



Research article

Ecosystem carbon and nitrogen gains following 27 years of grazing management in a semiarid alluvial valley

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ARTICLE INFO

Keywords:

Process based restoration
Rangeland
Meadow
Beaver
Groundwater dependent
Maggie Creek
Wetland
Great Basin

ABSTRACT

Soils in semiarid riparian ecosystems have large carbon (C) stocks that promote water and nutrient availability for productive plant communities consumed by grazing animals. Changes to riparian hydrologic conditions caused by channel incision result in different edaphic conditions and a greater abundance of upland plant species that may be associated with lower soil C stocks. Using riparian meadows alongside Maggie Creek in central Nevada, we show that 27 years of modified grazing practices can repair ecosystem processes and increase the C stocks. We compared C and nitrogen (N) stocks (of soils and plant biomass) on floodplains, terraces, and uplands of reaches where grazing was either modified or excluded to reaches where no changes to grazing practices were made. Grazing management allowed beaver to establish, improving hydrology and lengthening the growing season. These changes allowed C and N to accumulate on geomorphic surfaces that extended from the stream channel to the surrounding hillslopes. A stoichiometric relationship between C and N shows carbon sequestration can reduce nutrient runoff to nearby waterways and may depend on nitrogen availability. Gains in ecosystem carbon ranged from 93 to 452 g C m⁻² y⁻¹ and were dominated by increases in soil C. Gains in soil C occurred across the full depth range measured (0–45 cm) and were comparable to those found in restored wetlands and meadows located in more humid ecosystems. Carbon gains exhibited substantial variability caused by microtopography and plant community composition. While grazing exclusion resulted in the largest gains in ecosystem C, managed grazing that limited consumption of riparian plants increased ecosystem C relative to reaches where management wasn't changed. We demonstrate that managed grazing that maintains ecosystem process is compatible with projects aimed at increasing soil carbon in semiarid riparian rangelands.

1. Introduction

Adoption of management practices that increase carbon (C) storage in rangeland and cropland has the potential to sequester an additional 4–5 Gt of atmospheric carbon dioxide (CO₂) per year in soil organic matter (Paustian et al., 2019). In addition to climate benefits, C sequestration has the potential to generate revenue in developing voluntary C markets (Kreibich and Hermwille, 2021) and provide ecosystem services such as water storage and infiltration (Ankenbauer and Loheide, 2017; Apfelbaum et al., 2022; Dominati et al., 2010). Soil organic matter also contains substantial nitrogen (N), supporting plant productivity and reducing pollution of aquatic ecosystems. Consequently, soil C sequestration has the potential to increase soil N, benefitting terrestrial productivity and water quality. Soil C may be

especially important in semiarid rangelands where plant productivity can be limited by water and nitrogen availability (Burke et al., 1997; Van Groenigen et al., 2017).

Soil C sequestration through management or restoration may be most impactful in ecosystems where potential C storage is greater than the current C stock (Paustian et al., 2019). One such location may include riparian and low-gradient meadow valleys in montane regions. In riparian meadows, water is supplied to floodplains adjacent to stream channels (Weixelman et al., 2011). A shallow water table during the growing season supports herbaceous plant communities that form dense root mats which allocate substantial C belowground (Reed et al., 2021). The condition of riparian meadows can be altered by disturbance to vegetation and soils which initiates a feedback loop resulting in channel incision (Trimble and Mendel, 1995), increased depth to groundwater,

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and loss of wetland meadow vegetation responsible for streambank stabilization (Darrouzet-Nardi et al., 2006). Meadow condition may alter C or N stocks and fluxes. For example, in the Sierra Nevada of California, USA, soil C and N stocks were lower in non-hydrologically functioning meadow conditions than in properly functioning meadows (Norton et al., 2011). Meadows of the Sierra Nevada in California, USA that were net carbon sinks had a shallower depth to groundwater, lower soil bulk density, and greater aboveground plant and root biomass than meadows that were net carbon sources to the atmosphere (Reed et al., 2021).

Management that restores riparian plant communities may be able to arrest and reverse the disturbance feedback by reducing channel erosion, increasing channel roughness, and capturing sediment to reverse incision (Beechie et al., 2010). Twenty years of meadow hydrologic restoration in the Sierra Nevada resulted in average sequestration rates of 3 g C kg soil⁻¹ y⁻¹ and 0.18 g N kg soil⁻¹ y⁻¹ (Reed et al., 2022). Soil C to N ratios did not change following restoration, indicating that soil C sequestration may occur at roughly the same rate as soil N sequestration. The surface soil was most responsive to C and N sequestration following restoration (Reed et al., 2022), probably because root turnover and exudation contribute the most C inputs to surface soil (Reed et al., 2021). In some cases, beaver (*Castor canadensis*) contribute to the restoration of channels and associated terrestrial vegetation by engineering dams that impound water and capture sediment to further reverse incision (Fairfax and Small, 2018; Nash et al., 2021; Nummi et al., 2018). Hydrologic impoundments created by beaver can substantially increase the extent of wet riparian meadows (Hood and Larson, 2015). Beaver have been shown to impact ecosystem C and N. Riparian areas containing beaver ponds hold 8–23% of the total ecosystem C in Rocky Mountain National Park, USA (Wohl, 2013).

Meadow restoration in semiarid ecosystems may represent an opportunity for C and N sequestration. Per unit area, semiarid meadows hold nearly three times as much C as their surrounding uplands, making their contribution to regional C stocks relatively greater than meadows in montane regions that receive more precipitation (Morra, 2022). Additionally, semiarid meadows provide sources of forage, habitat, and late season water (Chambers et al., 2011). These resources often make meadows a focal point of grazing activities, which can contribute to the loss of meadow vegetation under heavy use (Bardgett et al., 2021; Trimble and Mendel, 1995). Similar to hydrologic restoration and beaver activity, modification of grazing can restore bankside vegetation (Green and Kauffman, 1995). In several semiarid watersheds in Nevada, USA, the regrowth of bankside vegetation following grazing modification has improved beaver habitat, supporting construction of beaver dams (Charnley, 2019). Grazing modification, subsequent bankside stabilization by plants, and development of beaver impoundments resulted in greater plant productivity and evapotranspiration in vegetation extending from the stream channel to the surrounding hill slopes of alluvial valleys in Nevada (Fairfax and Small, 2018).

We sought to quantify potential C and N sequestration following livestock grazing management and beaver activity in semiarid riparian ecosystems. We measured C and N stocks in beaver ponds and on floodplains, terraces, and uplands following 27 years of grazing management and in unmanaged baseline locations within the watershed to address the following questions.

1. How much do ecosystem C and N stocks in a riparian floodplain change in response to 27 years of grazing management and beaver activity in a semiarid region such as the Great Basin, USA? Using post-hoc tests we also estimate how many years would have been required after management actions for significant C sequestration to take place, where significant C gains occurred.
2. How does the impact of livestock grazing management and beaver activity on C and N stocks vary laterally from the stream channel, vertically within the soil profile, and which surfaces have the

greatest potential for C sequestration following spatial extrapolation of changes in ecosystem C?

- a. We expected the largest C and N gains would occur in shallow soils adjacent to stream channels where impoundments of water by beaver have increased water availability to vegetation.
- b. We expected that gains in N would occur commensurately with gains in C.

