of undercut banks. Left unchecked, the generated instability would likely promote vertical erosion leading to an incised “G” channel. The channel can alternate between these phases until sufficient pressure has been removed, allowing for a measure of recovery. These recovering channels fall within the purview of State 3. As has been mentioned ad nauseum, most of Maggie Creek is currently, and has been for some time, in this state. With thoughtful management, this state typically sees a progression of phases starting with a stabilizing analogue “C” channel (Photograph B10, Appendix B) which evolves into a “CE” channel (Photograph B11, Appendix B) and eventually forms an analogue “E” channel (Photograph B12, Appendix B) that mimics a more confined State 1 “E” channel. It is possible, though uncommon, to lock progression into a less evolved “CF” phase with steady levels of annual disturbance countered partially by a fairly consistent discharge, as witnessed at the CH1 reach (Photograph B13, Appendix B).

It was also observed that rapid bank stabilization by woody species can perpetuate a “C_E” phase of indefinite duration where the narrow, incised channel has little ability to ease its cramped circumstances through lateral expansion (Photograph B14, Appendix B). This phase was only observed at the northernmost end of the study area which encompasses the irrigated hay meadows and a few reaches just below the meadows. The signature of willows and hay meadows can be observed in aerial imagery as far back as 1966, though both were no doubt present prior to available photographic records. The circumstances leading to quick colonization by woody species, particularly willow, following incision are hypothesized to be the large-scale, pre-existing stands coupled with increased water availability supplied by irrigation runoff as well as sub-irrigation from the numerous springs present on adjacent valley toeslopes. The formation of an analogue “D_A” channel represents an additional phase tied directly to beaver activity. As discussed, this phase comes with its own evolutionary dynamic ranging from predominantly emergent wetland/open water complexes at the onset of damming to dense willow stands interrupted by periodic sedge meadows following abandonment (Photographs B15, B16 & B17, Appendix B). A graphical representation of this model is presented below in

Figure 2.12. A more complete description of ecological dynamics and phases is provided in Appendices, Appendix D.

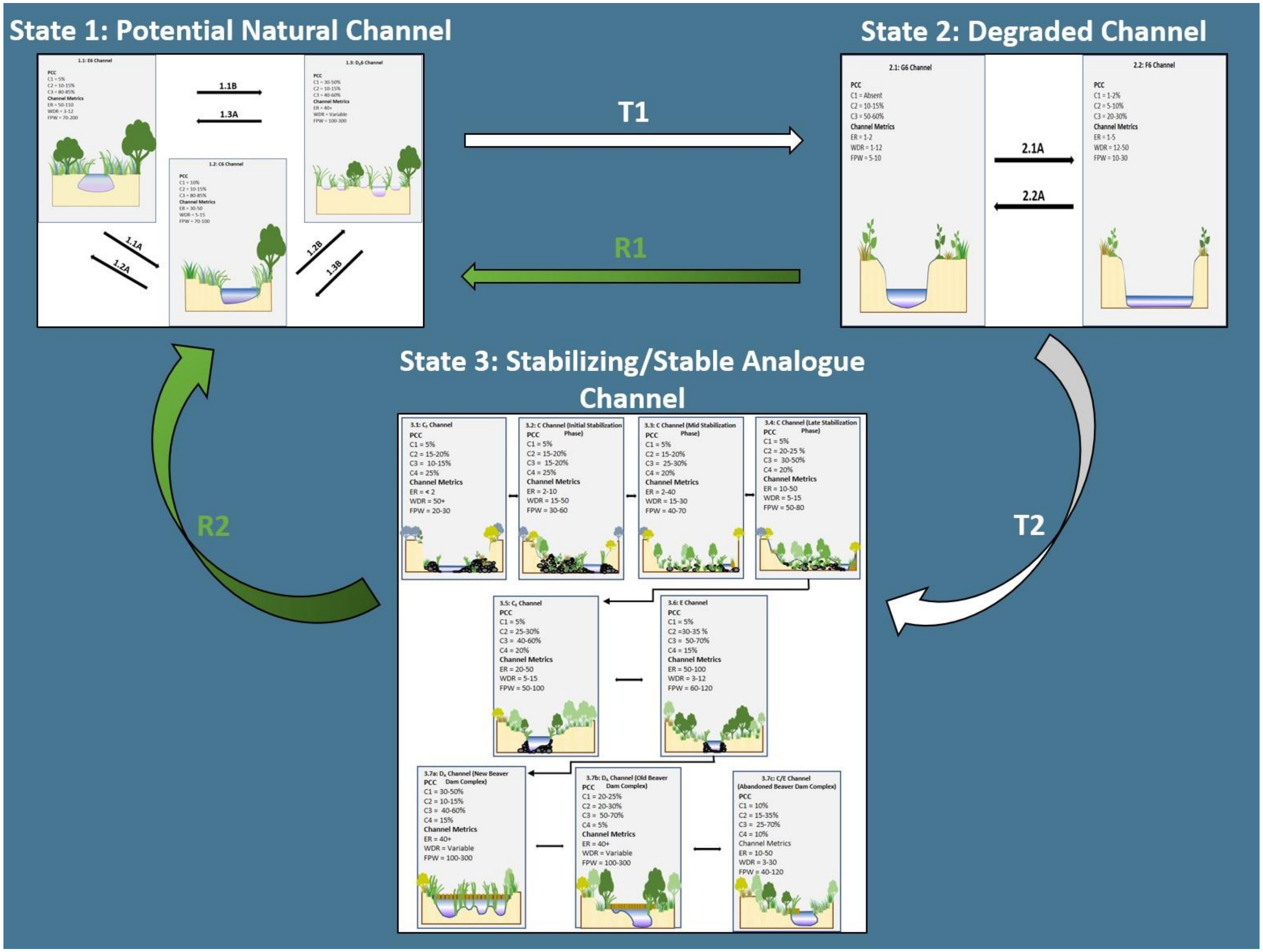


Figure 2.12. *Overview of State and Transition Model (T1 & T2 represent transitions, R1 & R2 represent restoration pathways)*

State and Transition Model in Relation to Land Use

The primary land uses at Maggie Creek are livestock grazing and hay production, both historically and contemporaneously. The key modern feature distinguishing current practices is the passive restoration begun in 1994 that altered grazing patterns, including construction of the livestock exclosure near the middle of the study area. How these uses fit within and influence the STM is at the heart of the second research question. The establishment of a railhead at Carlin, NV in 1868, near the confluence of Maggie Creek and the Humboldt River, marked the beginning of an extensive period of high use sheep, cattle, and horse grazing within the Maggie Creek watershed and surrounding basins (NDWP 2000). Records of channel erosion brought upon by large-scale floods are limited, but it is speculated that an enormous flood in 1910 probably started the main incisions still seen today (USDA 1963). The fundamental term embedded within this equation of flooding equals incision, is removal of stabilizing vegetation. Through the reading of historical documents, it can be deduced that the roughly 40-year period between the founding of Carlin and the 1910 flood experienced sufficient grazing pressure to denude much of the valley bottom. Strengthening this deduction is the photographic record of conditions within the study area in the 1970s and 1980s. These reveal that a continued history of annual, spring through fall use relegated the system to that of a wide, shallow water course working its way through mostly unvegetated gravel/cobble bars and flanked by steep, naked banks, surmounted by sagebrush and rabbitbrush (Photograph B18, Appendix B), or a narrow “F” channel configuration (Photograph B19, Appendix B). The above narrative serves to illustrate the primary mode in which Maggie Creek transitioned from State 1 to State 2 throughout most of its lower valley sections. Another trigger was channel straightening in order to accommodate hay production. A perfect example of this occurred at the north end of the study area during the late 1980's. A cost-sharing agreement between Maggie Creek Ranch and a government agency led to the straightening of several miles of "C"/" D" channel, complete with a flood berm (Photograph B20, Appendix B). Over the next few years the channel incised some ten meters forming a "G" channel, which later widened into a large "F" channel (Photograph B21, Appendix B) (Rosgen 1997). The situation was exacerbated by several years of post-construction drought, which weakened remaining vegetation. When larger-scale flooding next occurred, the straightened and leveed channel concentrated flows, increasing velocity and shear forces enough to dramatically erode the channel bottom. Other straightened reaches occur within the study area in conjunction with past haying operations, though whether they experienced the same pronounced downcutting as a result is not known. The transition into State 3 along the majority of lower Maggie creek appears to have been the result of two factors. The first is natural stabilization brought about by a decrease in erosive forces. Once the channel cut through the finer materials composing the former floodplain, it encountered coarser cobble material, which are visible when looking at the bottom of vertical terrace banks on the outside curve of channel bends. There is also a layer of dense clay present along much of the channel bottom that is visible at some of the southern "C_F" and early phase "C" channel reaches. This reduction of vertical erosion caters to a stream's natural tendency to develop a floodplain which is accomplished through channel widening by way of lateral bank erosion, which also works to reduce flood velocities (Rosgen 1997; Pollock et al. 2014). The widening process also provides space for the establishment of vegetation, further reducing stream energy (Hupp & Osterkamp 1996; Demmer & Beschta 2008). Even without a notable change in the disturbance regime, such as continuous, growing season grazing, a measure of stability is usually reached once entrenchment has ceased or slowed. This can be seen from the results of a study designed to assess channel response to the release of impounded groundwater from the open pit gold mine into Maggie Creek. Reaches above, at, and below CH1 were classified and found to be a mix of "F" and "C" channels (Rosgen 1992). This area did not undergo the change in grazing practices experienced at the outset of the MCWRP, but at least marginal improvement has occurred anyway. For example, the location of the CH1 "C_F" channel was mapped in the Rosgen (1992) report as an "F" channel. A much more dramatic upward trend is evident in the rest of the study area that switched from summer grazing to periodic spring/fall grazing. This form of passive restoration marks the second, and more expedient factor responsible for the transition from State 2 to State 3.

The creation and populating of floodplain features within the incised terrace banks were greatly facilitated by the easing of pressure during the summer "hot season" months. Livestock, exhibiting much good sense, naturally congregate along water courses as air temperature increases, both to relieve their thirst and to take advantage of the cooling aspects associated with riparian areas. Unfortunately, this often facilitates bank damage, channel widening, and a reduction of vegetation through consumption and mechanical damage. With a resurgence of plant growth, especially willows, another chain of events was set into motion; the reoccupation of Maggie Creek by beavers. Reading the "warm and fuzzy" literature pertaining to Maggie Creek's impressive metamorphous from scrappy, gravel bar wandering stream to lush riparian nirvana, leads one to believe that the beavers mysteriously appeared one day, conjured up much like the strange apparitions summoned by the three weird sisters to soothe Macbeth. This is not true. A thoughtful perusal of available materials offers clues that beaver refugia have persisted along the creek for a long time, especially where actively and formerly irrigated hay meadows have sustained islands of dense willow thickets. In fact, the presence of beaver is specifically called out in stream survey records going back to 1980 (note: though on private land, a partnership with the BLM fostered agency-led monitoring for about 20 years in and around the study area). It is however, as I step down from my literary high horse, very much true that beaver populations greatly expanded and grew in the years following implementation of the MCWRP. The resulting water-spreading, sediment accumulation, and channel roughening has been instrumental in progressing the assemblage of stable and near-stable analogue channels witnessed today. Though limited in occasion, there have also been attempts to move reaches in State 2 back to State 1. This involves active restoration and has thus far involved large-scale efforts using machinery, and more recently, low-tech, process-based actions involving temporary rock structures. In the first case, the aforementioned channel straightening-to-incised channel was addressed by Dave Rosgen in 1991 with a “plug and pond” approach (Rosgen 1992). Several miles of the entrenched channel were buried using the fill material from a series of adjacent, excavated ponds. This effort successfully reconnected a “C” type channel to its original floodplain which later took on attributes of an “E” channel as vegetation filled in, reducing the width to depth ratio (Photograph B22 & B23, Appendix B). The low-tech approach took place in the fall of 2022 at the deteriorating meadow site below the CH3 reach. It was instigated as part of a continuing education course taught by UNR staff and sanctioned by Maggie Creek Ranch. Headcuts were laid back and armored, numerous in-channel, one-rock dams and deflection walls were built and willows were planted next to the stream (Photograph B24, Appendix B). The immediate objective was to prevent further movement of instream and lateral headcuts. In the long term, it is hoped that subsequent efforts and time will be sufficient to bring at least the upper portions back towards conditions present at CH3. Restoration pathways back from State 3 to State 1 are both not recommended and not realistically feasible from a management standpoint. The rejuvenation of natural processes expressed following changes in grazing patterns, and later dilation of beaver colonies, suggests that riparian vitality and functioning is and will return without further inducements. The scale of disturbance and expense necessary to bring the current channel up to meet its former floodplain would be extremely high and, in some cases, would require infilling terrace widths in excess of 250 meters…with no guarantee of success. There are, however, locations, such as near CH7, where near-permanent beaver dams, in-place for nearly 20 years, have already come close to reconnecting the channel to the old floodplain. For reaches in State 3, the best course of action seems to be a continued practice of well-managed use coupled with patience. This applies to irrigation practices as well. Earlier practices tended to focus on production at the expense of the system. With a slightly different mindset, there can be rewarding benefits of flood irrigation that benefit the system while still providing hay and pasture for livestock. These will be discussed in detail in the next section. In the meantime, it is worth exploring the final land use at Maggie Creek, which is the grazing exclosure. Given riparian conditions in 1993, the construction of the exclosure in the following year was no doubt a great boon in terms of jump-starting the revival of herbaceous vegetation followed by willow establishment. Technically, some grazing was allowed within the exclosure for two years after it was built, but unmanaged grazing effectively ended with the stringing and stapling up of the last strand of wire. Its long tenure as a barrier to grazing, aside from the occasional trespass cow, begs the question; should it maintain its agreed upon duties into perpetuity? To resolve this question well enough to complete this exercise in model building, it is warranted to turn to this project's channel data; personal observation; a supplementary data set from plant production plots collected in the study area to support a concurrent study; and the growing body of literature pertaining to the surprisingly complex world of riparian exclosures and pastures. To first place this bovine obstacle in better context, its southern boundary crosses Maggie Creek just north of CH6, while its northern side passes upstream of CH9 for a valley distance of four miles (Figure 2.2). It is inclusive of the entirety of the incised floodplain and large portions of the terrace. Discounting the water gap site at CH8, the sampled locations within the exclosure are limited to CH7 and CH9. Considering sampled reaches in State 3, the CH9 site is the only one that met the specifications to be considered an "E" channel. The CH7 reach is within the previously described marshy, emergent wetland and classifies as a "D_A" channel. They have the two highest ASS scores, possess BHRs indicating stable banks, and aside from the "DA" channel at CH10, have the highest ERs. The CH9 site has the lowest WDR while CH7 has the fourth lowest. Based on these statistics, it would seem that the removal of grazing has been highly successful in accelerating recovery within the exclosure. However, there is reason to believe that a continued prohibition on grazing is not necessarily desirable. From aerial imagery, BLM stream survey reports, and remote sensing projects (see WHA 1995 & WHA 2011), it can be determined that beaver populations had expanded through the entirety of the grazing excluded reaches by at least 2006. From this it can be assumed that a discontinuance of grazing greatly helped create suitable conditions for beaver colonization. However, from personal observation, much of the big sagebrush community of the terrace is now quite dense with little available space for the development of an herbaceous understory. By contrast, terrace sites outside of the exclosure tend to have a more open shrub canopy due to mechanical damage from cattle. These observations are somewhat corroborated by the referenced supplementary plant production data set. This information comes from data collected by the author of this dissertation to help out a fellow graduate student studying soil carbon stocks in respect to riparian condition and functioning (see Morra et al. 2023). Six plots were established on the terrace west of CH9 and six on the same feature west of CH12. Plots that fell within relict channels or next to Jacks Creek, a tributary stream entering Maggie Creek downstream of CH9, were highly productive, supporting a mix of grasses and sedges. This was also true of plots close to the channel at CH7 where an elevated water table has allowed for the migration of graminoids, especially creeping wildrye, into the understory. This area has experienced some die back of sagebrush from wetter soil conditions as well as periodic harvesting of this species for dam material by beavers. Understory density and diversity was lower at plots with a relatively closed sagebrush canopy.

