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Riparian health improves with managerial effort to implement livestock distribution practices

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Abstract. Optimising the spatial distribution of free-ranging livestock is a significant challenge in expansive, grazed landscapes across the globe. Grazing managers use practices such as herding (i.e. droving), strategic placement of off-stream livestock drinking water and nutritional supplements, and strategic fencing in attempts to distribute livestock away from sensitive streams and riparian areas. We conducted a cross-sectional survey of 46 cattle-grazed riparian areas and associated stream reaches embedded in rugged range landscapes to examine relationships between implementation of these management practices, stocking rate, and riparian health. We determined in-stream benthic invertebrate assemblages at each site to serve as an integrative metric of riparian health. We also collected information from the grazing manager on stocking rate and implementation of livestock distribution practices at each site over the decade before this study. Off-stream livestock drinking-water sources were implemented at just two sites (4.3%), indicating that this was not a common distribution practice in these remote management units. We found no significant relationship of riparian health (i.e. invertebrate richness metrics) with stocking rate ($P \geq 0.45$ in all cases), or with the simple implementation (yes/no) of off-stream nutritional supplements, fence maintenance, and livestock herding ($P \geq 0.22$ in all cases). However, we did find significant positive relationships between riparian health and managerial effort (person-days spent per year for each individual practice) to implement off-stream nutritional supplements and fence maintenance ($P \leq 0.017$ in all cases). Livestock herding effort had an apparent positive association with riparian health ($P \geq 0.2$ in all cases). Results highlight that site-specific variation in managerial effort accounts for some of the observed variation in practice effectiveness, and that appropriate managerial investments in grazing distributional practices can improve riparian conditions.

Keywords: benthic invertebrates, best management practices, water quality.

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Introduction

Optimising the spatial and temporal distribution of free-ranging livestock across complex landscapes is a significant challenge facing managers in vast grazing areas (Malan *et al.* 2018; Creamer *et al.* 2019). Stocking rate (i.e. number of livestock per unit time per unit area) is a primary determinant of economic and ecological outcomes of grazing-management decision-making (e.g. Briske *et al.* 2011; Byrnes *et al.* 2018). However, it has long been recognised that, even at appropriate stocking rates, inherent spatial and temporal variability in landscape elements such as vegetation, forage quality, drinking water, topography, and micro-climate result in non-uniform patterns of livestock grazing at multiple scales (e.g. feeding station, home range, landscape) (Bailey *et al.* 1996; Hunt *et al.* 2007). Livestock behavioural tendencies and habitat preferences interact with these landscape elements to create further challenges for managers attempting to optimise livestock distribution (Bailey 2005; Ganskopp and Bohnert 2009; Roche *et al.* 2014; Creamer *et al.* 2019). In the absence of appropriate management, these interacting environmental and behavioural factors are primary drivers of excessive livestock damage to sensitive habitats such as streams and associated riparian areas, which provide drinking water, forage, and microclimates sought by livestock (O'Callaghan *et al.* 2019). Riparian areas make up a small percentage of these landscapes, but disproportionately provide critical ecosystem services such as flood attenuation, nutrient sequestration, habitat, and clean water (Norton *et al.* 2011; Acreman and Holden 2013). Excessive livestock grazing pressure impairs the health of riparian areas and associated stream reaches via a series of cascading impacts on plant communities, stream-channel stability, hydrologic function, and aquatic habitat (Fleischner 1994; Belsky *et al.* 1999; Bartley *et al.* 2017).

Fortunately, there is also evidence that contemporary conservation-grazing management strategies can reduce livestock damage to riparian areas in expansive grazed landscapes. Grazing management practices such as herding, strategic placement of livestock nutritional supplements and drinking water stations, and fencing have the potential to reduce negative impacts of livestock to rangelands and riparian areas (Bailey 2004; George *et al.* 2011; Hunt *et al.* 2014; Malan *et al.* 2018; O'Callaghan *et al.* 2019). In North America, herding (i.e. droving) is the common management practice of moving groups of livestock (i.e. herds) from one area to another by managers on horseback or all-terrain vehicles, often with the aid of livestock dogs; in this case, livestock are herded away from riparian areas to upland areas to reduce livestock impacts to riparian resources. O'Callaghan *et al.* (2019) demonstrate that fencing to exclude livestock from riparian areas can provide high efficacy for improving some aspects of riparian and stream health. Although generally effective, complete riparian fencing is not practical across vast, remote grazing lands. Thus, strategic use of fencing in conjunction with other livestock distribution practices is likely a more feasible approach. Reviewing the efficacy of non-fence-dependent practices such as off-stream livestock attractants (e.g. drinking water, nutritional supplements, shade), Malan *et al.* (2018) and George *et al.* (2011) conclude that these practices are generally effective across a diversity of grazing lands. However, both papers stress that the substantial variability observed across studies is likely due in large part to variable site-specific factors such as size of management unit, livestock type, season of use, vegetation patterns, and topography.