2. Methods

2.1. Site description

We worked in the Maggie Creek watershed, which is a tributary of the Humboldt River, located in the Tuscarora Mountains in northeast Nevada, USA. The 30-year average annual precipitation at Maggie Creek is 291 mm, which occurs mainly as winter snowfall; streamflow in Maggie Creek peaks after snowmelt in the spring (Kozlowski et al., 2016). The Maggie Creek watershed covers 102,850 ha and has been used for livestock grazing since the late 1800s (Horton, 2000). The United States Bureau of Land Management (BLM) manages 42% of the land, while 55% is privately owned. The remaining 3% are owned by the State of Nevada (Kozlowski et al., 2016). Nevada Gold (Formerly the Newmont Mining Corporation) operates an open pit mine near Maggie Creek. After increasing the pit depth in 1994, the mine began dewatering the surrounding riparian area. These actions required environmental mitigation of the reaches of Maggie Creek impacted by dewatering.

As part of this mitigation effort, the Maggie Creek Watershed Restoration Project (MCWRP) was implemented in 1994 by a partnership between the TS Ranch, owned by the Elko Land & Livestock Company (a subsidiary of Newmont), and the Elko (Nevada) BLM district. The goal of the MCWRP was to improve streams, riparian habitats, and watershed conditions in historically grazed riparian corridors. This was achieved by employing new grazing management or excluding grazing completely. Following a four-year rest period (1994–1998), grazing management consisted of short periods of use in the springtime (often a month or less) and fall use by cow-calf pairs. Annual grazing records from 1998 to 2006 can be found in (Evans, 2009) and are representative of the current use by livestock. These managed pastures have been under active, flexible management focused on riparian outcomes. Each year, between 60 and 353 animal unit months use this 97 ha pasture 1.61 to 0.27 ha/AUM (Evans, 2009). Stocking rates are determined by biological criteria (see Evans, 2009) designed to measure the quality of Lahontan cutthroat trout habitat. The consumption of riparian forage was monitored using exclusion cages and indicates less than 20% of the annual growth was consumed by livestock (Jon Griggs, personal communication). In the grazing exclusion pasture, fences were installed and consistent use by cattle ended in 1994. Implementation of the MCWRP allowed for the recovery of herbaceous vegetation and the growth of riparian shrubs, such as willows (genus *Salix*), reaching sizes that exceeded livestock browsing heights (ORC, 2017). These changes are an important part of beaver habitat and can encourage the natural colonization of beaver (Charnley, 2019). Beavers, and the dams they constructed, further enhanced the restoration process, expanding the riparian area by as much as 250% in some stream reaches (ORC, 2017) (Fig. 1), though beaver did not successfully establish in all reaches. Changes to the timing and intensity of grazing alongside benefits associated with beaver increased the length of continuous stream from 38 km to 46 km between 2006 and 2010 (Jensen, 2011). The recovery of stream channel function and the expansion of hydric vegetation along Maggie Creek has been reported through surface measurements by Kozlowski et al. (2016), and remotely sensed imagery contracted by Newmont (JBR, 2002; Jensen, 2011; ORC, 2017).

To measure the impact of changes in management on ecosystem C and N stocks, we made comparisons among three reaches within the MCWRP and a fourth reach on Maggie Creek where hot season use by



Fig. 1. Photo point comparison of three floodplains in Maggie Creek. Photo A shows the condition of floodplains in 1980 before grazing exclusion. Photo B shows the same floodplain after 27 years of grazing exclusion. Photo C shows the present condition of floodplains still experiencing summer season grazing. Historical photo provided by Carol Evans, United States Bureau of Land Management. Present-day photos in panel B and C were taken during a site visit by B. Morra.

cattle continues (Fig. 1C). In this fourth reach, no management is used to keep livestock away from the stream channel and stocking rates do not fluctuate with the stream channel condition. Stocking rates in this pasture were approximately 879 AUMs across a 190 ha pasture, equaling 0.22 ha/AUM during August and September (David Voth personal communication, 2022). Therefore, we refer to this region as “unmanaged grazing” throughout the text, though we acknowledge that most livestock grazing is, to some extent, managed. In two of the regions where grazing was excluded, two distinct reaches of beaver ponds formed. One reach was dominated by ephemeral beaver ponds which were smaller in size (30–40 m²) and failed during years of high precipitation, which we call “grazing excluded ephemeral ponds.” The second reach was dominated by permanent beaver ponds, and created a continuous impounded reach (~1.5 ha) (Table 1), which we call “grazing exclude permanent ponds.” Pollock et al. (2014) observed similar grazing patterns in beaver-built structures and found permanence required wide stream channels to dissipate stream energy. In the third reach, grazing was managed, but not excluded, and was associated with formation of ephemeral beaver ponds, which we call “managed grazing ephemeral ponds.” The two grazing excluded reaches were located in the same pasture, roughly 2 km apart while the managed grazing ephemeral pond reach was located in an adjacent pasture downstream. The unmanaged grazing reach, located outside of the MCWRP, was located 10 km downstream from the managed grazing reach. A map showing land ownership and grazing allotments can be found in

(Kozłowski et al., 2016). We used historical imagery and work by Evans (2009) to determine the unmanaged grazing reach was similar to the reaches within the MCWRP prior to grazing regime changes (Fig. 1).

To understand how changes in management affect surfaces extending laterally from the stream channel, we identified three geomorphic surfaces: floodplains that are actively being impacted by the surface flow of Maggie Creek, terraces that were previously influenced by surface flow, and upland areas that are outside of the influence of the stream channel, for each of the reaches. While uplands were likely not impacted by the stream channel, they were within the fences installed by the MCWRP and experienced the impact of changes to grazing management. When present, we also sampled accumulated sediment within beaver ponds. We were able to map the extent of all geomorphic surfaces within the reaches using satellite imagery (Google Earth, 2022). The extent of floodplains, terraces, and upland areas were ground-truthed at the time of data collection in the field.

2.2. Soil and pond sediment sampling

We sampled to a depth of 45 cm at six locations on floodplains and terraces and three locations in the upland soils in the three reaches of the MCWRP (Supplementary materials Fig. 2A, 2B, 2C). We sampled to this depth based on previous observations of soil C depth distributions in California, showing minimal C change in meadow soils deeper than 50 cm (Reed et al., 2021). We chose to concentrate our sampling in the

Table 1
Areas of geomorphic surfaces contained in a 1-km stretch of valley for each reach. Soil descriptions were collected from USDA, NRCS Web Soil Survey. Dominant plant species is the herbaceous plant species with the highest mean percent cover measured at coring locations.

Stream reach	Latitude, Longitude	Upland area (ha km ⁻¹)	Terrace area (ha km ⁻¹)	Floodplain area (ha km ⁻¹)	Pond area (ha km ⁻¹)	Upland soil type	Terrace soil type	Floodplain soil type	Dominant upland species	Dominant terrace species	Dominant floodplain species
Grazing exclusion ephemeral ponds	40.91°, -116.17°	42.3	44.2	6.2	0.2	Xeric Haplocambids	Aquandic Endoaquolls	Alluvial Land	<i>Elymus elymoides</i> , <i>Artemisia tridentata</i>	<i>Leymus triticoides</i> , <i>Artemisia tridentata</i>	<i>Carex pellita</i>
Grazing exclusion permanent ponds	40.89°, -116.184°	30.2	17.5	8.3	2.8	Xeric Haplocambids	Aquandic Endoaquolls	Alluvial Land	<i>Achnatherum speciosum</i> , <i>Artemisia tridentata</i>	<i>Muhlenbergia richardsonis</i> , <i>Artemisia tridentata</i>	<i>Carex pellita</i>
Managed grazing	40.86°, -116.21°	66.3	24.5	4.8	0.04	Xeric Haplocambids	Aquandic Endoaquolls	Vertic	<i>Distichlis spicata</i> , <i>Artemisia tridentata</i>	<i>Leymus triticoides</i> , <i>Artemisia tridentata</i>	<i>Symphoricarum ascendens</i>
Year-round grazing	40.77°, -116.134°	45.7	27.3	2.3	0	Xeric Haplocambids	Aquandic Endoaquolls	Halaquepts Aquandic Endoaquolls	<i>Elymus elymoides</i> , <i>Artemisia tridentata</i>	<i>Iva axillaris</i> , <i>Artemisia tridentata</i>	<i>Conyza canadensis</i>