In light of these findings, it is quite possible that a resumption of managed grazing that restricts season of use and incorporated periods of rest, would be beneficial in increasing graminoid cover and diversity on the terrace. The terrace is no longer part of the riparian corridor, but should bed aggradation reach a point where flood waters once again spread across this feature, a well-vegetated plant community is necessary to prevent potential lateral erosion where waters come back to the channel. In the riparian area itself, conditions have improved to the point that periodic use should not cause a reversion to former, degraded conditions, especially if use is limited to the spring and fall. In a study conducted along streams in Idaho, Nevada and Utah, Platts and Nelson (1985) reported a 13 percent higher use of adjacent upland forage when cattle were allowed access to the research pastures during the spring versus the summer. Another Idaho study found light to medium spring grazing maintained or improved riparian pastures in a mountain meadow setting (Clary 1999). A preference for upland vegetation in the late summer/fall was noted during a study conducted in Oregon (Bryant 1982). Additionally, though the ramifications of improper grazing are well-documented, there are several benefits of proper grazing that should be considered. These include stimulating herbaceous plant growth by trampling and consuming decadent plants; reducing thatch and incorporating it into the soil and increasing forage quality and diversity (Clary 1995; Clary 1999; Boyd & Svejcar 2004; Swanson et al. 2015). Within the growing season, regrowth following herbivory tends to decline in later months (July and August), again suggesting that spring use or post-growing season fall use is more appropriate. However, the resiliency of riparian zones is often sufficient to counter occasional summer use providing adequate post-grazing rest is built into the grazing plan (Swanson et al. 2015). Whether the Maggie Creek exclosure remains or becomes a managed pasture(s) is yet to be determined, but the implications of either choice are worth presenting here, even if they cannot be fully predicted by a generalized STM. As a final note, this exclosure offers good potential for future riparian grazing studies. In an extensive review/critique of riparian exclosure studies, Sarr (2002) mentioned that many exclosures were constructed to represent reference conditions, even though the areas fenced off had a long history of grazing. Another main issue is that many of these features are small in size, creating conditions not representative of the greater area being studied. In this case, the fence was constructed to promote passive restoration, rather than represent undisturbed conditions. It is also quite large, encompassing well over 1,000 acres and at least some data, both survey and photographic, are available to showcase conditions prior to its construction.

State and Transition Model – Riparian Ecosystem Service Elements

This research incorporated several elements have previously not been explored in other riparian STMs. These are bird diversity, stream temperature and carbon sequestration. It would be lovely to be able to definitively proclaim the expected amount and character of each of these elements in relation to a given state and phase of the model. Unfortunately, the necessary commitment to attempt such a feat was not within the time scale of the project or energy of the graduate student. But, it is definitely possible to point out some generalities. Bird diversity manifested a distinct pattern based upon channel type and associated vegetation. At Site 1, inclusive of the CH1, "C_F" channel, the riparian area consists of exposed gravels and cobble and an herbaceous streambank community comprised of early-seral graminoids and forbs. Coyote willow is beginning to colonize the occasional gravel bar, but is thus far a minor component of riparian plant communities. Compared to the other six bird sampling sites, diversity here was roughly a third of that recorded at the most diverse site and approximately half that of the second lowest diversity score. Additionally, species recorded at riparian points were often the same as those picked up at adjacent upland points. Occasional overlap occurred at other sampling locations, but a more distinct separation was evident between species recorded at riparian plots versus upland stations. Beyond the straight-forward correlation of less-developed riparian equals less diversity, patterns also emerged pertaining to type of vegetation. The highest diversity values were calculated for the three riparian locations at the north end of the study area. These sites were not significantly different when compared to each other and all exhibit heterogeneity within the woody plant community in terms of species and age class. This community is also well represented within the riparian corridor, extending from water's edge to the lower terrace slope is most places. Woody riparian species are particularly important to perching (i.e. passerine) birds and documented declines in this type of habitat have demonstrated similar declines in bird species (Sprunt 1975; Krueper 1993). Thus, it is not surprising to witness higher diversity along the late-phase "C_E" channels that, within the study area, are prone to woody species development. Interestingly, the State 3 "E" channel at CH9 (included within the Site 4, point count area), though double that of CH1/Site 1, had the lowest diversity when compared to all other sites. This is thought to be due to two reasons. One is that much of the sampled riparian area is within a new beaver wetland complex composed of, much like at CH7, emergent marsh vegetation and relatively low cover of woody species. This is exemplary habitat for some species, such as *Geothlypis trichas* (common yellowthroat), *Agelaius phoeniceus* (red-winged blackbird) and *Cistothorus palustris* (marsh wren), but not conducive to supporting the more diverse assemblages found in association with areas of high woody species development or a combination of woody and herbaceous habitat. An example of the latter occurs near CH7, where the marshy vegetation present at the channel data collection site progressively becomes more of a scrub-shrub wetland downstream at older beaver wetland locations. This transition was captured within the greater area of the Site 3 bird survey and diversity was indeed higher (Site 4 = 1.55, Site 3 = 2.09). The second reason diversity was lower is attributed to the time of day that sampling occurred. Initially, it was hoped to sample each location at least twice, once in June and once in late July/August. One survey would have taken place during the more active early morning period and the other during the evening in hopes of capturing any species more prone to dusk/nocturnal activity. Unfortunately, time only allowed for one sampling effort with some areas being sampled in the morning and others in the evening. Site 4 was one of the evening locations, and thus, it is quite possible full species representation was not attained. As a side note, the low diversity Site 1 was sampled in the morning. Diversity at Site 2, characterized by early phase “C” channel reaches, fell between that of Site 1 and the more developed riparian areas found at Sites 3, 5, 6 and 7. Though much more robust studies, inclusive of multi-sampling periods/years and more detailed habitat descriptions, are recommended before drawing any substantial conclusions, there does appear to be an overall trend of increased bird diversity with increased riparian complexity and a greater divergence in bird composition between riparian and adjacent upland areas.

As called out in the Results Section, stream temperature data was not extensive and much of what was collected was compromised due to loggers being stranded above water for periods of time and/or claimed by Maggie Creek during the flooding of 2023. The placement of surviving data within the context of the STM seems most appropriate from an expected outcome of land use, namely flood irrigation, perspective. As previously discussed, real problems can arise when a stream is modified in such a way to achieve a desired goal at the expense of the riparian system. However, this does not have to be the case and inefficient, flood irrigation appears to be a good example of suitable hay production coinciding with riparian health maintenance and improved wildlife habitat (Essaid & Caldwell 2017; Gordon et al. 2020; Donnelly et al. 2024). Meadow shrinkage was the predicted result should flood irrigation cease at a stream/meadow site in east-central Oregon (Stringham 1996). In this study, the level of summer stream temperature buffering observed within the lower hay meadow reaches has great implications for cold water fishery management, especially regarding Lahontan cutthroat trout. Ultimately, it would be good to separate out irrigation returns from subsurface seep/spring contributions, but even the limited data from this study suggest that irrigation is likely playing at least some role in keeping stream temperatures cool and stable through the hottest months. Though less pronounced, the lower temperatures recorded below the mouth of Coyote Creek also highlight the importance of maintaining tributary health as part of the larger aim of stream management.

The irrigated hay fields also feature prominently in the final element considered, that of soil carbon storage. Elevated releases of carbon to the atmosphere related to changes in land use, followed by changes in climate patterns, have been a concern within various scientific disciplines for at least the past 50 years (Stuiver 1978; Hansen et al. 1981; Schnitzer 1991; Lorenz et al. 2017). Along with these concerns has been the recognition that soil carbon stocks can be quite high in areas such as grasslands and riparian zones, thus promoting both their importance and need to maintain functioning systems and restore degraded ones. Overall mean carbon stocks were highest for the floodplain surface near CH9 (14.6 kg/m^2), followed by terrace sites around CH12 (12.8 kg/m^2) and

CH9 (11.2 kg/m^2), respectively. The hayfield terrace was among the next highest sites (5.8 kg/m^2) along with the combined average of terrace sites that coincided with CH6. Factoring in surface covered by geomorphic feature, the terrace sites at the hayfields and adjacent to CH12 had the highest total amounts of soil carbon. The hayfield dataset from the active floodplain was incomplete, but available samples had a mean slightly lower than the terrace. Combined, these results do offer at least a rough correlation of soil carbon stocks to phases within State 3. Or more accurately, a rough proportional gauge. As seen here, beaver-related wetlands with a D_A channel can be expected to have elevated soil carbon stocks due to expanded width of the floodplain, high plant productivity and anaerobic conditions that slow down organic matter decomposition. Beyond this, floodplain sites were narrower and either currently lacking beaver dams or possessed of a series of small dams with less occupied floodplains. Carbon amounts varied by site, but not dramatically. The early “C” type channels present along CH6 had lower soil carbon than found at CH9 or the hay meadows. This makes intuitive sense as lower amounts would be expected for sites with less floodplain vegetation and more consistent perturbation from floods. However, the more degraded floodplain around CH1 actually had higher amounts of soil carbon than the other three locations. Barring sampling error, the higher levels are not readily explained, but may be attributed to flood attenuation brought about by the managed release of water to the creek from the cooling towers. Active floodplains experiencing frequent inundation have been documented as having lower soil carbon than features adjacent to floodprone surfaces (Doetterl et al. 2016; Saint-Laurent et al. 2017).

The terrace is largely independent of the current State 3 channels, but overall carbon sequestration is much higher near CH7 and CH9, followed by the managed grazing site by CH6. The hay meadows and CH1 terraces had the lowest and similar amount of soil carbon. This points towards land use, both historically and more recently, and conditions upon the terrace. There is some potential that the higher water table associated with the CH9 wetland complex has at least partially reconnected the network of relict channels on the adjacent terrace. Half of the sampled plant community plots for the carbon study were located within or just above these channels and had a mix of dry and mesic meadow-type species. The deepest channels have also been observed to hold water during the spring/early summer and maintain a community of predominantly wetland obligate and facultative wetland graminoids. From a land use standpoint then, practices that encourage beaver colonization may lead to more carbon sequestration on the terrace, even if it is largely disconnected from the current channel. This obviously would be facilitated by the presence of numerous low-lying features, such as the relict channels. These channels also occur upstream in the vicinity of CH9, although terrace hydrology appears to be more affected by Jacks Creek than Maggie Creek. This tributary had two channels that enter Maggie Creek in this area and though not technically perennial, flows persist through much of the summer. The floodplain is also connected to the terrace except near the mouths of the channels, where the elevation drop into Maggie Creek has created headcuts and subsequent channel incision. As a result, many portions of the Jacks Creek floodplain and adjacent, relict channels support dry, verging on mesic meadows. Careful grazing management and shrub thinning projects could potentially increase carbon sequestration. The headcuts have mostly stabilized courtesy of infrequent high flow events and establishment of coyote willow within the channel from the head cut areas back to Maggie Creek. The hay meadow and CH1 terrace are somewhat puzzling. Due to regular irrigation and production of luxurious stands of hay, it was expected that carbon sequestration would be high at the hay meadows. Clearly the removal of the hay crop limits soil organic matter inputs. But it was thought that overall rooting structure and incorporation of material by pastured cows during the fall would be sufficient to maintain higher levels of soil carbon, at least when compared to CH1 terrace. A study in Italy found that when comparing various crop rotation systems to a no-till meadow that was hayed, carbon actually increased over a 30-year period and plant nitrogen use was more efficient (Castelli et al. 2017). The similar sequestration recorded at the CH1 terrace may be attributed to past use. In the original mitigation plan for releasing intercepted groundwater to Maggie Creek, it was stated that a portion of the water stored in the detention reservoir would be diverted for irrigation purposes (Rosgen 1992). How this plan was enacted is not known, but irrigation risers, likely for a side-roll system, are present along the edge of the terrace downstream of the reservoir and cooling towers. There is also evidence of land of past land clearing, and though big sagebrush is now ubiquitous, dense canopy only occurs in a strip above the current floodplain. Perhaps former irrigation and hay production were sufficient to build up soil carbon prior to the drier, more upland conditions observed today. It was originally hoped that the carbon data would be sufficient to link carbon sequestration potentials to the STM, which could then be linked to carbon trading by way of providing a means of assessment without frequent, and expensive soil sampling. Aside from the broad generalities mentioned above, this seems not the case, especially as data from the State 1 meadow is lacking along with impacts resulting from a transition to State 2. However, the implications of carbon storage and management should at least foster a greater amount of consideration when developing riparian management plans. For example, the overall higher accumulation on the terrace versus the floodplain suggests that despite being generally disconnected, it is worthwhile to identify areas that may have alternate hydrological inputs through surface discharge from tributaries and ground water interception along former floodplain features. Despite being outside the traditional riparian zone, applied management here may increase potential carbon storage and help facilitate floodplain connectivity as stream conditions improve and channel bed aggradation increases.