Managerial effort invested in implementation (i.e. person-days per year spent on implementation) can also influence variability in effectiveness of distributional practices. For example, a manager may 'implement' an off-stream station with nutritional supplements to entice livestock away from a riparian area. However, during initial implementation, the manager may not invest enough in site-specific assessments of livestock utilisation patterns to allow sufficiently informed decisions on locating the station with the best chance of improving distribution. Subsequently, the manager may then inadequately invest effort (time) in monitoring the station (e.g. ensuring that livestock locate and utilise the nutritional supplements) and maintaining it (e.g. replacing supplements as consumed, moving station to a new area as associated forage is depleted) to achieve desired reductions in livestock damage to the riparian area of concern. Benthic invertebrate metrics can be robust indicators of stream and riparian health (Merritt *et al.* 2008), and have been found to be sensitive to site-specific grazing management (e.g. Herbst *et al.* 2012; Magierowski *et al.* 2012). For example, while comparing the impacts of three different grazing systems on riparian health, Magner *et al.* (2008) found significant, congruent responses of soil, vegetation, and invertebrate metrics to cattle impacts. In this paper, we report the results of a cross-sectional survey of 46 grazed riparian areas, with the specific objective of evaluating relationships of stocking rate and managerial effort to implement livestock distributional practices with riparian health as assessed by in-stream benthic invertebrate assemblages.

Materials and methods

Study area and site selection

We selected 46 grazed riparian montane meadows and associated stream reaches in this cross-sectional survey. Study sites were within active grazing-management units across six National Forests ($n = 34$ sites) and 12 privately owned ranches (Fig. 1) throughout the Sierra Nevada and Southern Cascade mountain ranges and the semi-arid plateaus of eastern California, USA (37.04–41.80°N; 121.55–119.12°W). This subsample of grazing units typifies the challenges that managers face in achieving optimal livestock distribution (Bailey *et al.* 1996) and thus enhancing riparian health (e.g. challenges such as livestock congregating in riparian areas, trampling streambanks and sensitive plant species, and polluting surface waters with faecal waste) (Oles *et al.* 2017; Lewis *et al.* 2019). Elevations at study sites across this rugged terrain ranged from 1025 to 2610 m. The mountainous region is influenced by California's Mediterranean climate with warm, dry summers and cold, wet winters. The majority of precipitation falls as snow between December and March, with snowmelt typically occurring between April and May. Thirty-year mean annual precipitation for the study area ranges from 61 to 130 cm. Following peak spring flows, streamflow declines rapidly to base flow conditions during the summer (June–September).

All sites were alluvial systems with Rosgen Category C or E stream-channel morphology, characterised by riffle/pool bedforms, fine to gravelly substrates, and stream channel profile slopes <2% (Rosgen 1996). These perennial flowing, headwater stream types and associated small (e.g. 1–10 ha) meadow riparian areas are particularly sensitive to grazing disturbance (George *et al.* 2011). Riparian plant communities were dominated by herbaceous species such as sedges (*Carex* spp.), rushes (*Juncus* spp.), and grasses. Stream-channel aerial canopy cover from riparian woody species such as willows (*Salix* spp.) and was <10% for all sites. Depending on elevation and precipitation, upland ecosystems ranged from mesic and xeric mixed conifer forests to xeric shrublands.

Data collection

We collected information on grazing management for the 10 years before the study from the grazing-unit managers at each site. All sites had consistent grazing management during the 10-year period and were grazed by commercial cow–calf operations using conventional crossbred beef cattle (i.e. Angus × Hereford) typical of the region. Some grazing-management units had interior cross fences, creating paddocks to improve grazing distribution throughout the overall management unit. Some grazing-management units had 'drift' fences strategically placed to keep livestock from 'drifting' into areas that managers did not want grazed at that time. For example, a drift fence placed across a trail over a mountain pass can prevent livestock movements between catchments until the manager opens the fence. We quantified implementation (yes or no) and managerial effort (days per year) invested in each of the following individual activities: (*1*) strategic placement of off-stream nutritional supplements (e.g. salt, mineral, protein) to attract livestock away from riparian areas and into upland areas of the management unit; (*2*) maintenance of fences to improve