riparian corridor (floodplains and terraces) as management of these areas were the primary focus of the MCWRP. Additionally, cattle tend to concentrate in these areas which have accessible water and palatable forage. We expected upland areas would contain smaller C and N stocks and were less likely to experience change due to grazing management than the riparian corridor and sampled these areas at a lower spatial intensity. In unmanaged reaches outside of the MCWRP, soils and plant communities showed little difference among geomorphic surfaces (Supplementary materials Fig. 2D). Therefore, we collected three samples on each geomorphic surface due to the deep channel incision that minimized the influence of the stream channel on geomorphic surfaces. Minimal channel influence was observed as spatial homogeneity of soils and plants in the unmanaged reach. All soil samples were collected in 15-cm increments using a five cm-diameter AMS slide-hammer (AMS Equipment Corp, American Falls, Idaho, USA).

To measure C and N stocks in accumulated sediment within ponds associated with managed grazing and the two excluded grazing reaches, we first estimated sediment volume within the ponds. To estimate volume, we measured the pond surface area using transect tapes, crossing the pond with a stand-up paddleboard when necessary. We then measured the depth of the sediment by repeatedly inserting a wooden rod to the sediment surface and then to the rocky substrate below the sediment in transects across the pond (perpendicular to the stream axis). We then sampled sediment for bulk density and C/N concentration by inserting a 5-cm diameter polyvinyl chloride (PVC) tube into the sediment. To retrieve the sample, we filled the remaining airspace in the tube with water and sealed the top using a 5-cm gripper plug (Oatey, Cleveland Ohio) to ensure the sample didn't slip back out of the PVC tube. We applied average depths of 20.6, 31.2, and 43.6 cm for ponds in the managed grazing ephemeral pond reach, the grazing exclusion ephemeral pond reach, and the grazing exclusion permanent pond reach respectively. The unmanaged reach outside of the MCWRP had no beaver activity. In reaches with ephemeral beaver dams, we multiplied the mean ponds size by the number of dams observed in satellite imagery (Google Earth, 2022). The total impounded area in the grazing excluded permanent pond reach was estimated by drawing a polygon around the end points of the three transects used to estimate sediment depth.

We calculated soil C and N stocks (kg m⁻²) as the product of C or N concentration in each sample (both soils and pond sediments), its bulk density, and the sampling depth. Bulk density was the rock-free, root-free mass of the soil dried at 105 °C. Roots and rocks were removed from soils by first passing them through a 2 mm sieve to capture the rocks. Roots were separated from the rocks by floatation and the supernatant was strained using a 0.5 mm sieve, air dried at 60 °C and weighed. We tested soils for the presence of inorganic C by adding several drops of 1 M HCl which generates bubbles in the presence of carbonate minerals. Subsamples (~2 g) of rock free and root free soils that contained carbonate minerals were treated with 5 mL of 0.1 M HCl. After acidification, samples were shaken vigorously and spun in a centrifuge at 3000 rpm for 5 min. We tested the pH of the supernatant and added more HCl and repeated the shaking and centrifuging if the pH was above 6.3, assuming all the acid had been consumed. Once carbonates had been removed, the HCl supernatant was poured off and samples were rinsed 3 times using 20 mL of DI water by shaking and centrifuging the soil samples. We measured C and N concentration on each soil and sediment sample using a Costech 4010 Elemental Analyzer (Costech Analytical Technologies Inc., Valencia, CA, USA).

2.3. Plant C and N sampling

We quantified the C and N in herbaceous biomass by clipping vegetation in 0.89 m² hoops placed near each core location. Clipped vegetation was dried at 60 °C and weighed. We used estimates of C and N concentrations from herbaceous biomass in Nevada meadows of 0.35 g C g⁻¹ and 0.02 g N g⁻¹ biomass (Morra, 2022). To estimate the C and N

stocks of shrubs, we measured all shrubs within a 2.5-m radius of soil cores and used species specific allometric equations to convert measurements of length, width, and height to estimates of shrub biomass (SI Table 1, Reiner et al., 2010; United States Geological Survey, 2008). We assumed sagebrush tissue was 0.5 g C g⁻¹ C and 0.009 g N g⁻¹ C 0.9% N based on measurements collected by Austreng (2012) and Garcia-Moya and McKell (1969).

2.4. Estimates of C credits earned by management

We report the change in ecosystem C stocks between 1994 and 2021 (kg C m⁻²) by making comparisons between unmanaged geomorphic surfaces (floodplains, terraces, and uplands) and their managed counterparts. We considered ecosystem carbon as the sum of root C, soil C, and the C contained in aboveground biomass. In geomorphic surfaces where change in ecosystem C stock was significantly different than zero, (see following section) we estimated the yearly change in ecosystem C stock by assuming a linear rate of change. We present the resulting mass eCO₂ sequestered based on the area of geomorphic surface standardized by a 1-km stretch of valley length contained in a reach (Table 2). We recognize that interannual C stock change may not be linear from one year to the next, but linear models fit soil C increases in riparian meadow grasslands (Reed et al., 2022), abandoned agricultural fields (Knops and Bradley, 2009), and in fine textured soils of grasslands (Baer et al., 2010) over two decades following restoration.

2.5. Statistical analysis

All statistical tests were conducted using R (R Core Team, 2021). This study sought to understand how management changes ecosystem C relative to unmanaged surfaces. Therefore, we calculated an effect size, which was the difference between values measured on a geomorphic surface of the MCWRP minus the corresponding values from surfaces of the unmanaged reach. Because ecosystem C is made up of aboveground pools split into two plant functional groups (shrubs and herbaceous vegetation) and two belowground pools (soil C and root C split by depth), effect sizes were calculated for each C pool individually. This includes changes in soil bulk density (g cm⁻³), soil C concentration (% C), root C stock (kg m⁻²) soil C (kg m⁻²), aboveground biomass C (kg m⁻²), at each depth interval (0–15 cm, 15–30 cm, and 30–45 cm) of each geomorphic surface. We also calculated an effect size for changes in ecosystem C and N by summing C or N contained in roots and soils (0–45 cm) and aboveground biomass. We then calculated a 95% confidence interval assuming a one-tail t-distribution to show where changes were significantly different than zero. These same 95% confidence intervals are used to show a range of uncertainty presented in the results.

To compare the rates and evenness of gains in ecosystem C accumulation among surfaces where significant changes in ecosystem C stock occurred, we conducted a power analysis using the pwr package in R (Champlsey, 2020) to estimate the amount of C gain required to create an observable change in ecosystem C stock. This estimate uses our sample size, variance, and an alpha equal to 0.05 (Schrumppf et al., 2011). We used two different levels of statistical power, 0.1 and 0.2, to show the uncertainty of our power analysis. We then used the estimates of mean annual gains in ecosystem C to calculate the number of years these management treatments would require to cause a minimum detectable difference in ecosystem C stocks relative to those measured in the unmanaged reach. We calculated the coefficient of variance for ecosystem and soil C stocks to compare variance among fluvial surfaces.

We also made comparisons among ecosystem C stocks and soil C stocks of geomorphic surfaces by fitting linear mixed models using the lme4 package (Bates et al., 2015). These models contained a random intercept term to account for unmeasured differences in geomorphic surfaces. Where differences were observed, we made pairwise comparisons using least squared means. The *p*-values of multiple pairwise comparisons were adjusted using Bonferroni corrections.