Application of the State and Transition Model to Similar Great Basin Systems

To determine if the Maggie Creek STM has applicability to other, similar streams, the logical starting point was Susie Creek. This creek lies about 10 miles to the east, has numerous sections of low gradient channel within Valley Type VIII, underwent passive restoration similar to and concurrent with the MCWRP, and contains beaver. The longest record of conditions along Susie Creek consist of photos that extend from recent times back to 1978. Quantitative data is more limited, but was collected in 1978, 1989, 1994 and 2017 (BLM 1995; BLM 2017). There are seven photo point/stream survey points along the lower eight miles, all located on BLM land. They are also within a riparian pasture that was built in 1990 to control livestock access, which has been intermittent since construction. All reaches within this area are incised with an average of about 50m between incision banks, but many areas exceed 100m. From the first three years of sampling, the average WDR increased from 24 to 50 in 1989, then reduced down to 15 in 1994, four years after the riparian pasture was fenced. Sedimentation was also much reduced by 1994 and mean percent bank cover and bank stability increased by 20 percent and 60 percent, respectively. Unfortunately, the 2017 survey was more subjective and only included bank cover and stability as comparable metrics. Average cover increased an additional 17 percent from 1994, though stability was reduced 15 percent. As a whole, the pattern at Susie Creek matches well with Maggie Creek with past vertical and horizontal erosion eventually stabilizing into a State 3 condition following a shift from summer to occasional spring/fall grazing and expansion of beaver populations (Photograph B25, Appendix B). One difference between the two systems is that sediments along Susie Creek have a much higher proportion of sand, silt and small gravel. Without the "bottoming out" effect observed at Maggie Creek, where vertical incision has essentially stopped, this could necessitate an additional "at risk" phase indicative of a transition back to State 2. In the 1978 photo, Photograph B25 – Appendix B, it can be seen that the current channel is well within steep incision banks, but has begun to incise again. This process has been alleviated following the passive restoration and beaver occupation, but is worth noting as a potential problem. Reading reports, there is also some concern that beavers may have reoccupied Susie Creek before it was "ready". The BLM biologist responsible for initiating so much of the restoration, Carol Evans, has mentioned that willow mortality through beaver use and drowning, may have set back recovery to a degree and it may have been better if they had delayed their return by 5-10 years (NU 2015). Flooding in 2017 also removed many dams formerly observed at the various photo/sampling points. None the less, beavers still persist and patterns are reminiscent of the beaver wetland succession in place at Maggie Creek. A little more data is actually supplied to the model where cattail/bulrush/open water wetlands have lost dams. Dam breaching was only observed at Maggie Creek in wetlands with substantial willow populations. At Susie Creek, the emergent wetlands are now transitioning towards large, open meadow communities populated with sedges and grasses (Photograph B26, Appendix B). Willows are also colonizing and infilling at other reaches, which is consistent with development towards the scrub-shrub wetland phase (Photograph B27, Appendix B). Although pre-incision conditions are less well documented than at Maggie Creek, it can be assumed that they would have been similar and at the very least, the riparian corridor would have been much wider. By incorporating an additional beaver wetland phase and recognizing the potential for a reversion to State 2, when a relatively impervious layer is not present, it appears that the Maggie Creek STM works for the nearby Susie Creek. Species composition data was not available, but appears similar from the photo log. The finer sediments are suitable for rapid establishment of wetland graminoids, which would likely necessitate adjustments to the proportions and makeup of PCC2 and PCC3.

Conclusion

State and transition models provide a useful framework in which to distil down the interrelated, intricate processes inherent in riparian systems. Highlighting features that can serve as indicators of underlying function allows managers to focus their attention and resources thereby increasing the chances of recognizing at risk sites before they become major problems. These models also provide a measure of guidance when contemplating current and future land use. It is beyond the scope of an STM to contrast and compare the fine-scale effects of different disturbances, but the emphasis on underlying processes allows for a level of predictive power in terms of recognizing when a state or phase shift is about to occur and the implications of such transitions. This study reinforced ideas framed by others (Stringham & Repp 2010; Ratcliff et al. 2018; Meehan & O'Brien 2019) considering the application of quantitative data and pattern analysis in determining states, phases, thresholds and overall riparian resiliency. The applicability of the Maggie Creek STM to other, similar streams was also proven, though it is recognized that a more rigorous testing should be conducted at other locations before accepting its broad suitability for additional, low-gradient systems in the region. The data exists, but sufficient time to incorporate it into this study, unfortunately, did not.

The general implications of this effort are that these type of riparian environs are resilient and show a remarkable ability to recover from even severe degradation. The level of human-caused and natural disturbance that caused much of the Maggie Creek riparian area to become much reduced in capacity was acute, yet a level of recovery was evident following shortly after simple changes were made to grazing patterns. This was accelerated exponentially following the diaspora of beavers from isolated refugia into the whole of the study area and beyond. This does not suggest complacency, but rather promotes the idea and benefit of proactive, vigilant monitoring and management adjustments. The STM then provides a roadmap in which to navigate, compare and steer such adjustments. Though cursory in depth, the inclusion of model augmentation demonstrated that bird diversity, stream temperature and soil carbon sequestration, all important elements of riparia, are enhanced with increasing size, complexity and functionality of the system. It is the hope of this author that this study will not only aid in maintaining and improving analogous riparian areas, but will serve as fertile ground for future research, especially concerning remote sensing possibilities and carbon sequestration potential.

Chapter 3: Remote Sensing Applications

Abstract

Remote sensing techniques offer exciting opportunities to assess areas that are typically under-represented in monitoring efforts due to access issues or simply the vast size of areas under management. Increased resolution and number/type of sensors provides a high degree of flexibility and power from resulting from imagery procured from satellites and other airborne craft. This study made use of high-resolution NAIP imagery (0.6m) from 2019 and 2022 to test the ability of 12, imagery-derived indices to classify selected land cover types (water, gravel, vegetation) along Maggie Creek, a small, low-gradient stream system in northeastern Nevada. The best results were achieved using NDVI in combination with either GLI or VDVI. The improvement over NDVI alone was very small, leading to the use of only NDVI for the rest of the study. This consisted of determining the average NDVI value for 14, 250m reaches along Maggie Creek. These reaches were previously used to develop a riparian complex state and transition model. NDVI values were assessed within the context of the model in order to see if a given channel feature could be tied to NDVI values sufficiently enough to predict states and phases, which could then be applied to other, similar stream systems. Flood prone width was the chosen channel metric as it reflects other, less-detectable features and can usually be delineated from imagery alone. It was hypothesized that wide floodprone widths would correlate to high NDVI values and functioning states/phases, while low values

would be expected for narrow flood prone widths consistent with more degraded states in the model. This was partially true. The State 1 – Potential Natural Channel was easily distinguishable from State 3 – Stabilizing/Stable Analogue Channel based upon these features and some State 3 phases also had recognizable NDVI/floodprone width signatures. The approach was not sufficient to detect all states and phases within the model and required substantial effort to train spatial data during the initial land cover classification effort. However, potential applications, such as detecting short-term changes in riparian condition and riparian restoration monitoring were evident following the conclusion of this study.

Introduction

The vast scale of landscapes in the western U.S. alone make it challenging to adequately assess conditions across a wide area, especially on public lands where staffing and budgets within land management agencies are often limiting factors. Recent advances in camera sensor technology and free access to finer resolution satellite data have increased possibilities of remotely assessing rangeland for management purposes. This has particular relevance to riparian areas, especially on small streams, where previous Landsat data, though free, is often too coarse to be of much use beyond long-term, trend studies (Congalton et al. 2002; Huylenbroeck et al. 2020). Conversely, resolution increases dramatically with the application of imagery taken from either manned or unmanned aircraft. Obtaining drone imagery is less expensive, and less dangerous, than low-altitude, manned flights and has a greater degree of flexibility in terms of repeat flights, different flying heights and timing of flights (Rango et al. 2009; Yang et al. 2019). However, there is still the initial cost wall of purchasing a drone, training personnel and post-flight data processing. A decent compromise is the National Agriculture Imagery Program (NAIP). This program began in 2003 and acquires flyover imagery that spans the agricultural growing season across the U.S. NAIP imagery, at 1-meter resolution and with four bands of data (Red, Blue, Green, and NIR), became possible for all states beginning in 2010, allowing for the calculation of various indices, such as NDVI. In 2019, resolution increased to 60-centimeter resolution (USDA 2022). Imagery is available and free through the NRCS's NAIP GeoHub. There are drawbacks, especially with flight frequency, which, since 2009, is on a three-year basis, unless user needs are sufficient to warrant a shorter recurrence (USGS 2018). As a relatively young program, there is also no opportunity to conduct long-term trend studies. Another potential option is using imagery obtained from the Sentinel 2 satellite missions. Resolution is lower (10-meter) and data only goes back to 2015, but fly-over periods are much more frequent (every 5 days) and additional sensors provide 13 spectral bands to work with, allowing for the calculation of more complex indices (Kaplan & Avdan 2017). This data is also free and can be accessed from the Copernicus Open Access Hub (https://scihub.copernicus.eu/).

For this project it was decided to make use of NAIP imagery for its higher resolution. The primary goal was to see if any aspect of channel morphology or riparian characteristic could be detected and linked to a given state and/or phase within the model. The idea being that the model extrapolated through remote sensing would serve as way for managers to garner an initial assessment across large areas, thus allowing ground reconnaissance to be more focused and efficient. Being able to recognize at least different states would help prioritize areas that may be transitioning into a more degraded condition, thus making them prime targets for restoration efforts.

Materials and Methods

Study Area and Approach

The logical place to start was the same Maggie Creek study area footprint used to develop the STM. The majority of states and phases have quantitative data, allowing for direct comparisons between on-the-ground conditions and imagery features. The same holds true for image classification as intimate knowledge of the location allows for confident assignations of pixels to desired categories (e.g. water, herbaceous vegetation). As an entry point, it was decided to calculate various indices, such as Normalized Difference Vegetation Index (NDVI), to see if differences could be detected between various locations in the study area and to see if a single index, or combination of indices, worked better in this regard (Table 3.1). Indices allow for the gathering of specific, detailed information beyond which can be obtained from true color images (Mehrotra & Srinivasan 2019). In particular, manipulation of the blue, red, green, and near infrared bands can help distinguish healthy from stressed vegetation; separate water and bare ground from plants; reduce atmospheric interference and aid in many other desired applications. As such they can be very useful when monitoring landscapes, especially in the context of land health, agricultural production, forestry, and conservation (Xue & Su 2017). To start the process, the most recent NAIP imagery for the area, summer of 2022, was downloaded and imported into ArcMap®. Sets of “training” points were created that covered a variety of conditions within the study area. Points were placed to capture a large subset of riparian features which were categorized as Water, Gravel and Vegetation. Additional polygons were created at selected, 250-meter reaches at each of the channel data collection sites from Chapter 2 (CH1-CH14). Training and reach polygons were then brought forward into the analysis. As conditions such as light intensity, time of day and time of year can affect reflectance, this same process was repeated for summer of 2019 imagery in order to average results for a more accurate assessment.

***Table 3.1**. Description of Remote Sensing Indices used in Random Forest Analysis*

Index[1]	Formula[2]	Description
C3	arctan(B/(max(R, G,NIR)))	Shadow detection
CVI	NIR * R/G^2	Chlorophyll detection
EVI	2.5 * ((NIR – R)/(NIR + (6*R) – (7.5*B) +1))	Enhanced vegetation index for areas of high biomass
GARI	(NIR – (G – (1.7 * (B - R))))/(NIR + (G – (7.5 * (B – R))))	Less sensitive to atmospheric effects and more sensitive to chlorophyll concentrations than NDVI
GLI	((G - R) + (G - B))/((2*G) + (B + R))	Detects green leaves and stems
GRVI	NIR/G	Sensitive to changes in leaf pigments
MSAVI	(2 * NIR + 1 - √((2 * NIR + 1)^2 – 8 * (NIR – R)))/2	Minimizes effect of bare ground on SAVI
NDVI	(NIR – R)/(NIR + R)	Measure of plant “greenness”
NDWI	(G – NIR)/(G + NIR)	Detects water bodies utilizing the strong absorbability and low radiation of water in the visible and near infrared bands
SAVI	(1.5 * (NIR – R))/(NIR + R + 0.5)	Used in areas with sparse vegetation to suppress effects of soil pixels
VDVI	((2*G) – R – B)/((2*G) + R + B)	Extracts NDVI values using only the visible portion of the electromagnetic spectrum
WDRVI	((0.1 *NIR) - R)/((0.1 * NIR) + R)	Sensitive to higher leaf area indices than NDVI

[1] All indices except for NDII were calculated for both NAIP and Sentinel data. The shortwave infrared band is only available for the Sentinel imagery.
[2] Band designations: R = Red, B = Blue, G = Green, NIR = Near Infrared

Statistical Analysis

All statistical analyses were performed using R version 4.3.2 (R Core Team 2023). Desired imagery polygons were imported from ArcMap in Tag Image File format (TIFF) and converted into multi-layered raster objects using the brick function from package “raster” (Hijmans 2023). The array of indices brought into the analysis were calculated from raster bricks. The writeRaster function was used to create raster objects from each index and combined with reach TIFs using the stack function. Both of these functions are from the “raster” package. The next step was to read in a training TIF for a given reach and extract spatial data in relation to the training points using terra::extract from the “Terra” package (Hijmans 2024). Spatial data was stored in a data frame and corrected for missing values. A series of regression models were then created using the randomForest function from the “randomForest” package (Liaw & Wiener 2002). RandomForest is a supervised, machine-learning algorithm that runs through a series of decision trees to arrive at a single solution (Belgiu & Drågut 2016; HUB 2023). In this case, it used training data from the data frames to classify pixels of a selected reach into one of the predetermined categories using the majority decision reached among all the trees and the suite of independent variables included in the model. The model results provide the total estimated amount of classification error and the class error for each category. The first model incorporated all indices and, as separate variables, the Red, Blue, Green and NIR bands. Using the importance function from the “RandomForest” package, it was possible to determine which of these independent variables provided the strongest prediction results based upon the mean decrease in accuracy as computed from the individual tree errors. Additional models were tested using combinations of the indices with highest importance values. This process provided the “best-fit” model and reduced the number of independent variables. Graphical representations of each prediction were also created by first using the predict function to create a raster object based upon the random forest model. This was then imported into a level plot courtesy of the “rasterVis” package (Lamigueiro & Hijmans 2023). The resulting visualization helped determine where the majority of classification errors were occurring and mask out problematic pixels, yielding a tighter prediction.