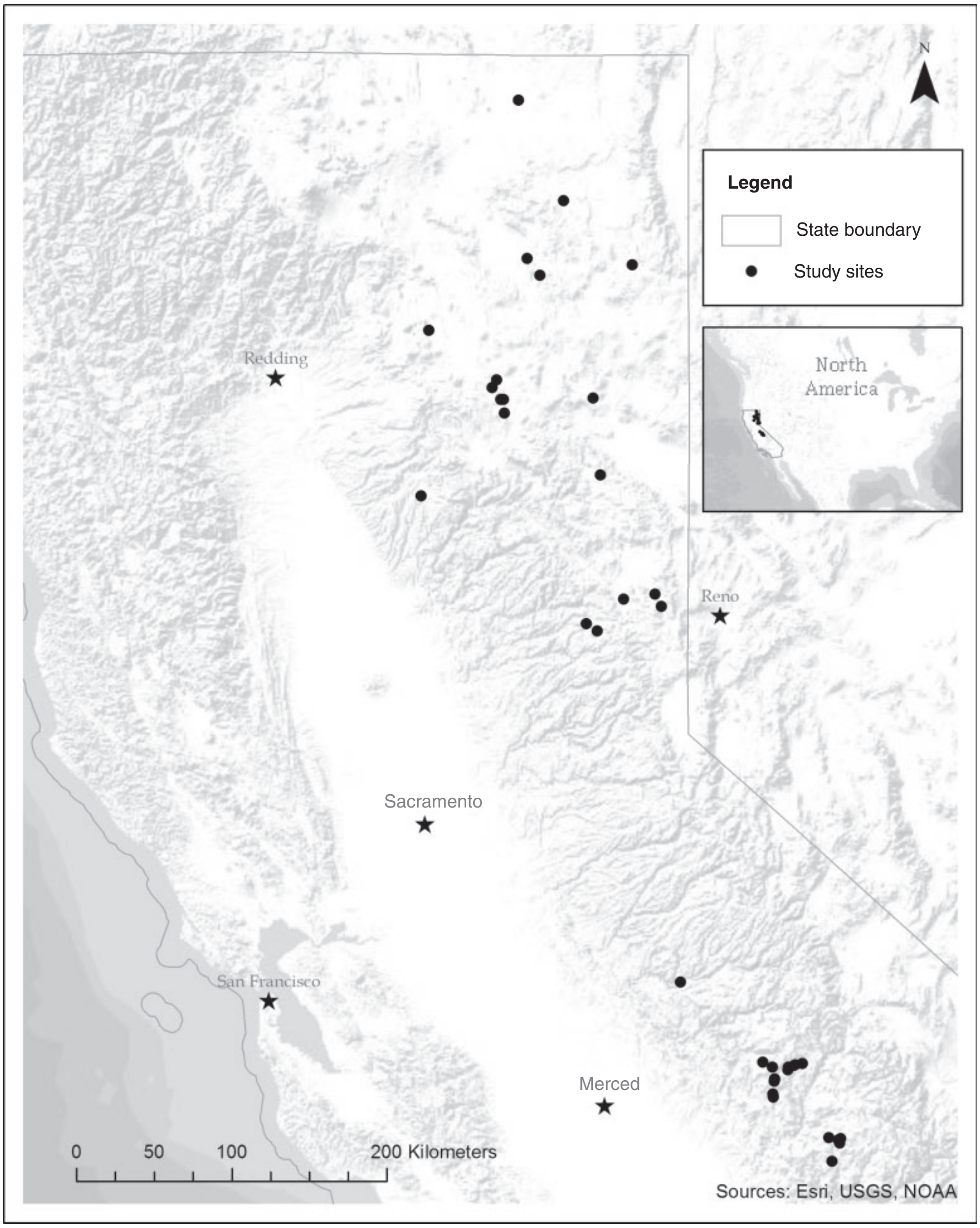

Fig. 1. Study sites (total 46) enrolled in this study across extensive, mountainous grazing lands in east-central and north-eastern California.

managerial control over livestock utilisation of riparian areas within the management unit (e.g. monitoring the management unit's boundary fence, interior cross fence, and drift fence conditions and effectiveness; repairing and maintaining fences and gates); and (*3*) active herding of livestock to improve grazing distribution across the management unit. We also quantified mean annual stocking rate for all management units as animal-unit months (forage demand of a 450-kg cow with or without calf during a 30-day period) per hectare (AUM/ha).