3. Results

3.1. Changes in C, N, roots, and bulk density by depth

The floodplains in the managed grazing ephemeral pond reach had similar bulk density, C concentration, root C, soil C stock, and soil N stock, relative to the floodplains in the unmanaged reach (Fig. 2A). In terrace soils, C concentration increased by 0.99% ± 0.86 and 0.97% ± 0.72 in soils 0–15 cm and 15–30 cm respectively. Similarly, carbon stock increased by 1.41 ± 1.34 kg C m⁻² and 1.41 ± 1.27 kg C m⁻² in soils 0–15 cm and 15–30 cm respectively. In upland soils of the managed grazing ephemeral pond reach, root C increased by 0.16 ± 0.10 kg C m⁻², and soil C and N stock increased by 1.49 ± 1.24 kg C m⁻² and 0.18 ± 0.13 kg N m⁻² respectively at the middle depths of soils (15–30 cm) (Fig. 2A).

In the grazing excluded permanent pond reach, soil C concentration of 15–30 cm floodplain soils increased by 1.06% ± 0.63 (Fig. 2B). Root C increased at the 0–15, 15–30, and 30–45 cm depths by 1.11 ± 0.45, 1.23 ± 0.53, and 1.20 ± 0.46 kg C m⁻² respectively. Soil C stocks also increased by 3.55 ± 2.76 and 3.16 ± 2.74 kg C m⁻² in the 15–30 cm and 30–45 cm depths. In the terrace soils of the grazing excluded permanent pond reach, bulk density of soils decreased by 0.39 ± 0.19 and 0.41 ± 0.34 in soils 0–15 and 15–30 cm respectively (Fig. 2B). Percent C increased in soils 0–15 and 15–30 cm by 2.31% ± 1.17 and 1.41% ± 0.91 respectively. Similarly, soil C stocks increased by 2.64 ± 1.81 and 1.89 ± 1.72 kg C m⁻² in soils 0–15 and 15–30 cm. In these soils, N stock only increased at the shallowest depth (0–15 cm) by 0.19 ± 0.16 kg N m⁻². Upland soils in this reach were unchanged relative to the unmanaged reach (Fig. 2B).

In the grazing excluded ephemeral pond reach, floodplains were unchanged relative to their unmanaged counterparts (Fig. 2C). In the terraces of this reach, bulk density decreased by 0.48 ± 0.23 and 0.51 ± 0.28 in 0–15 and 15–30 cm soils respectively. Soil C concentration also increased 2.62% ± 1.48, 1.77% ± 0.85, and 1.73% ± 1.29 in 0–15, 15–30, and 30–45 cm soils. Despite a decrease in bulk density, soil C increased by 2.24 ± 1.24 and 2.26 ± 1.61 kg C m⁻² in 0–15 and 15–30 cm soils. In upland soils of the grazing excluded ephemeral pond reach, soil C concentration increased by 0.53% ± 0.49 in 0–15 cm soils and 0.42% ± 0.32 in 30–45 cm soils, root C increased by 0.09 ± 0.08 kg C m⁻² in 15–30 cm soils, soil C stock increased by 0.78 ± 0.70 kg C m⁻² in 0–15 cm soils, and N stock increased by 0.06 ± 0.05, 0.13 ± 0.12, and 0.12 ± 0.11, kg N m⁻² in soils 0–15, 15–30, and 30–45 cm respectively (Fig. 2C).

3.2. Components of ecosystem carbon

Ecosystem C stocks measured across all geomorphic surfaces and reaches were dominated by soil C (Fig. 3), which differed among reaches (*p* < 0.001), geomorphic surfaces (*p* < 0.001), and the interaction of geomorphic surfaces among reaches (*p* < 0.001; Supplementary Materials Table 1). Per unit area, the largest soil C stocks were in the floodplains of the grazing excluded permanent pond reach, where mean soil C stocks were 14.61 ± 2.07 kg C m⁻² (Supplementary materials Table 2). The next largest soil C stock (12.79 ± 1.35 kg C m⁻²) was in the terraces of the grazing excluded ephemeral pond reach (Supplementary materials Table 2). In the uplands, soil C ranged from 3.18 ± 0.70 kg C m⁻² in the unmanaged grazing reach to 5.52 ± 0.74 kg C m⁻² in the managed grazing ephemeral pond reach. However, upland soil C stocks were not significantly different among reaches (Supplementary materials Table 2). Soil stocks were spatially variable, but the spatial variation differed among reach and management history (Supplementary materials Table 2). After soil C, the next largest C pool was root C, which did not differ among geomorphic surfaces (*p* = 0.33) but differed by reach (*p* = 0.004) and the interaction of reach and geomorphic surface (*p* = 0.001) (Supplementary materials Table 3). Aboveground herbaceous plant C increased in the terraces and uplands of the grazing excluded ephemeral