Two tibbles were then created with columns for cell number and each variable from the “best fit” model in one, and cell numbers and pixel category from the predicted model in the second. These were merged with a defined mutation such that a new column was formed for classification and one broken out by “vegetation” and “not vegetation”. Summary statistics could then be calculated for a desired index providing the mean of actual and predicted values as well as adjusted to separate out water and gravel from vegetated pixels. The creation of tibbles and subsequent steps were performed using functions from the “dplyr” package (Wickham et al. 2023). A final step was to try and link the riparian vegetation aspect of NDVI to a channel characteristic. Few channel metrics can be determined from imagery, but many can be intuited based upon width of the floodprone area. A fully occupied riparian zone, such that vegetation extends to the limits of topographical boundaries, can be indicative of minimal vertical incision, low bank height ratios, frequent floodplain inundation and similar factors. The floodprone width can also be observed and measured using mapping programs, making it a useful metric to employ, at least for initial, desktop assessments. This is not a precise practice and can still be difficult to determine when canopy is dense, especially in forested systems, but is more readily discernable in the low gradient, semi-arid systems relevant to this study. For this project it was possible to use the known floodprone widths calculated at the sample reaches for building the STM.

Results

The full random forest model yielded classification error rates ranging from 0-14.89 percent depending upon location. All reaches within the incised portions of Maggie Creek had error rates between 0-2. A lack of complexity along the lower reaches (CH1, CH4-CH6), coupled with distinct pixel reflectance values (e.g. exposed gravel, water and vegetation pixels differed substantially from each other) accounted for the very low results. The 14.89 value was associated with the meadow area at CH2 and CH3. The greater difficulty in classifying pixels lent itself well to paring the model down by finding the index or indices were best able to detect differences between different pixel classifications. Individual color bands and Near Infrared (NIR) had the highest rates of error, which was reduced to 15.25 when combined (Table 3.2). These four band were retained for subsequent model tests as all indices are calculated from them. The importance values associated with the full model indicated that least useful indices were NDWI, WDRVI and SAVI (see Table 3.1 for index descriptions). The estimate error using the remaining indices was slightly higher (15.6) than the full model, but fell lower (14.54) when removing EVI and MSAVI. Continuing the process, the best results were obtained when using only NDVI in combination with either GLI or VDVI (13.48). Singularly, the error rate for these three indices increased to 13.83 (Table 3.2).

Table 3.2. *Random Forest Model Results – NAIP Imagery (C2-C3 Meadow Location)*

Model	**Estimate of Percent Classification Error**[1]
Full Model (All indices)	14.89
Red	26.95
Blue	18.09
Green	16.67
NIR	29.08
Red + Blue + Green + NIR	15.25
All Bands + NDVI + GRVI + CVI + C3 + GARI + GLI + EVI + VDVI + MSAVI	15.6
All Bands + NDVI + GRVI + CVI + C3 + GARI + GLI + VDVI	14.54
All Bands + NDVI + GRVI + CVI + C3 + GARI	16.67
All Bands + NDVI + GRVI + CVI + C3	16.67
All Bands + NDVI + GRVI + C3	15.96
All Bands + NDVI + GRVI + CVI	15.96
All Bands + NDVI + GRVI	14.89
All Bands + NDVI + CVI	14.89
All Bands + NDVI + C3	14.89
All Bands + NDVI + GARI	15.25
All Bands + NDVI + GLI	13.48
All Bands + NDVI + VDVI	13.48
All Bands + NDVI + GLI + VDVI	14.18
All Bands + NDVI	13.83
All Bands + VDVI	13.83
All Bands + GLI	13.83
All Bands + GLI + VDVI	15.25

[1] A set, random start was selected for model reduction in order to compare models using the same set of data

It was decided to use only NDVI for the remaining analyses given the negligible difference in error when adding GLI or VDVI and, NDVI is currently one of the most recognized and used indices (Aguilar et al. 2012). It also has broader applications than agricultural monitoring, which is the main use of the other two indices, both of which have also been found to produce nearly identical results (Biró et al. 2024).

The NDVI values for 2019 and 2022 are presented in Table 3.3, including a column showing the mean of both years and calculated after removing pixels classified as water and gravel. The lowest two-year average was found at CH1, followed by the water gap site at CH8. The CH2 and CH3 reaches produced the only 0.4 values, while all others, except CH1, were between 0.2-0.38. Correlations between NDVI and floodprone width were not especially strong. For CH2 and CH3, high NDVI numbers were associated with the two widest floodprone areas (142 and 200 respectively). Compared to CH1, which had the lowest NDVI value, these widths are six to eight times wider than the floodprone width at CH1. These results are encouraging, but discernable patterns fall apart when comparing floodprone widths and NDVI values for the other reaches. For example, CH13 and CH14 have far higher NDVI values than the CH1 average, but actually have even narrower floodprone widths. The opposite is true of the two beaver wetland sites (CH7 and CH10) where floodprone widths are wide, but NDVI numbers are fairly low (0.28 and 0.31 respectively). A moderate correlation was picked out based on willow production. Density and extent of willows are highest at CH11-CH14 and these sites also have the highest NDVI values of any of the incised reaches (0.35, 0.36, 0.37 and 0.38). Though less abundant, there also fairly well developed stands at CH6 (NDVI = 0.33) and CH10 (NDVI = 0.31).

Table 3.3. *NDVI Values Calculated from 2019 and 2022 NAIP Imagery*

Location	**2019 (Mean)**	**2019 (Minus Water)**	**2019 (Vegetation Only)**	**2022 (Mean)**	**2022 (Minus Water)**	**2022 (Vegetation Only)**	**2019/2022 Average (Vegetation Only)**
CH1	-0.107	-0.059	-0.057	0.117	0.118	0.157	0.05
CH2	0.349	0.416	0.432	0.336	0.354	0.367	0.40
CH3	0.423	0.44	0.441	0.349	0.348	0.37	0.41
CH4	0.109	0.165	0.272	0.209	0.202	0.24	0.26
CH5	0.136	0.168	0.278	0.227	0.231	0.264	0.27
CH6	0.24	0.302	0.35	0.271	0.288	0.301	0.33
CH7	0.295	0.318	0.318	0.207	0.208	0.241	0.28
CH8	0.118	0.145	0.177	0.182	0.181	0.221	0.20
CH9	0.154	0.171	0.19	0.221	0.221	0.26	0.23
CH10	0.26	0.284	0.296	0.304	0.306	0.323	0.31
CH11	0.271	0.321	0.383	0.346	0.346	0.364	0.37
CH12	0.319	0.352	0.382	0.361	0.361	0.383	0.38
CH13	0.292	0.327	0.348	0.346	0.346	0.377	0.36
CH14	0.225	0.296	0.342	0.317	0.317	0.348	0.35

Discussion

All of the performed imagery classification steps can be eliminated by using tools available from platforms such as the Google Climate Engine that brings in imagery from a variety of sources and has the ability to calculate numerous indices. The “re-inventing the wheel” approach used in this study was done for two reasons. The first was simply to gain a better understanding of the processes involved in remote sensing so as to better understand how it may be applied to riparian settings found at places like Maggie Creek. This knowledge was painfully accrued over several months of self-taught, trial and error and is by no means complete. There was also a belief that a higher level of precision could be achieved by using personal training data and incorporating a level of control by being able to focus solely upon a desired land classification, such as vegetation, when deriving a given index.

The precision of the final model in classifying pixels at incised reach locations was striking. It is not uncommon to have high error rates when attempting these types of projects, especially on small, narrow riparian corridors (Congalton et al. 2002; Gergel et al. 2007; Pu et al 2021). Accuracy in this study was due in part to the high-resolution imagery provided by NAIP and most certainly to the availability of ground-truthed data and great familiarity with the study area. However, it was still often necessary to delete problematic pixels, both to achieve low classification error and to produce a level plot graphic that corresponded with actual, on-the-ground conditions. This process was often repeated several time for a given reach. A common problem was water, coupled with narrow channels. The absorptive quality of water leads to low reflectance in the Red Edge and NIR bands, which demonstrate high reflectance in healthy vegetation (Kumar

& Reshmidevi 2013). This typically makes separating the two one of the easier remote sensing tasks. In this case, especially with the 2022 imagery, reflectance was confounded by a proliferation of aquatic plants. During the summer months, Maggie Creek discharge drops dramatically and much of the surface water becomes obscured by algal mats, *Azolla* sp. (water fern), and *Ranunculus aquatilis* (white water crowfoot). Water typically produces negative or very low positive NDVI values (Szabó et al. 2016). The interference of vegetation produced values similar, and sometimes higher, than those produced for actual vegetation. This problem was less evident in 2019 imagery, likely due to above average flows during the growing season, whereas flows were below average during the summer of 2022 (USGS 2024). This problem could therefore not be addressed by traditional thresholding techniques that mask pixels below, or above, a predetermined index value. This necessitated manually sorting and removing outlier water pixels from the spatial data frame created in R version 4.3.2. in order to force a degree of separation between water and vegetation. Narrow channels were also problematic as a combination of low summer flows and overhanging plants, especially in the more willow-lined reaches, made it difficult to place training points so that were not overlapping with vegetated pixels. Where distinguishing between water and vegetation was less problematic, a different problem often arose whereby shadows were often treated as water, especially in the areas of dense willow where shadows were more prevalent. This also occurred where patches of dark vegetation, like mountain rush, occurred in patches, such as in the meadow containing CH2 and CH3. With patience and perseverance, the above issues were alleviated, thus providing accurate calculations of NDVI, but it did prove to be time consuming.

The relatively weak correlation of floodprone width to average NDVI was disappointing, but not surprising. With the wide floodprone areas at the CH7 and CH10 beaver wetland sites, plant production is high and vigorous as a whole, but the NDVI signal was weak due to the nature of the herbaceous species composition. For much of the growing season, the high proportion of cattails and bulrush species have a spectral signature that produces lower NDVI values than willows, grasses and sedges. This suggests an additional step of screening the imagery to find suitable time periods when reflectance is higher in order to obtain more accurate results. With resolution of this issue, the correlation becomes stronger. Incorporating the STM, floodprone widths and NDVI values were much lower at the summer-grazed, “C_F” and “C” channels found at CH1 and CH8, as compared to the high values associated with the State 1, “E” and “E_C” channels at CH2 and CH3. With more precise valuations of NDVI at the wide floodprone areas of CH7 and CH10, these State 3, DA channels would likely be similar to meadow sites. This sets up a way to broadly distinguish between State 1 and State 3, as well as between less evolved and complex phases of State 3. With the 3-year interval of NAIP flights, it may prove a way to detect stressed vegetation over time as, in State 2, the channel starts disconnecting from its floodplain. Distinguishing between more intermediate phases in State 3 would be difficult, though the stabilizing “C” channels at CH4-CH6 due show a rough relationship of higher NDVI values as floodprone width increases. The “E” channel at CH9 has a moderately high floodprone width for the study area, but average NDVI was one of the lowest of all reaches. This is explainable, but only by virtue of knowing the recent history of this reach. The failure of a large beaver dam in the spring of 2019 left behind largely unvegetated sediments throughout much of the reach. The NDVI value was 0.19 for this year, which increased to 0.23 by 2022. Clearly this value will continue to increase as vegetation becomes more established. This illustrates the possibility of arriving at skewed conclusions if using imagery from a single growing season. Though an original intent of this study was to see if narrow floodprone widths would consistently track with low NDVI values, and visa versus, this fell apart when analyzing the four, uppermost reaches. As discussed in Chapter 2, the ability of willows to become established prior to channel incision and persist following incision, has created stable, willow-laden reaches, but has not allowed for channel widening and movement. Knowing that this scenario is possible along these types of streams, the combination of high NDVI values and narrow floodprone width, may serve as hallmarks of this condition when applying remote sensing to areas where on-the-ground knowledge is not available.

Conclusion

The results of this study were interesting and potentially some use in the larger scheme of riparian management and monitoring. However, it is clear that many issues remain when using remote sensing for detailed assessments of small, low-gradient streams. The amount of effort needed to counter these issues does not make for a rapid, translatable approach that can be readily adapted by users outside of an academic setting. There is also the question of how much is even gained by applying an index, like NDVI, to these systems, in the context of an STM, when much of the same information can be gleaned by simply studying high-resolution imagery. This is not a defeatist attitude, simply recognition that this superficial foray into remote sensing did not produce extravagant results. The potential of remote sensing to aid in the assessment of riparian condition, at both the reach and system levels, of small streams is much greater than what was explored here. None-the-less, there is some hope that the more detailed, index approach can help spot potential problem areas, such as an incised channel that is hidden by willow cover, through the identification of areas with high NDVI values, but narrow floodprone widths as seen at the upper end of the study area. High-resolution imagery and remote sensing indices may also provide useful insight for monitoring changes within riparian areas. As alluded to, the prior knowledge of the CH9 beaver dam, and eventual failure, provided insights as to why NDVI values were lower than expected. This information is not likely to be on hand in many cases, but unexpected results can at least flag areas worth conducting field investigations. NAIP imagery was not available for the time periods coinciding with the two, active restoration projects in the study area (see Chapter 2), but it is suspected that this product would be very useful for monitoring future projects and those taking place since the inception of NAIP.