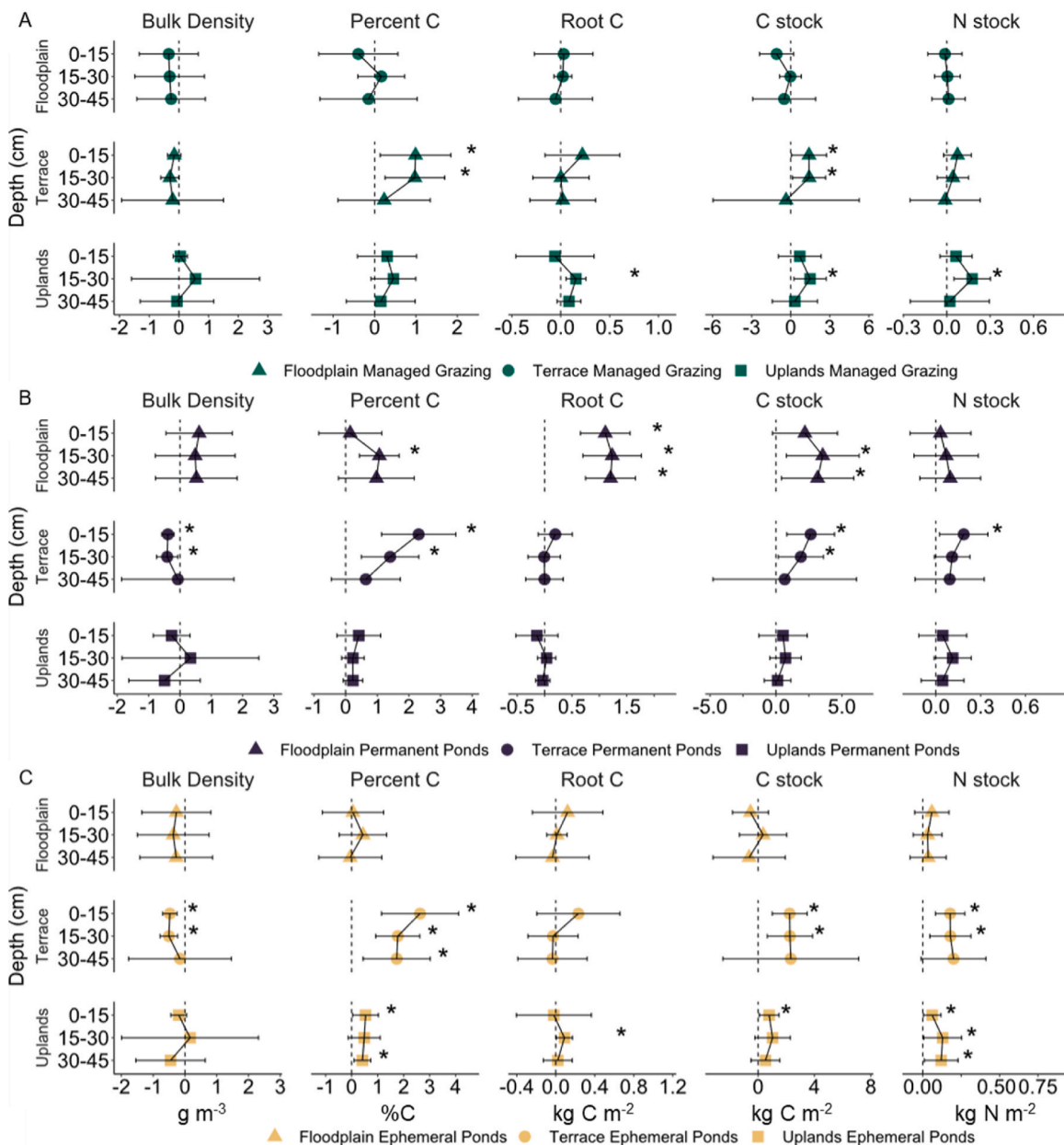


Fig. 2. Change in the soil of the managed grazing ephemeral pond reach (2A), change in the soil of grazing excluded permanent pond reach (2B), or change in soil of the grazing excluded ephemeral pond reach (2C) relative to unmanaged grazing. Error bars are the 95% confidence interval of a one tail t-distribution. An asterisk indicates changes that are different than zero.

pond reach and the terraces of the managed grazing ephemeral pond reach relative to corresponding surfaces in the unmanaged reach (Fig. 4A). Shrub C increased in the uplands of the grazing excluded permanent pond reach and the managed grazing ephemeral pond reach relative to corresponding surfaces in the unmanaged reach (Fig. 4B).

3.3. Changes in ecosystem carbon

There were five geomorphic surfaces where ecosystem C was significantly higher than the corresponding geomorphic surface within the unmanaged reach: floodplains of the grazing excluded permanent pond reach, terraces of the two grazing excluded reaches, and when considering only 0–30 cm soils, terraces and uplands of the managed grazing ephemeral pond reach (Fig. 5). Ecosystem C stocks were different among reaches ($p < 0.001$), geomorphic surfaces ($p < 0.001$), and the difference among geomorphic surfaces differed by reaches ($p <$

0.001) (Supplementary materials Table 4). The grazing excluded permanent pond reach was the only reach where ecosystem C increased on floodplains. Here, floodplains gained $12.19 \pm 5.39 \text{ kg C m}^{-2}$ of ecosystem C relative to floodplains in the unmanaged reach (Fig. 5) and the ecosystem C stock was $18.52 \pm 1.86 \text{ kg C m}^{-2}$, the largest ecosystem C stock we measured (Supplementary materials Table 5). Ecosystem C stock of ponds did not differ between permanent and ephemeral ponds within grazing excluded reaches where ecosystem C stocks were $5.3 \pm 0.58 \text{ kg C m}^{-2}$ and $5.0 \pm 0.66 \text{ kg C m}^{-2}$, respectively (Fig. 5, Supplementary materials Table 5). Both permanent and ephemeral ponds found in grazing excluded reaches contained more ecosystem C than the ponds in the managed grazing ephemeral pond reach (Supplementary materials Table 5). We consistently saw ecosystem C increased on terraces, where ecosystem C stock increased by 7.22 ± 5.57 and $5.68 \pm 5.58 \text{ kg C m}^{-2}$ in the grazing excluded ephemeral pond reach and the grazing excluded permanent pond reach, respectively, relative to the

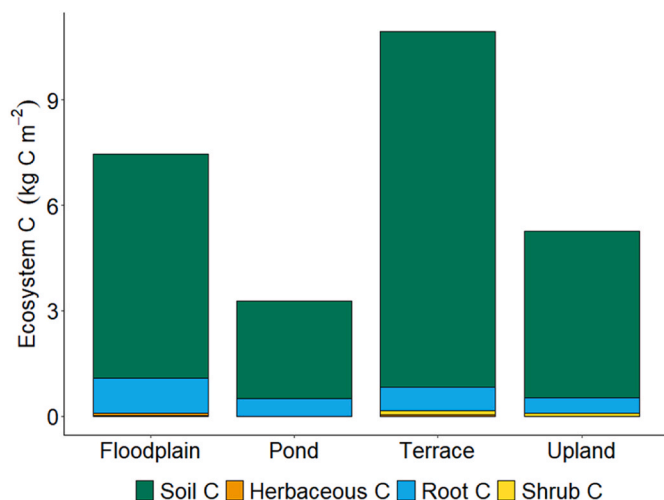


Fig. 3. Ecosystem C stock (kg C m^{-2}) is the sum of four components: aboveground shrub, aboveground herbaceous vegetation, root, and soil C stock found on floodplains, ponds, terraces, and upland areas calculated as the mean across all reaches. Root and soil C stocks are in the 0–45 cm mineral soil depth.

terraces found in the unmanaged reach (Fig. 5). When ecosystem C stocks were constrained to 0–30 cm soil depths, there was also an increase of $3.24 \pm 2.02 \text{ kg C m}^{-2}$ in ecosystem C on terraces in the managed grazing ephemeral pond reach. We observed small increases in ecosystem C in the uplands of the grazing excluded ephemeral pond reach where ecosystem C increased by $2.67 \pm 2.62 \text{ kg C m}^{-2}$ relative to uplands of the unmanaged reach (Fig. 5). When omitting 30–45 cm soil depths, ecosystem C in the uplands of the managed grazing ephemeral pond reach increased by $2.43 \pm 2.16 \text{ kg C m}^{-2}$ relative to uplands in the

unmanaged reach. We found no correlation between area impounded by beavers and ecosystem C stocks of ponds, floodplains, and terraces (Supplementary materials Table 6). Ecosystem C showed similar patterns of variance as soil C (Supplementary materials Fig. 1).

3.4. Potential sequestration from increases in ecosystem C

Mean annual gains in ecosystem C per unit surface area were similar among geomorphic surfaces, but when the annual gains in ecosystem C were multiplied by the area of a geomorphic surface occupying 1-km of valley length (Table 1), the amount of C sequestered by geomorphic surfaces ranged three orders of magnitude (Table 2). The largest spatially extrapolated mean C sequestration rate occurred on the terraces in the grazing excluded ephemeral pond reach, followed by uplands of the managed grazing ephemeral pond reach (Table 2). While ponds had similar increases in C stocks to terraces, they cover the least surface area and therefore sequester the least C annually of any geomorphic surface. We found spatially extrapolated C sequestration rates were better predicted by geomorphic surface width than C gains per unit area (Supplementary materials Fig. 3A). Assuming a linear increase over the project period, significant changes in ecosystem C gain may have been detectable within 5–13 years after the start of management (Table 2).

3.4.1. Changes in ecosystem nitrogen

Changes in ecosystem N were correlated with changes in ecosystem C ($p = 0.005$) (Fig. 6). Ecosystem N increased by $\sim 0.03 \text{ kg}$ for every kg increase in ecosystem C on most geomorphic surfaces (Fig. 6). The largest changes in ecosystem N, relative to the unmanaged grazing reach, occurred on the terraces of the two grazing excluded reaches (Table 3). The only surface in the managed grazing ephemeral pond reach where ecosystem N increased relative to unmanaged grazing was ponded areas. There was not a significant change in floodplain

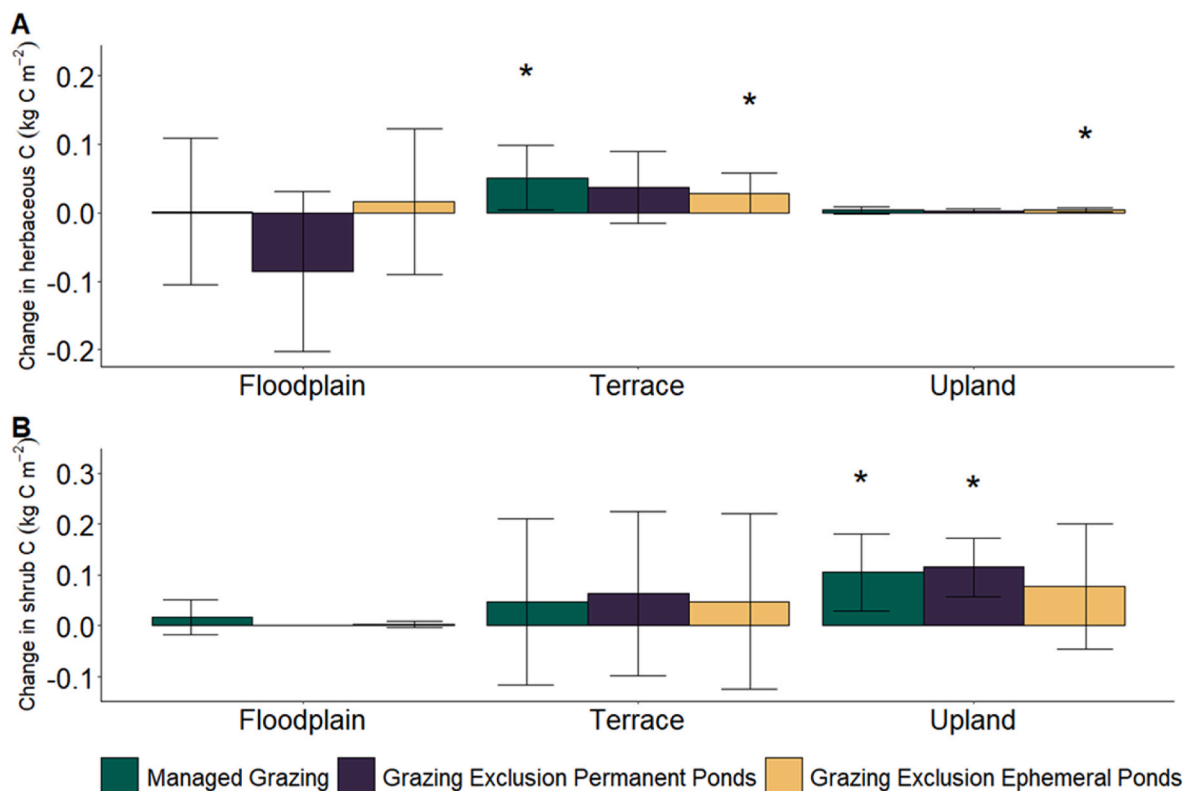


Fig. 4. Changes in aboveground C stocks in herbaceous plants (kg C m^{-2}) (4A) and shrubs (kg C m^{-2}) (4B) of the managed grazing ephemeral pond reach, grazing excluded ephemeral pond reach, and the grazing excluded permanent pond reach relative to corresponding surfaces in unmanaged reaches. Error bars are the 95% confidence interval and stars indicate changes that are significantly different than zero.

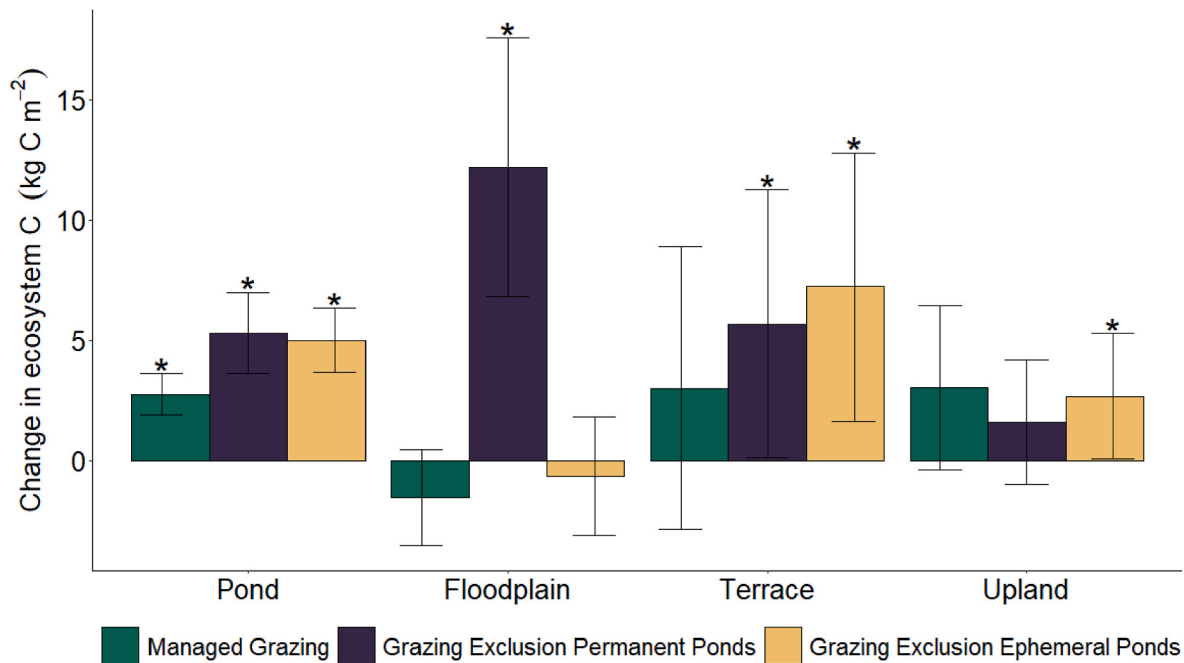


Fig. 5. Change in ecosystem C stocks (kg C m^{-2}) of the managed grazing ephemeral pond reach, grazing excluded ephemeral pond reach, and the grazing excluded permanent pond reach relative to corresponding geomorphic surfaces in the unmanaged reach. Results are shown for the full depth (0–45). Error bars are the 95% confidence interval and stars indicate changes that are significantly different than zero according to a one-tail *t*-test.

Table 2

Annual change in ecosystem C and the resulting generation of C credits ($\text{Mg eCO}_2 \text{ km}^{-1} \text{ yr}^{-1}$) normalized to 1-km of valley length. Ranges come from the 95% confidence interval of the difference between mean ecosystem C stock of a geomorphic surface from a reach within the MCWRP and the mean ecosystem C stock of its corresponding geomorphic surface outside of the MCWRP. Years to change represents the time required to surpass an effect size threshold generated by power analysis given our sample size, sample variance, alpha of 0.05, and statistical power ranging from 0.1 to 0.2.

Reach	Surface	Depth (cm)	Mean Annual Change kg C m^{-2}	Annual C credits per km of valley ($\text{Mg eCO}_2 \text{ km}^{-1} \text{ yr}^{-1}$)	Years to create significant change
Grazing excluded permanent ponds	Terrace	0–45	0.21 ± 0.2	2.4–268.0	6–13
Grazing excluded permanent ponds	Floodplain	0–45	0.35 ± 0.21	76.8–198.1	6–12
Grazing excluded permanent ponds	Ponds	0–45	0.20 ± 0.06	13.6–26.3	–
Grazing excluded ephemeral ponds	Upland	0–45	0.1 ± 0.01	3.3–303.5	6–12
Grazing excluded ephemeral ponds	Terrace	0–45	0.27 ± 0.21	98.9–767.4	5–11
Grazing excluded ephemeral ponds	Ponds	0–45	0.19 ± 0.05	1–1.7	–
Managed grazing	Uplands	0–30	0.09 ± 0.08	37.5–412.9	10–12
Managed grazing	Terrace	0–30	0.10 ± 0.01	40.7–175.2	6–13
Managed grazing	Ponds	0–45	0.10 ± 0.03	0.06–0.2	–

ecosystem N stocks among reaches. Based off the estimated annual increase in ecosystem N and the area of geomorphic surface held in 1-km of river valley within each reach (Table 1), across all geomorphic surfaces, annual gains in ecosystem N stocks range from 1 to ~22,000 g of N km^{-1} of valley length.