It is also quite possible that the greater array of bands from satellite imagery can be used to calculate other indices more proficient at sorting out the complexities of these systems. An example is the Normalized Difference Infrared Index (NDII) which uses NIR and SWIR (short-wave infrared) to determine various metrics, including soil moisture content (Sriwongsitanon et al. 2015). Though largely uneventful, this exercise in remote sensing further demonstrated both its power and utility in assessing landscape conditions and the importance of bolstering results with ecological knowledge, area history and most importantly, direct, field-based corroboration.

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Appendices

Appendix A. Additional Figures and Tables

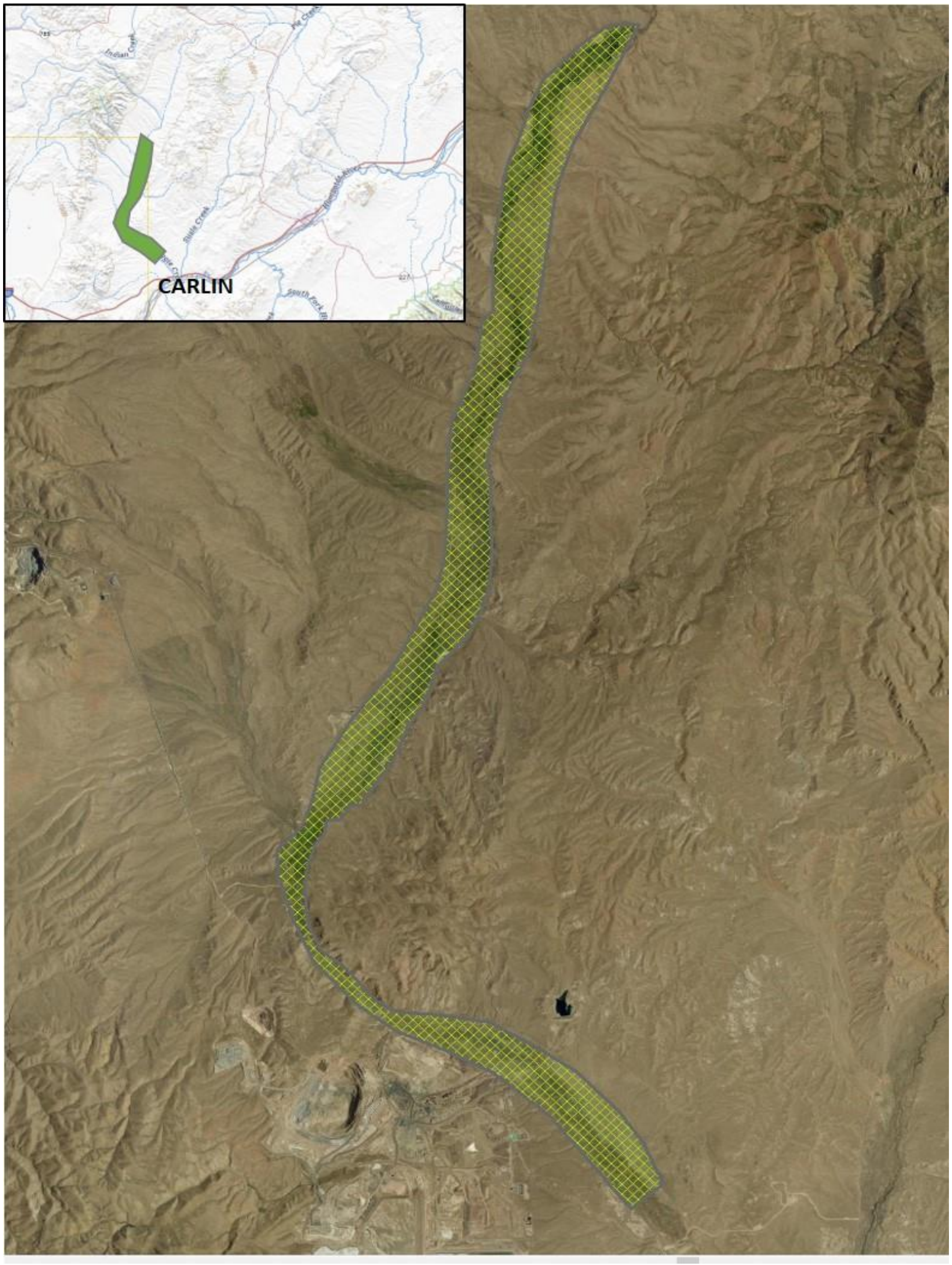


Figure A.1: *Overview of the Study Area*

Table A.1. *List and Description of Environmental Variables used for Multivariate Analysis*

Variable	Description
Entrenchment Ratio	The quotient of the floodprone width divided by the bankfull to bankfull width. Measurements were taken at each cross-section location.
Width:Depth Ratio	Ratio of the bankfull to bankfull width to average depth of the bankfull channel. Average depth was calculated by obtaining the area of the bankfull cross-section and dividing this value by bankfull width. Measurements were taken at each cross-section location.
Sinuosity	Distance along thalweg divided by straight line distance between end points of each longitudinal profile at sampled reaches.
Bankfull to Bankfull Width	Horizontal distance, perpendicular to the channel, from the bankfull location on one bank to the bankfull location on the opposite bank.
Bankfull Cross-section Area	Cumulative area derived from a series of horizontal and vertical measurements taken between each bankfull location along a given cross-section.
Maximum Depth	Deepest portion of the channel (thalweg) calculated by subtracting the bankfull rod height from the thalweg rod height. Measurements were taken at each cross-section location.
Floodprone Width	Determined in the field by multiplying Maximum Depth by 2. The survey rod was then set to this height. The first occurrence of this height (along both sides of the channel) were marked on the cross-section transect, with the difference between the two horizontal measurements denoting the floodprone width.
Bank Height Ratio	Obtained by dividing the lowest bank height by the maximum bankfull depth. Measurements were taken at each cross-section location.
D50 Sediment Size	Particle size in which 50 percent of the samples are larger and 50 percent are smaller based upon a cumulative distribution.
Greenline to Greenline Width	Average of 40 measurements, per sampled reach, capturing the perpendicular distance (in meters) between the greenline location on one bank and the greenline location on the opposite bank.
Wetland Rating	Weighted average based on wetland indicator status of species from greenline plant community data (Upland = 1, Facultative Upland = 25, Facultative = 50, Facultative Wetland = 75, Obligate = 100).
Adjusted Species Score	Value incorporating the wetland indicator status and Windward Stability Rating.

Table A.2. *List of all Encountered Bird Species from Point Count Stations*

Common Name	Species	Common Name	Species
American Black Duck	*Anas rubripes*	Lazuli Bunting	*Passerina amoena*
American Crow	*Corvus brachyrhynchos*	Lesser Goldfinch	*Spinus psaltria*
American Robin	*Turdus migratorius*	Mallard	*Anas platyrhynchos*
Barn Swallow	*Hirundo rustica*	Marsh Wren	*Cistothorus palustris*
Belted kingfisher	*Megaceryle alcyon*	Mountain Bluebird	*Sialia currocoides*
Bewick's Wren	*Thryomanes bewickii*	Mourning Dove	*Zenaida macroura*
Black-billed Magpie	*Pica hudsonia*	Northern Flicker	*Colaptes auratus*
Black-crowned Night-heron	*Nycticorax nycticorax*	Northern Harrier	*Circus hudsonius*
Black-headed Grosbeak	*Pheucticus melanocephalus*	Northern Rough-winged Swallow	*Stelgidopteryx serripennis*
Brewer's Blackbird	*Euphagus cyanochephalus*	Orange-crowned Warbler	*Leiothlypis celata*
Brewer’s Sparrow	*Spizella breweri*	Pied-billed Grebe	*Podilymbus podiceps*
Brown-headed Cowbird	*Molothrus ater*	Red-tailed Hawk	*Buteo jamaicensis*
Bullock's Oriole	*Icterus bullockii*	Red-winged Blackbird	*Agelaius phoeniceus*
Bushtit	*Psaltriparus minimus*	Sage Thrasher	*Oreoscoptes montanus*
Canada Goose	*Branta canadensis*	Sagebrush Sparrow	*Artemisiospiza nevadensis*
Chipping Sparrow	*Spizella passerina*	Sandhill Crane	*Antigone canadensis*
Cliff Swallow	*Petrochelidon pyrrhonota*	Savannah Sparrow	*Passerculus sandwichensis*
Common Nighthawk	*Chordeiles minor*	Say's Phoebe	*Sayornis saya*
Common Raven	*Corvus corax*	Sharp-shinned Hawk	*Accipiter striatus*
Common Yellowthroat	*Geothlypis trichas*	Snowy Egret	*Egretta thula*
Eastern Kingbird	*Tyrannus tyrannus*	Song Sparrow	*Melospiza melodia*
Forster's Tern	*Sterna forsteri*	Spotted Sandpiper	*Actitis macularius*
European starling	*Sturnus vulgaris*	Spotted Towhee	*Pipilo maculatus*
Golden Eagle	*Aquila chrysaetos*	Swainson's Hawk	*Buteo swainsoni*
Gray Flycatcher	*Empidonax wrightii*	Turkey Vulture	*Cathartes aura*
Gray Partridge	*Perdix perdix*	Vesper Sparrow	*Pooecetes gramineus*
Great Blue Heron	*Ardea herodias*	Western Kingbird	*Tyrannus verticalis*
Great Egret	*Ardea alba*	Western Meadowlark	*Sturnella neglecta*
Great-horned Owl	*Bubo virginianus*	Western Tanager	*Piranga ludoviciana*
Green-winged Teal	*Anas crecca*	Wilson's Snipe	*Gallinago delicata*
Horned Lark	*Eremophila alpestris*	Wilson's Warbler	*Cardellina pusilla*
House Finch	*Haemorhous mexicanus*	Yellow-rumped Warbler	*Setophaga coronata*
House Sparrow	*Passer domesticus*	Yellow Warbler	*Setophaga petechia*
Killdeer	*Charadrius vociferus*	Yellow-breasted Chat	*Icteria virens*
Lark Sparrow	*Chondestes grammacus*		

Appendix B. Photographs

Photograph B1. *Migrating Headcut in Meadow between CH2 and CH3.*

Photograph B2. *Dense Willow/Rose/Currant Plant Community*

Photograph B3. *Slump Blocks Colonized by Woolly Sedge*

Photograph B4. *Woolly Sedge Infilling Behind and Adjacent to Slump Blocks*

Photograph B5. *State 1: "E" Channel*

Photograph B6. *State 1: "C" Channel*

Photograph B7. *State 1: "D_A" Channel*

Photograph B8. *State 2: "G" Channel*


Photo Credit: Carol Evans

Photograph B9. *State 2: "F" Channel*

Photograph B10. *State 3: Analogue "C" Channel*

Photograph B11. *State 3: Analogue "CE" Channel*

Photograph B12. *State 3: Analogue "E" Channel*

Photograph B13. *State 3: Analogue "C$_F$" Channel*

Photograph B14. *State 3: Analogue "CE" Channel with Steep, Narrow, Heavily Vegetated Bank*

Photograph B15. *State 3: Analogue "D_A" Channel – Beaver Occupation (Emergent Wetland)*

Photograph B16. *State 3: Analogue "DA" Channel – Beaver Occupation (Scrub-shrub Wetland)*

Photograph B17. *State 3: Analogue "D_A" Channel – Beaver Occupation (Recently Abandoned)*

Photograph B18. *Study Area Conditions (1980)*

Photograph B19. *Study Area Conditions – "F" Channel (1980)*

Photograph B20. *Channel Straightening/Berm Construction*

***Photograph B21**. Incised "F" Channel Formation Following Channel Straightening*

Photograph B22. *Active, Plug and Pond Restoration of Straightened Channel Incision*

Photograph B23. *Plug and Pond Restoration in the Summer of 2014*

Photograph B24. *2022 Repair at Time of Construction, During 2023 Flooding, and Post-Flooding*

Photograph B25. *Representative Conditions at Susie Creek in 1978 and 2017*

Photograph B26. *Abandoned Beaver Wetland Succession – Meadow (Susie Creek)*

Photograph B27. *Willow Development (Susie Creek)*

Appendix C. Review of Multivariate Statistical Methodology

Multivariate Assessments

Multivariate analysis is a core aspect of the analytic side of community ecology. It is therefore worth spending a moment exploring some of more popular methods, including the considerations and controversies, inherent when selecting a given method, ranging all the way from data pre-processing to interpreting the results. This review will focus on ordinations and cluster analysis, both as they represent the most commonly used assessments in ecology and because they were the ones used during the multivariate analysis portions of this study.

Rare Species

An initial decision to be made prior to conducting analyses is how to treat rare species occurring within species count data. In most ecological applications, rare species are simply those with a minimal presence in the data set, regardless of whether the rarity is due to natural niche partitioning, small sample size, or environmental pressures (Cao & Larsen 2001). However, their retention in the data is often done to try and capture effects of environmental stress. For instance, commonly occurring species are usually adapted to a wider range of conditions, thus increasing the danger of missing important environmental thresholds where more scarce species fall out of the community, but common generalist species can still persist (Queheillalt et al. 2002; Poos & Jackson 2012).