4. Discussion

In less than three decades, management that reduced grazing pressure on bankside vegetation resulted in increased ecosystem C and N on geomorphic surfaces extending from the stream channel to the edge of the hillslopes surrounding Maggie Creek. Instead of consistently finding the largest gains in ecosystem C and N near the stream channel, we found gains in ecosystem C and N differed by the interaction of geomorphic surface, grazing season and duration, and beaver pond

permanence. The largest changes in ecosystem C occurred in shallow soils (0–15 cm), but on most geomorphic surfaces, soil C increased to greater depths than measured in other restoration studies. Correlations between C and N gains suggests that increasing ecosystem C reduced N runoff to Maggie Creek and indicates C gains are reliant on N supply. We found high spatial variation in ecosystem C stocks, showing ecosystem C exhibits a diverse response to management within geomorphic surfaces in alluvial valleys.

Our results suggest C in semiarid riparian ecosystems responds rapidly to management that improves hydrology. Annual C gains in the MCWRP were much higher than those found in sagebrush-dominated ecosystems like those surrounding Maggie Creek. The annual ecosystem C gains we measured were an order of magnitude larger than net ecosystem exchange measurements from semiarid sagebrush steppe ecosystems in Wyoming, USA (Hunt et al., 2004). In fact, the

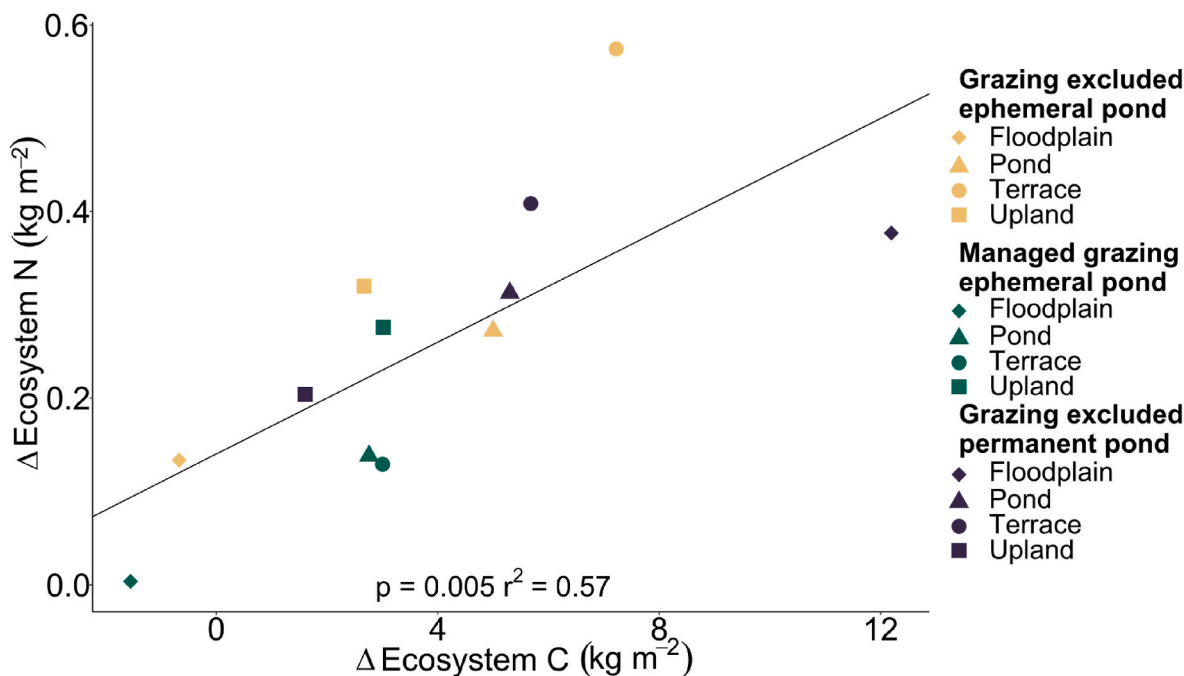


Fig. 6. Change in ecosystem N (kg N m^{-2}) plotted against change in ecosystem C (kg C m^{-2} ; soils 0–45 cm). Line is the result of a linear model plotting change in ecosystem N as a function of change in ecosystem C.

Table 3

Annual gains in ecosystem N for geomorphic surfaces and accumulation of N on the landscape normalized as g of N accumulated per 1-km valley length. Values are only shown where significant increases in ecosystem N were observed relative to unmanaged counterparts according to a one-tailed *t*-test.

Reach	Surface	Depth (cm)	Mean Annual Change (g N m^{-2})	Annual accumulation ($\text{g N yr}^{-1} \text{ km}^{-1}$)
Grazing excluded permanent ponds	Ponds	0–45	5.56–14.44	231.13–600.11
Grazing excluded ephemeral ponds	Ponds	0–45	3.37–19.59	10.30–59.62
Managed grazing	Ponds	0–45	1.86–8.52	1.06–4.88
Grazing excluded permanent ponds	Terraces	0–45	2.97–27.41	780.8–7205.93
Grazing excluded ephemeral ponds	Terraces	0–45	9.26–32.96	6133.8–21,832.52
Grazing excluded ephemeral ponds	Uplands	0–45	5.18–32.96	3289.3–11,760.33

accumulation of C in the MCWRP was similar to, or greater than, annual gains found in restored ecosystems located in more humid climates (Baer et al., 2002; Meyer et al., 2008). Relative to ecosystem C gains in restored wetlands in Nebraska, USA (0.22 and $0.26 \text{ kg C m}^{-2} \text{ yr}^{-1}$; Meyer et al., 2008), terraces and ponds of both grazing excluded reaches were similar, and floodplains of the grazing excluded permanent pond reach in the MCWRP gained ecosystem C one and a half to two times faster ($0.45 \pm 0.20 \text{ kg C m}^{-2} \text{ yr}^{-1}$). The behavior of livestock can have implications for ecosystem processes (Augustine and McNaughton,

1998). Under managed grazing, annual gains in ecosystem C of ponds and terraces (0.10 and $0.11 \text{ kg C m}^{-2} \text{ yr}^{-1}$ respectively), were similar to restored Minnesota, USA grasslands ($83.3 \text{ g C m}^{-2} \text{ yr}^{-1}$; Baer et al., 2002). Therefore, ecosystem C gains are of a similar magnitude, or greater than, those found in other wetland and grassland ecosystems located in more humid ecosystems.