Proponents of exclusion believe that rare species often contribute little to interpretation and increase overall noise in results (Marchant 2002; Poos & Jackson 2012). There is sufficient evidence of this occurring from papers coming out during the time period when multivariate analyses were first being developed and applied to community ecology data (e.g. Webb et al. 1967; Austin & Greig-Smith 1968; Day et al. 1971). There can also be issues with chi-squared distances, used in correspondence analysis (CA), such that distances between abundance values of common species are underweighted in comparison to values of rare species, prompting either adjustments to their values or removing them entirely (Legendre & Gallagher 2001). Conversely however, principal component analysis (PCA), which uses Euclidean distance, weighs abundant species more heavily. In general, there is agreement in contemporary times that PCA and CA are not always well-suited for community composition data (Cao & Larsen 2001; Legendre & Gallagher 2001). Additionally, older attempts at organizing and viewing this type of data tended to focus on large, broadscale groupings and used the results of reduced species matrices to compare against the original, full-species results, which negates the ability to determine if added, rare species are actually creating noise. These studies also seldom accounted for anthropogenic effects which can be subtle and hard to capture without the inclusion of rare species (Faith & Norris 1989; Cao & Larsen 2001). Ultimately, it is the nature of the study and questions being asked that should determine if rare species be included. There is likely little to be gained by their inclusion when assessing general patterns over a large landscapes or when obvious environmental gradients are exerting strong influences on species assemblages. However, with newer modes of multivariate analysis and methods of weighting rare species, or down-weighting abundant species, there is similarly little to lose by incorporating the full species data set.

With contemporary computing capabilities, it is also not especially onerous to run multiple scenarios and compare the results.

Data Transformations and Standardizations

Clustering and ordinations require a series of decisions to be made prior to running the analysis. The first concerns the data itself and whether to leave it in its raw form, standardize it and/or perform a transformation. This is an important step as it can profoundly affect the outcome, thus reinforcing the importance of considering the type of data being analyzed, appropriate applications of statistical methods, and purpose of the study (Kenkel 2006). When examining how sampling sites fall out based upon environmental factors, the associated dataframe or matrix is typically comprised of various, continuous measurements on different scales (e.g. meters, millimeters, unitless ratios), thus necessitating some form of standardization. A common approach is to convert values to “z-scores” in which the data is “centered” by subtracting a variable’s mean from an individual observation. A variable is then “scaled” by dividing the difference by the variable’s standard deviation. This effectively creates a new distribution with a mean of zero and standard deviation of one, thus increasing comparability, while leaving variable correlation intact and maintaining outliers (Urban et al. 2002; Rooney & Bayley 2010; Field et al. 2012).

Species data, which is typically measured the same way (e.g. abundance, frequency, percent cover), is more controversial, with arguments ranging from leaving it in its original form to “normalizing” the distribution through methods such as square root or log transformations. These discrepancies are most evident when dealing with count data, which is prone to numerous zeros and a skewed-right distribution due to a prevalence of species with low abundance coupled with a minority of species having especially high abundance (ter Braak & Šmilauer 2015). Once again, the choice is largely dictated by the desired information. Transformation of species data effectively equalizes species, such that differences between rare and abundant ones are minimized which allows small differences to be detected (Noy-Meir et al. 1975; Májeková et al. 2016). This is particularly useful for data when counts can span multiple orders of magnitude depending upon the species (Anderson 2008). In another sense, the need to transform data is dependent upon the statistical test. In parametric testing, count data routinely violates underlying assumptions (e.g. normal distribution and homoscedasticity), requiring that the data either be transformed, or nonparametric testing be applied (O’Hara & Kotze 2010). Regardless of parameters, there exist compelling arguments for transforming count data, especially if there is an interest in the community as a whole and not just commonly occurring species. For example, Clarke (1993) mentions that transformations allow for a greater contribution by rare species, while analyses using raw data tend to form patterns based primarily around those with a greater presence in the data set. Logarithmic transformations are typically used given the multiplicative nature of count data (ter Braak & Šmilauer 2015). Rank-abundance plots are another common method of transformation that take the proportional abundance of each species and ranks them in order of most to least common. This method is sometimes advocated for in preference to log transformations which tend, mathematically, to produce a “hump-shaped” distribution, even when untransformed data contain no such distribution (Nekola et al. 2008).

Ecological Resemblance/Similarity/Dissimilarity Measurements

How sites differ based upon beta diversity (difference between local and habitat-wide diversity) across time, treatments or other aspect of interest, is a central part of many ecological studies (De Cáceres et al. 2013). For decades, a standard statistical approach to assessing these differences in complex settings is to create a distance matrix which shows the dissimilarity between all possible pairs of sampling locations as related to species' attributes, such as composition or structure. Patterns within the matrix are then grouped in either ordination space or a cluster diagram (e.g. dendrograms in the case of hierarchical clustering) (Clarke et al. 2006; Zelený 2023). As a quick note, "similarity" is an index calculated based upon the number of species/factors shared by two sites, with a "1" representing complete agreement and a "0" indicating no overlap. For abundance data, a percentage similarity is typically used, as described further below in connection to the Bray-Curtis coefficient. Other indices typically used for binary data are Jaccard, Sørensen, and Simpson (Zelený 2023). The term "dissimilarity" is simply 1 minus a similarity value, and resemblance is a general term used to describe comparisons when neither of these two terms are implicitly stated (Lengyel & Botta-Dukát 2023). A further designation is made between "metric" and "non-metric" methods for calculating similarity/dissimilarity. In this connotation, metric refers to distance measures, such as Euclidean, that are always positive; have the same distance regardless of direction of comparison; the minimum distance is zero; and distances can be drawn in geometric space (Zelený 2023). Typical syntax in ecology denotes metric options as "distance" measurements and others, based off of scaled indices, as "dissimilarity" measures (Ricotta & Pavoine 2022).

Ways to calculate resemblance are legion (see Legendre & Legendre 2012). This leaves the analyst in the precarious position of selecting either a proper dissimilarity coefficient/distance measure or appropriate analysis in cases where a single method, such as Euclidean distance in PCA, is already baked into the process. This condition warrants a brief discussion on how various distances and indices work.

In terms of distance, one of the oldest and most straightforward measures is Euclidean distance, but this is problematic when dealing with numerical values, such as count data. This is due to the resolution of the equation which, in part, sums the squared differences between abundances of each species for a given pair of sites, quadrats, etc. Comparing two sites that do not share any species (double zero scenario), will yield a shorter distance (i.e. greater similarity) than two sites with the same species, but in which one has proportionally higher abundance values. This makes mathematical sense, but leaves something to be desired in terms of ecological interpretation (Kindt & Coe 2005; Ricotta & Podani 2017). This issue can be resolved to some extent by transforming the count data or using presence/absence in which all non-zeros take the value of 1 (Kindt & Coe 2005). In either case, the emphasis is shifted towards composition, rather than species abundance, which is usually of more interest to an investigator. The transformation route forms the basis of many subsequently developed distance measures such as chord distance, which calculates the Euclidean distance after each matrix column is standardized by dividing each value by the column total. Another popular method, Hellinger, figures the Euclidean distance after first taking the square root of relativized abundance values. For non-metric methods, the Bray-Curtis dissimilarity coefficient is most often used for species abundance data.

The Bray-Curtis coefficient handles instances of double zeros by dividing shared species abundance by the sum of the total abundance for each site. This is then subtracted from 1, which creates a percentage-based index where 1 indicates no concurrence of species and at the opposite end, 0 shows that all species and abundances are the same. This scaling effect has appeal in the ecological realm as maximum distance is applied to all pair-wise comparisons where no species are shared, irregardless of species abundance (Kindt & Coe 2005; Ricotta & Pavoine 2022). In addition, resemblance between two sites or samples is not affected by changes in scale (e.g. grams and milligrams in the case of biomass) or addition of more samples/sites (Clarke et al. 2006).

Numerical classifications

The most commonly used forms of numerical classification in ecology are hierarchical and K-means clustering which use an algorithm that seeks to sort data into groups based upon similarity between in-cluster members while simultaneously maximizing the distance between clusters (Pritchard & Anderson 1971; Zelený 2023). K-means clustering is a simple, non-hierarchical approach that partitions points that are most similar, with the mean value of each group (such as species composition or environmental factor) forming a group's center. The default algorithm, typically Hartigan-Wong which builds off of earlier approaches, seeks to minimize a point's Euclidean distance to its respective group center (Mohamad & Usman 2013). A partial drawback to this method is that it requires the user to pre-select the number of clusters to be tried in the analysis. However, there exist several options for cluster determination such as Broken Stick graphs, Akaike Information Criterion and Mantel tests (Ratcliff et al. 2018; Sinaga & Yang 2020).

In Hierarchical clustering, clusters are nested such that primary groups are further subdivided into smaller groups all sharing some characteristics of the parent. This can be done in a top-down (divisive) way that starts with the whole data set and breaks it up into smaller groups, or bottom-up (agglomerative) fashion, which treats individual samples/sites as a cluster initially and then assembles them into larger groups. The choice of distance/dissimilarity measure is used to first place the two samples/sites with the shortest distance together, with increasingly larger distances being considered, both between samples/sites and clusters, as the algorithm moves towards completion. The way in which clustering occurs is dependent upon the type of algorithm. Common ones include single linkage, complete linkage, and average linkage which assess cluster membership based upon distances between two members of different clusters, all members, or the average distance between all members, respectively. Ward's minimum distance is an algorithm often used for ecological applications that works by minimizing increases in the sum of squared distances as sites, sites and clusters, and clusters and clusters are joined together. As can be seen with methodology, this requires the use of metric distances, such as Euclidean, or the transformation of non-metric measures (e.g. the square root transformation of Bray-Curtis values) (Kindt & Coe 2005; Zelený 2023). The wider range of available choices when using hierarchical methods is beneficial in many ways, but does necessitate a greater degree of pre-analysis selection, thus once again requiring a thorough understanding of the methods, data, and analysis goals. As with k-means clustering, there are also ways to help determine the optimum number of clusters, but with either method the ultimate choice must be made by the investigator. For example, the dendrogram resulting from hierarchical clustering helps visualize differences among sites, but it is still necessary to decide if clusters can be further clumped together. This is typically done with a “pruning” method, available in most statistical software packages, that uses some user-determined criteria, such as height of branches to yield the final classification.

Ordination

Unlike classification, which parses data into groups based upon similarity and dissimilarity, ordinations, as the name suggests, seek to order information along gradients in dimensional space. They can be run as unconstrained, which assembles either a species or environmental matrix in isolation, with each species or environmental factor representing a separate dimension (Zelený 2023). The “job” of an ordination is to reduce the number of dimensions, which appear as axes when plotting the results, into a meaningful, hopefully, and interpretable array of patterns. The unconstrained approach is often used as an initial, exploratory examination of the data. For example, it may be noted with riparian data that plant species adapted to wet conditions tend to group closer together or that, with the environmental matrix, as entrenchment ratios increase, the flood prone width also increases, while the greenline to greenline width decreases. The variety pack of unconstrained ordination choices is fairly substantial. The classic approach of inputting raw data contains options for when a linear or monotonic relationship exists between species and environmental factors and for when the species response reaches an optimum, then declines (unimodal response). The first case is handled well by PCA. An example would be a location, such as a playa margin, where a salinity gradient exists with some species exhibiting a negative response as salinity increases and vice versa. PCA detects linear trends and arranges the original variables into composite variables along axes that best explain the variance, much like classical linear regression. The first axis is the combination of variables with the most explanatory power, with the subsequent axes explaining progressively less until, eventually enough axes are generated to explain all the variance (McCune et al 2002; Kenkel 2006; Richardson 2009). As variance will usually be maximally explained by the first few axes, these are considered the principle components that are driving differences in a given community (Abdi & Williams 2010). Further, the default PCA diagram only displays the first two axes, as even three axes engender a level of visual complexity that begins to undermine the purpose of the analysis in the first place (Palmer 2024). The percent variation explained by a given axis is termed its “eigenvalue” which is similar to the Pearson’s correlation coefficient in linear regression, though in this case applied to multiple dimensions and irrespective of dependent and independent variables (Kenkel 2006).

This type of analysis is done using Euclidean distance, thus the same issues with zero-prone data as discussed in Ecological Resemblance section, apply. Inputting a matrix of raw abundance data for instance often yields a horseshoe pattern in the resulting diagram. This is because the first axis passes through the centroid of all the species or environmental factors, thus explaining the majority of variance. The second axis must pass through this same midpoint, but at a right angle so as to capture a slice of the remaining variance. With Euclidean distance, to pairs with zero values are treated as being similar, and are therefore plotted close to each other in dimensional space, causing a distortion which typically lacks ecological significance (McCune et al. 2002; Zelený 2023; Palmer 2024). Because of this, PCA on raw community data is often not attempted, though it does work well for presence-absence data and when analyzing a matrix of continuous, environmental data. It is also possible to transform community data first and then run it through PCA, but this is not always the best solution, which leads into tests available for data with non-linear distributions. The first of these is CA, which is well suited to handle species having unimodal distributions.