We expected most soil C gains to be in shallow (0–15 cm) soil depths because prior research in other riparian ecosystems measured significant C gains in the top 15–20 cm and modest C gains below 20 cm (Knops and Bradley, 2009; Reed et al., 2022). Our results partially support this concept: across reaches and geomorphic surfaces, the greatest gains in soil C occurred in 0–15 cm depths (Fig. 2A, 2B, 2C). Yet to our surprise, we measured significant soil C gains as deep as 45 cm in the soil profile of some areas of the MCWRP. Annual gains in soil C stocks ranged from 86 to $329 \text{ g C m}^{-2} \text{ yr}^{-1}$ across reaches, showing similar soil C gains as those in the California Sierra Nevada, USA ($232 \text{ g C m}^{-2} \text{ yr}^{-1}$; Reed et al., 2022), but distributed to greater soil depths. We speculate that deeper soil C increases in this ecosystem may be due to deep rooting characteristics of semiarid vegetation (Fan et al., 2017) and downward percolation of dissolved organic C when bank-full height of the stream channel is exceeded (Mikutta et al., 2019). Carbon stored at greater depths might be more stable due to its isolation from microbial communities and other resources required for decomposition (Dyarski et al., 2020). Semiarid ecosystems constitute 41% of Earth’s landmass (Anderson and Inouye, 2001). Thus, identifying ecosystems where soil C may rapidly increase in response to management within semiarid regions may be an important step to increase terrestrial C sequestration.

We anticipated that gains in ecosystem C would increase with proximity to the stream channel resulting in the largest gains in floodplains and smallest gains in uplands. This occurred where permanent ponds developed but not in the two reaches containing ephemeral ponds. We ascribe this outcome to dam failure, which can cause over-bank flooding and floodplain scouring (Westbrook et al., 2011), limiting C gains in floodplains and allowing C gains in terraces away from the stream channel to exceed those in the channel. Conversely, the grazing excluded permanent pond reach contained the widest floodplains among reaches (Table 1). Reduced erosion provided by wide floodplains and permanent beaver ponds likely supported the large increases in root

mass and soil C found in the grazing excluded permanent pond reach (Fig. 2B). This result is consistent with our expectation that changes in root mass accompany gains in soil C. In addition to improved hydrology, changes to grazing likely improved plant productivity by allowing time for plant establishment and regrowth under managed grazing (Swanson et al., 2018) and reduced soil bulk density where grazing was excluded (Fig. 2B and 2C). Reduced soil compaction can increase the thickness of soils, providing increased water storage and lengthening the growing season by as much as 35 days in meadows (Lowry and Loheide, 2010).

Soil C gains were likely the result of a longer growing season leading to further primary productivity and belowground C allocation. Compared to stream reaches without beaver ponds, Fairfax and Small, 2018 found plant communities adjacent to the MCWRP had higher rates of evapotranspiration and had a longer growing season where beaver had built ponds. A longer growing season and greater soil water availability for plants in otherwise semiarid ecosystems likely extended the duration of plant contributions to soil C in terraces and uplands that gained ecosystem C.

Though mean ecosystem C gains were large, high spatial variability of ecosystem C gains demonstrate the inherently patchy nature of C responses to management. Ecosystem C stocks are inherently variable, even within a constrained landscape. In alluvial valleys such as the one that surrounds Maggie Creek, frequent channel avulsion creates heterogeneity by leaving behind riparian plant communities which still have access to shallow groundwater as well as bands of well sorted soil particles from relict stream channels, which influence water and nutrient availability (Austin et al., 2004; Naiman et al., 2005; Noy-Meir, 1973). The variance in ecosystem C at Maggie Creek ranged from 10% to 60% and was higher than variance reported by Schrupf et al. (2011) who found coefficients of variance ranging from 12% to 29% in grasslands globally. High spatial variability has management implications, such as making small changes in ecosystem C more difficult to detect. Therefore, in similar ecosystems, high sampling intensity and careful experimental design may improve estimates of ecosystem C change. Furthermore, managers should expect that restoration impacts in ecosystems with variable C stocks may require more time to observe than ecosystems experiencing C gains with lower variation.

While grazing management focused primarily on riparian condition, we observed increased ecosystem C stocks of geomorphic surfaces that were more than 500 m from the stream channel, indicating that studies measuring C gains in riparian areas should consider the entire valley width. The cross-valley influence of grazing management on ecosystem C stocks of Maggie Creek created the counterintuitive result that smaller unit area C gains found in uplands and terraces resulted in more C sequestration at the reach scale than in geomorphic surfaces constrained by hydrology (ponds and floodplains). Therefore, considering C gains across the full valley width, and applying management to greater reach lengths, could be the most reliable way to sequester large amounts of C. The value of building C on terraces and uplands is especially important when considering pond failure which resulted in limited C build up on floodplains of reaches with ephemeral ponds. While floodplains found in the grazing excluded permanent pond reach sequestered a large amount of C, the permanence of beaver dams is likely controlled by valley width rather than the exclusion of grazing (Pollock et al., 2014).

We found C:N ratios remained roughly constant across stream reaches and geomorphic surfaces, fitting our expectation that there is a stoichiometric relationship between C and N in plants and soils. In addition to sequestering C, Grazing management in the MCWRP likely improved soil fertility and water quality of the adjacent waterway by facilitating C sequestration and accumulating N. Depending on expected yield, recommended N application to pastures from fertilizer ranges from 5 to 22 g m⁻² (Koenig et al., 2002) which was similar to annual gains in the MCWRP, which ranged from 5 to 25 g m⁻². The correlation between C and N means the continued C gains facilitated by improved hydrology and grazing management may also rely on the availability of N. For example, in restored tailings dams in Ziaochinlin, China, gains in

soil C and N occurred rapidly during the initial five years following restoration, but further gains in soil C decreased and no change in soil N occurred beyond five years (Wang et al., 2018).

5. Conclusions

We show the positive impact grazing management can have on ecosystem function of riparian meadows in a semiarid climate. Improvement of bankside vegetation was important for slowing channel incision and the reestablishment of beaver. Improved hydrology lengthened the growing season, and increased ecosystem C and N stocks on geomorphic surfaces that extended to the valley edge. While increased soil nutrients and water storage might be of interest to land users, voluntary C markets may also provide monetary incentives to change land management or fund restoration in the near future (Kreibich and Hermwille, 2021). The MCWRP includes 128 km of stream channel (JBR, 2002), meaning between 10,022 and 137,292 Mg of eCO₂ could be sequestered in a single year. At scales larger than Maggie Creek, the potential for grazing management to increase C storage may be of great importance in Nevada, where 63% of the land area is authorized for grazing by the BLM (BLM, 2022) and where there are an estimated 2526 km² of riparian area (Saito et al., 2020), representing a potentially substantial opportunity for C sequestration in the driest state in the USA. As a result of its aridity, the total area used for production of crops in Nevada is only 1117 km² (USDA, 2017) making the large fluxes and large area occupied by riparian ecosystems of Nevada a potentially greater contributor to C sequestration than croplands. While we found the largest gains in ecosystem C in areas where grazing was excluded, managed grazing successfully increased ecosystem C of ponds, terraces, and uplands, showing C sequestration and grazing don't have to be mutually exclusive. This was because managed grazing and grazing exclusion both help promote ecological recovery, leading to the recuperation of ecosystem C stocks. Management should focus on restoration that repairs ecological processes of riparian ecosystems (hydrology, plant communities, and habitat (Beechie et al., 2010)), as they can directly improve conditions of the subsurface.

Credit author statement

Brian Morra: Data collection, data analysis, lab work, writing – original draft **Ben Sullivan:** Writing – original draft, supervision, resources, conceptualization **Tamzen Stringham:** Writing- review and editing, provided site access, resources, conceptualization related to land management **Hondo Brisbin:** Significant data collection in the field, Writing- review and editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

This work was supported by the USDA National Institute of Food and Agriculture, Hatch Project NEV05299 (Accession No. 1016043). This work would have been impossible without the historical data provided by Carol Evans (BLM) and access to private lands provided by David Voth (Nevada Gold) and Jon Griggs (Maggie Creek Ranch). Thanks to Julia Stiltz and Kevin Charpentier for field assistance. We thank four anonymous reviewers for their efforts.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jenvman.2023.117724>.

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