The algorithm in this method projects into a theoretical ordination space rather than the geometrical species space associated with the direct measurements inherent in PCA. A species' representation in this space is based upon the point where it is most abundant (Palmer 2024). This is accomplished by converting the species matrix into scores, one set pertaining to rows and another to columns. Each score is based off of species abundance and is calculated by taking its row average, multiplying it by its column average and dividing by the matrix average, which gives the "expected" value. A residual, association between a row and column, is then calculated by subtracting the expected value from the original value. Each row (i.e. sampling unit) is then plotted along with each column (i.e. species) using the residuals as coordinates (Abdi & Béra 2017; Bock 2024). As with PCA, eigenvalues are generated which, in this case are a measure of how well rows and columns correspond with each other, rather than showing maximized linear variation (Kenkel 2006). The CA approach reduces the horseshoe effect experienced when analyzing data with unimodal species distributions using PCA. However, though often less pronounced, the nature of the unimodal distribution still results in an arch effect when a single environmental gradient is exerting a majority effect on species. Results are displayed using two axes as two-dimensional space works well for displaying the underlying Chi-square distances that underly CA (ter Braak & Šmilauer 2015). The second axis, though orthogonal, is not independent of the first axis, so with only one dominant gradient, the second axis becomes a distorted reflection of the first axis, leading to the formation of the arch (Kenkel 2006). This has been described as a quadratic mathematical artifact with no bearing on the actual structure of the data (Hill & Gauch 1980). Correspondence analysis has also been criticized for giving undue wait to rare species. This is a simplified way of summarizing the CA mode of operation which places rare species at the tails of the ordination diagram giving rise to a clumped mid-section with outliers. These outliers can have the same chi-squared distance as more abundant species, but appear graphically to be different (Hill & Gauch 1980; ter Braak 1985; Legendre and Gallagher 2001). Detrended correspondence analysis (DCA) was devised to address and resolve these issues. The main way it accomplishes arch removal is by dividing up the first axis into segments and centering the mean of scores along the second axis by each segment. Another way, given the quadratic nature of the arch, is to force the second axis to not only be linearly independent, but quadratically independent as well (Borcard et al. 2011). Outlier issues are handled by standardizing the within-sample unit variance (Hill & Gauch 1980). Though effective, the method of arch removal has since been viewed as a brutal way to force linearity into unimodal data and may further distort the results (Kenkel 2006; Zelený 2023). However, a happy coincidence of axis-segmenting is that axis lengths are reported in standard deviations which yields gradient length. The longer the axis length, the more heterogeneous the data. Circling back to deciding if a raw dataset is more suited for PCA (linear, short gradient) or CA (unimodal, long gradient), a rule of thumb is axis lengths of 4 or higher indicate unimodal distributions in the data and one should bypass PCA in favor of CA. Solid candidates for PCA should have an axis length of 3 or less. The mid-ground between 3 and 4 is usually deemed to be suitable for either analysis (Zelený 2023). A more conservative range is suggested in Legendre & Birks (2012), who advocate for PCA when the length is 2.5 or less and CA or data-transformed PCA for anything else.

As previously discussed, raw data can be transformed in various ways in an attempt to normalize distributions, increase linearity, etc. when using PCA and CA. Two other options involve either metric or non-metric scaling. In the first case, principle correspondence analysis (PCoA), sometimes called principle coordinate analysis, explains variation in observations using eigenvalues, the same as metric ordinations PCA and CA. It applies an additional step by computing a matrix of distances using any distance/similarity/dissimilarity metric, not just Euclidean. These distances are then used to create a third matrix of uncorrelated variables (principal coordinates) that are similar to the principle components of PCA. Though very similar to PCA, this ability to compute distances using measures that, depending upon the data, are more suitable makes it a good option when the goal is to explain as much variation as possible, while still projecting the ordination onto geometric or Euclidean space so that the space between objects represents the amount of similarity/dissimilarity (Legendre & Birks 2012; Buttigieg & Ramette 2014). By contrast, non-metric multidimensional scaling (NMDS) uses a similarity/dissimilarity matrix and plots objects based upon a ranking of distances, rather than direct measurements. In an actual-distance analogy, when plotting a matrix of geographical distances between towns on a map, the algorithm would find the longest distance between two points and assign it a rank of 1, followed by the next as rank 2 and so on. This is done on an iterative basis until, ideally, there is a convergence between the original distance matrix and the ranks. The degree of agreement is termed as “stress”, with 0 representing a perfect agreement. The literature generally agrees on the following guidelines: stress of < 0.05 indicates an excellent fit with little chance of misinterpretation; 0.05 - < 0.1 is a strong ordination, though any tight clusters may be worth exploring separately in another ordination; 0.1 - < 0.2 is a useable ordination, but one better suited for assessing general patterns; 0.2 < 0.3 should be used with much caution when the values are close to 0.2 and discarded as not bearing any meaningful representation of the data (Clarke 1993; Buttigieg & Ramette 2014). Some of the merits of NMDS are; 1) like PCoA, any distance/dissimilarity measure can be used; 2) it can handle a wide array of data sets, including those that violate assumptions of normality; 3) the user has the ability to set the number of dimensions prior to running the analysis; and 4) it can handle large data sets (Urban et al. 2002; Dexter et al. 2018).

Constrained, also known as direct or canonical, ordinations are adaptations of the previously discussed methods that allow for the comparison of a species matrix along with an environmental matrix. They include redundancy analysis (RDA) for PCA, transformation-based redundancy analysis (tb-RDA) for PCAs using pre-transformed data, canonical correspondence analysis (CCA) for CA, and distance-based redundancy analysis (db-RDA) for PCoA (Zelený 2023). For regular or transformation-based redundancy, a matrix each of species and explanatory variables are regressed forming a new matrix with the fitted results as well as the residuals. The fitted values, and the residuals if so desired, are then run through a PCA to produce a triplot showing species, sites, and environmental variables (Legendre & Gallagher 2001). The CCA takes a similar path whereby regressions are performed on the species and environmental data, though in this case a weighted average is taken such that abundance values are relative, rather than absolute (ter Braak & Verdonschot 1995; Makarenkov & Legendre 2002). Distance-based RDA is the application of RDA to the dissimilarity matrix calculated from PCoA (Legendre & Anderson 1999; Zelený 2023). This currently is the best option when a non-metric measure, such as Bray-Curtis, is desired as an analogous constrained method for NMDS has not been fully developed at the time of this writing.

In summary, PCA, CA, and PCoA are metric ordinations with PCA preserving Euclidean distances, CA keeping chi-square distances, and PCoA being used when it is desired to use some other measure of distance or similarity. For non-metric ordinations, the usual choice is NMDS which uses rank-order to plot points in the ordination space. All of these methods have complimentary forms that are able to incorporate explanatory variables in order to see what factors may be having an influence on community structure. There are no hard and fast rules for choosing one method over the other, but the data format and analysis objectives will largely dictate which is the most appropriate. In some cases, such as with abundance data with a lot of zeros, issues can be overcome by first transforming the data. In other instance, a large degree of noise and violation of normality assumptions pushes the investigator towards the more flexible NMDS ordination. Interpretation of ordinations can be tricky, but legitimate structural patterns are usually evident when selection criteria are understood. These methods remain powerful tools for visualizing and studying communities and how member interactions and environmental factors influence organization.

Appendix D. State and Transition Model Narrative

Ecological Dynamics

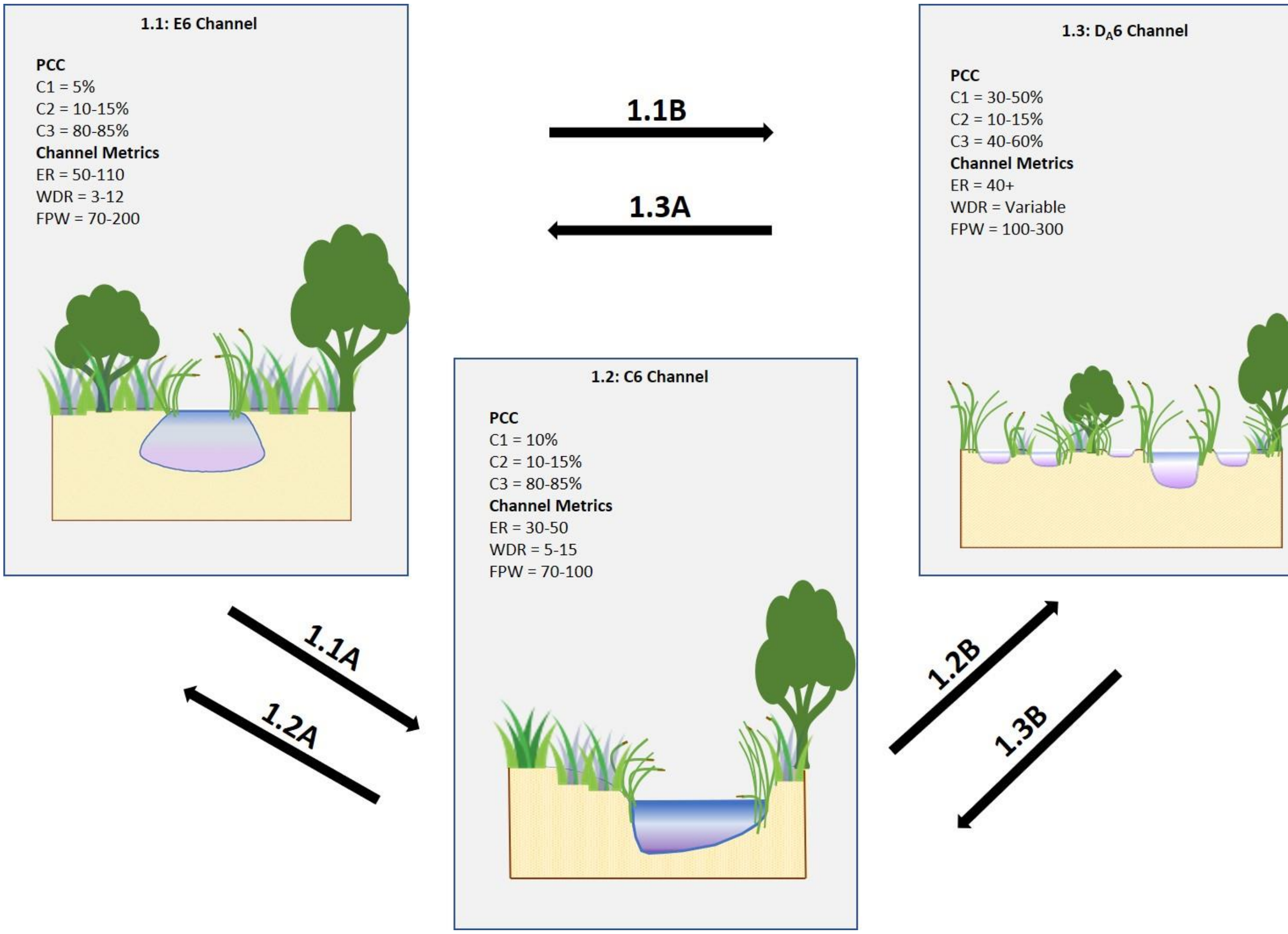
In its natural state the stream channel has a high Entrenchment Ratio (> 2.2), a low Width to Depth Ratio (< 12), has high sinuosity (≥ 1.5) and substrates are predominantly of the silt and clay size class. This is a low-gradient channel in a wide valley setting fitting with a Rosgen Valley Type VIII. Hillslopes are usually uncoupled from the channel allowing for the development of meander bends and channel migration. If beavers are not present in the system, this state is represented by a complex of Rosgen "E" and "C" channels. The "C" channel can occur naturally where valleys narrow somewhat and channels are often coupled to one side of the valley. This is also a common occurrence at confluence points where influxes of sediment can shift the channel to the opposite valley slope. Weakening of streamside vegetation through large flooding events, grazing, drought or some combination of these factors can also lead to "C" channel formation due to moderate increases in Width to Depth ratios and reductions in sinuosity. Allowing for recovery, these typically revert back to their former "E" channel morphology. When beavers are present, a "D_A" channel is typical, especially where large, wetland complexes have become established. The proliferation of channels is a result of beaver-created channels to access food and building material under cover of water and periodic avulsions and channel anastomosis forming as ponded water cuts around the sides of beaver dams. There is also an increase in open water and generally a higher degree of plant species diversity. The same holds with bird diversity as habitat variation accommodate water fowl, species preferring dense shrubs and those more apt to select herbaceous marsh environments.

With “E” and “C” type channels, plant communities at the water’s edge (PCC 1) are narrow, discontinuous and marked by species capable of sustained inundation and high-velocity flood waters (e.g. *Eleocharis* spp., *Schoenoplectus* spp., *Veronica americana*). Streambanks (PCC2) are typically anchored by dense stands of *Carex nebrascensis*, which is also a common component of the floodplain (PCC3). In systems that have experienced, or are experiencing, regular herbivory, this species will be dominant, otherwise it is often codominant with other sedge species and grasses, such as *Deschampsia cespitosa*. Pockets of willow species, especially *Salix lutea,* may also be present.

Consistent pressure coupled with other disturbances, such as high-flow flood events, cause the channel to widen as stabilizing vegetation is removed and eventual vertical erosion. This leads to the formation of “G” and “F” type channels which mark the transition from State 1 to the State 2 - Degraded Channel. This transition can be rapid as is usually followed by period of further downcutting and widening until a solid channel bottom is reached and widening is sufficient to dissipate flood velocity and create space for the development of a new, lower-elevation floodplain. This marks the transition into the State 3 - Stabilizing/Stable Analogue State. The initial phase is characterized by a “C_F” channel that bears similarities to the proceeding “F” channel with a shallow channel and high width to depth ratio. Reaches can transition from this phase to other phases and back again, but the general upward, linear trend is from a “C_F” channel to a “C” channel, then to a “C_E” channel and the eventual resumption of an “E” channel. These are considered analogue channels as they mimic conditions present in State 1, but are confined within high incision banks, thus reducing the original extent of the riparian corridor. Plant community components are variable, but key species, especially *Carex pellita*, are absent in the early phases of channel evolution and only become major parts of streambank composition along the "C_E" and "E" type channels. Restoration pathways are available, but limited. Active restoration, such as channel armoring, has the potential to shift State 2 back to State 1 if implemented early in the transition process. Once fully in State 2, more aggressive restoration, such as plug and pond, has been successfully carried out. Once in State 3, passive restoration, such as eliminating summer grazing and reducing overall use, is the suggested alternative with the aim of facilitating the process of stabilization. If beavers are not present in the watershed, it may be possible to reintroduce them once willow establishment has reached sustainable levels for dam construction, lodge building and food resources.

State and Transition Model

State 1. Potential Natural Channel



PCC	**Plant Association**
1	*Eleocharis* spp./*Veronica americana*/*Schoenoplectus* spp.
2	*Carex nebrascensis*
3	*Carex nebrascensis*/*Deschampsia cespitosa*/*Juncus arcticus* ssp. *littoralis*/mesic forbs

Phase 1.1

This phase has a narrow, deep, sinuous “E” channel with overhung banks secured by deep-rooted perennial sedges, typically *Carex nebrascensis*. The floodplain is wide and frequently inundated during flood events, especially in the spring coincident with snowmelt. The stream channel has a high Entrenchment Ratio (> 50), low Width to Depth Ratio (< 12), has high sinuosity (≥ 1.5) and substrates are predominantly of the silt and clay size class. The plant community present at the edge of the water (PCC 1) is narrow, discontinuous and marked by species capable of sustained inundation and high-velocity flood waters (e.g. *Eleocharis* spp., *Schoenoplectus* spp., *Veronica americana*). Streambanks (PCC2) are typically anchored by dense stands of *Carex nebrascensis*, which is also a common component of the floodplain (PCC3). In systems that have experienced, or are experiencing, regular herbivory, this species will be dominant, otherwise it is often codominant with other sedge species and grasses, such as *Deschampsia cespitosa*. Pockets of willow species, especially *Salix lutea*, may also be present.

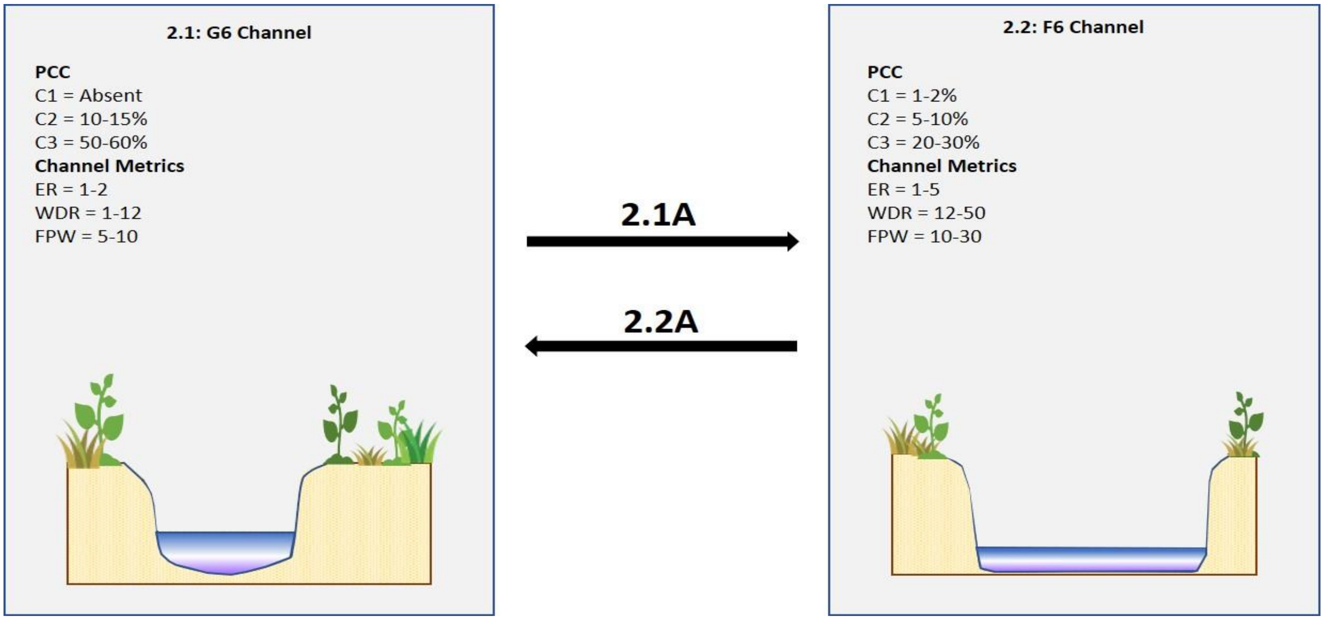
Phase 1.2

This phase is characterized by a “C” channel which is similar to the Phase 1.1 “E” channel, but is less deep and narrow and often has less meanders. It can represent a transition from either Phase 1.1 or 1.3, following moderate removal of stabilizing vegetation from disturbances such as herbivory (Phase 1.1 transition) or abandonment of a beaver wetland complex (Phase 1.3 transition). It can return to either state with a cessation or reduction of disturbance-related plant loss or subsequent beaver occupation. It may also occur naturally along reaches that are moderately confined due to topographic setting or large inputs of sediment from tributaries that effectively move the channel to one side of the valley. Plant community components are very similar to Phase 1.1, though proportions often change. The wider channel, reduction of overhanging bank and sediment accretion, especially where banks are trampled and broken down, allows for more substrate available for colonization in PCC1. Greater diversity in PCC2 is typical as the dominance of *Carex nebrascensis* and is lessened allowing for the inclusion of other graminoids and forb species.

Phase 1.3

This is the beaver occupied phase distinguished by a multi-thread channel “E” channel with high Entrenchment Ratios (40+) and variable Width to Depth Ratios. The Floodprone Width often extends further than that of Phase 1.1 and Phase 1.2 as water ponds and spreads behind dams and is conveyed throughout the wetland complex via beaver-created channels. Initial colonization is marked by large expanses of open water and a proliferation of herbaceous, emergent wetland supporting *Typha latifolia* and a variety of *Schoenoplectus*, *Carex* and *Juncus* species. If present, Salix exigua is often drowned at this point and supplanted by the obligate wetland species, *Salix lutea*. Expansion of both PCC1 and PCC2 is common as saturated conditions support establishment throughout the much of the riparian area. As the complex matures, willows tend to become more established and emergent vegetation is gradually replaced by more mid-to-late seral wetland graminoids, such as *Carex pellita*, *C. nebrascensis* and *C. utriculata*. Abandonment fosters the creation of Carex meadows in sediments deposited behind dams and infilling of willows becomes more prominent. This typically leads to a transition back to Phase 1.1.

State 2. Degraded Channel



PCC	**Plant Association**
1	*Eleocharis* spp./*Veronica americana*/*Schoenoplectus* spp.
2	*Carex nebrascensis*
3	*Carex nebrascensis*/*Deschampsia cespitosa*/*Juncus arcticus* ssp. *littoralis*/mesic forbs

Phase 2.1

Transition from State 1 to State 2 occurs when sufficient stabilizing vegetation has been removed from channel bank allowing for bank erosion and eventual channel incision sufficient to severe the channel from its floodplain. Vertical erosion usually occurs when the damaged banks are exposed to additional disturbance, such as flood events, and headcuts are common features. Removal of disturbance factors at this point will not return State 2 to State 1, though active restoration, such as headcut repair and channel armoring, will at least slow or halt further degradation. If installed early in the transitional process, repairs may lead to a reversion to State 1 over time. More direct and aggressive restoration is also possible, such as plug and pond, in order to achieve State 1 in a more expedient, though more expensive, fashion. Phase 2.1 is denoted by the formation of "G" channel as the bed elevation drops. Once enough depth has been reached, herbaceous streambank vegetation (PCC2) loses contact with the water table and dries out to be replaced by species adapted to drier conditions (e.g. *Poa pratensis*, *Leymus triticoides*, weedy forbs). Mortality of stabilizing species often results in additional bank sloughing as roots dieback, decreasing stability. Banks lose concavity becoming vertical and stream velocity increases. This usually results in the loss of PCC1.

Phase 2.2

This phase occurs as the “G” channel of Phase 2.1 widens out following bank collapse. The Entrenchment ratio increases slightly as does the Width to Depth ratio. At this point, there is still no development of a new floodplain and PCC1 is lacking while PCC2 and PCC3 become further reduced in size. If the channel bed is firm enough to prevent further downcutting, this phase will mark the transition into State 3 as channel widening and stream flow velocity reductions are sufficient to support the formation of a new floodplain. However, as the State 1 meadow systems are typically composed of deep layers of fine material, the channel will usually go through a series of “G” to “F” conversions before eventually ceasing as the channel cuts through the entirety of fine materials and reaches buried, large coarse material typical of alluvial valleys.

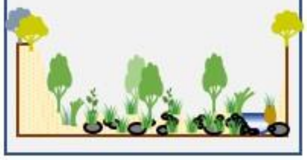

Photo Credit: Carol Evans

State 3. Stabilizing/Stable Analogue Channel

3.1: C_F Channel
PCC
C1 = 5%
C2 = 15-20%
C3 = 10-15%
C4 = 25%
Channel Metrics
ER = < 2
WDR = 50+
FPW = 20-30

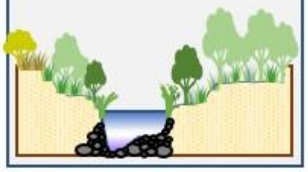
3.2: C Channel (Initial Stabilization Phase)
PCC
C1 = 5%
C2 = 15-20%
C3 = 15-20%
C4 = 25%
Channel Metrics
ER = 2-10
WDR = 15-50
FPW = 30-60

3.3: C Channel (Mid Stabilization Phase)
PCC
C1 = 5%
C2 = 15-20%
C3 = 25-30%
C4 = 20%
Channel Metrics
ER = 2-40
WDR = 15-30
FPW = 40-70

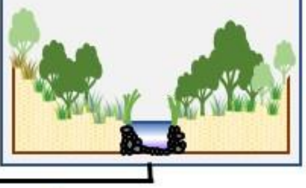


3.4: C Channel (Late Stabilization Phase)
PCC
C1 = 5%
C2 = 20-25 %
C3 = 30-50%
C4 = 20%
Channel Metrics
ER = 10-50
WDR = 5-15
FPW = 50-80

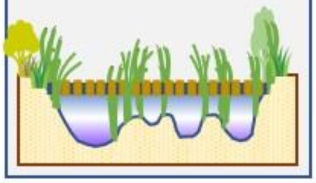
3.5: C_E Channel
PCC
C1 = 5%
C2 = 25-30%
C3 = 40-60%
C4 = 20%
Channel Metrics
ER = 20-50
WDR = 5-15
FPW = 50-100



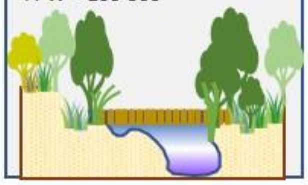
3.6: E Channel
PCC
C1 = 5%
C2 =30-35 %
C3 = 50-70%
C4 = 15%
Channel Metrics
ER = 50-100
WDR = 3-12
FPW = 60-120



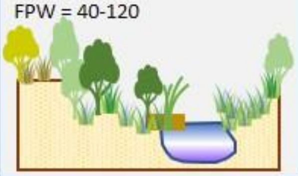
3.7a: D_A Channel (New Beaver Dam Complex)
PCC
C1 = 30-50%
C2 = 10-15%
C3 = 40-60%
C4 = 15%
Channel Metrics
ER = 40+
WDR = Variable
FPW = 100-300



3.7b: D_A Channel (Old Beaver Dam Complex)
PCC
C1 = 20-25%
C2 = 20-30%
C3 = 50-70%
C4 = 5%
Channel Metrics
ER = 40+
WDR = Variable
FPW = 100-300



3.7c: C/E Channel (Abandoned Beaver Dam Complex)
PCC
C1 = 10%
C2 = 15-35%
C3 = 25-70%
C4 = 10%
Channel Metrics
ER = 10-50
WDR = 3-30
FPW = 40-120



PCC	**Plant Association**
1	*Eleocharis* spp./*Veronica americana*/*Schoenoplectus* spp.
2	*Carex pellita/Agrostis stolonifera/Salix exigua/Salix lutea/*mesic forbs
3	*Salix exigua/Salix lutea/Leymus triticoides/Carex praegracilis/Juncus arcticus ssp. littoralis/Poa pratensis*
4	*Artemisia tridentata ssp. tridentata/Ericameria nauseosa/Leymus triticoides/Carex praegracilis/*mesic forbs

Phases 3.1-3.6

The transition to State 3 happens when the channel downcutting of State 2 has ceased and the channel becomes wide enough to foster the development of a new floodplain within the incision banks. Active restoration in this state should be restricted to work within the new floodplain area, rather than attempts to return it to State 1, which is not feasible in terms of financial and logistical factors. Passive restoration, such as changes in grazing timing, duration, and intensity, or beaver colonization, are effective ways to facilitate stability and functionality of the now reduced riparian corridor. The first six phases depict channel evolution from a stabilizing “C_F” channel, which forms once a new floodplain is developed and riparian vegetation starts becoming reestablished, up to an “E” channel similar to that of State 1. All phases within this state are considered analogue as they are similar to the natural channel, but occur within the old incision banks. The “C” channels tend to be mobile, shifting around as sediments and flood events pass through the reaches. This becomes less pronounced in the later stabilization phases (3.3-3.4) and PCC2 picks up *Carex pellita* which is absent in early phases. It is possible for Phase 3.5 (C_E channel) to become a near steady state in certain conditions. Where willows persisted following the deterioration that led to State 3, such as around irrigated hay meadows, they may stabilize the banks before the channel becomes very wide, thus locking reaches into a stable channel, but one that no longer has the ability to achieve a wide floodplain. Other characteristics, such as Entrenchment ratio and Width to Depth ratio can still improve to a point,

hence placing this state further along the evolutionary trajectory than the “C” channels.

Phases 3.7 a-c

These phases represent different iterations of beaver-affected reaches. A similar pattern is observed as with State 1 where initial colonization is marked by predominantly emergent, herbaceous wetland with many areas of open water and a multitude of small channels. Similar processes occur as the wetland complex mature and are eventually abandoned due to lack of resources or multiple, flood-related dam breaches. In this latter case, a secondary round of succession takes place as the exposed sediments, formerly underwater, are initially bare, then become covered in early-seral, ruderal species and eventually become lush *Carex* meadows intermingled among stands of willows. Unique to State 3, channel aggradation behind beaver impoundments offers some potential to reconnect the channel to the old terrace that was part of the State 1 floodplain. This is marked initially by the movement of wetland species up the terrace slope to eventually start establishing on the terrace. There is concern that a return of floodwaters to the terrace may result in lateral headcuts as water pours into the channel below the beaver dam complex. This is a likely scenario if the terrace is comprised primarily of shrubland with a depauperate understory. This event has not been recorded, thus documented consequences are not available to reference. Should channel aggradation be enough to make this a possibility, it is recommended to implement terrace treatments, such as brush thinning and seeding, that will provide a measure of stability.