We collected benthic invertebrate assemblage samples at each of the 46 sites between June and August (summer growing and grazing season) during summer base-flow conditions. Base-flow stream widths during the collection period ranged from ~0.5 to 2.0 m, with mean width ~1 m across the 46 study sites. Base-flow stream discharge during sample collections ranged from ~10 to 50 L/s with a mean of ~30 L/s across the 46 study sites. Collections occurred once per site within a 100-m stream reach at each site. Lower elevation sites were sampled earlier in summer than higher elevation sites to account for seasonal progression in development of macro-invertebrate assemblage. Each sample (n = 46) was a composite of six subsamples collected along two transects (three subsamples per transect) placed perpendicular to streamflow in two riffle areas of each study reach. Number of riffle areas within the 100-m

Table 1. Summary of livestock distributional management practices used at 46 study sites on US Forest Service grazing allotments and privately owned pastures in north-eastern and central California

Management activity	Implementation frequency (% of sites)	Managerial effort (person-days/year)			
		Minimum	Median	Mean	Maximum
Off-stream nutritional supplements	78	0.0	1.1	1.9	8.0
Pasture fence maintenance	70	0.0	2.5	5.9	25.0
Livestock herding	67	0.0	6.0	9.9	30.0

stream reach ranged from two to five with a mean of approximately three across the 46 study sites. Each subsample was collected by using a 30-cm-wide D-ring kick net with a 500-µm mesh collection bag (Barbour *et al.* 1999) during a standardised 3-min collection effort over an area of 30 by 30 cm of the streambed substrate. All subsamples for a study site were immediately composited as one sample and stabilised in the field with denatured ethanol. These samples ($n = 46$) were then taxonomically identified to family, genus, and in some cases species according to a standardised taxonomic effort (SAFIT level II; Richards and Rogers 2011).

Data analyses

The following benthic invertebrate assemblage metrics, which are commonly used in biomonitoring procedures (Barbour *et al.* 1999), were calculated for each study site and used as indicators of stream and riparian health in the data analysis (response variables): (*1*) total taxa richness; (*2*) total *Ephemeroptera*, *Plecoptera*, and *Trichoptera* (EPT) taxa richness; and (*3*) intolerant taxa richness. Intolerant taxa in this study were defined as taxa rated 0 through 3 on the Hilsenhoff Biotic Index (HBI) for sensitivity to pollution and water-quality degradation (Hilsenhoff 1988). The HBI scale ranges from 0 (sensitive species present, indicating excellent water quality, pollution unlikely) to 10 (insensitive species present, indicating very poor water quality, severe pollution likely). Taxa scored 0–3 on the HBI are considered intolerant of even slight levels of pollution. All three richness metrics decrease with habitat perturbation and pollution, have a positive correlation to riparian health (Barbour *et al.* 1999; Merritt *et al.* 2008), and have been found sensitive to site-specific grazing management in the region (Herbst *et al.* 2012).

We utilised negative binomial regression analysis to assess relationships between (*i*) the three count-based richness metrics described above; (*ii*) stocking rate (AUM/ha); and (*iii*) the six metrics of livestock distribution management, which comprised implementation (yes/no) of the three livestock distribution practices of off-stream nutritional supplements, maintenance of management unit fencing, and livestock herding, as well as the average annual managerial effort (person-days per year, days/year) expended to implement each of those practices individually (i.e. days/year invested in each practice). Individual regression analyses were conducted for each of the three richness metrics (response variables) and stocking rate and each of the six livestock distribution management metrics, for 21 individual analyses. Standard diagnostics confirmed that the assumptions of negative binomial regression analysis were valid. All statistical analyses were conducted in Stata/SE 13.1 (StataCorp, College Station, TX, USA).

Results

In total, the grazing-management units containing the 46 sites enrolled in the study covered ~255 000 ha. Over the 10 years before the study, management units were grazed during the summer growing season, from June (spring) to September (fall), with a median annual grazing period of 64 days. The grazing units were large (mean ± s.e.: 9233 ± 2013 ha) with low cattle numbers (mean ± s.e.: 189 ± 23 head) and associated stocking rates (mean ± s.e.: 0.13 ± 0.04 AUM/ha) and riparian areas within the units were all open to livestock access (i.e. no riparian exclusion fencing). There were no off-stream livestock drinking-water sources within 36 of the management units. An additional eight management units did have drinking water sources; however, the sources were within the riparian area and thus did not serve as a distributional practice. Only two units had drinking-water sources intended to attract livestock away from riparian areas. Off-stream nutritional supplements were annually implemented at 78% of units, fence maintenance was annually implemented at 70% of units, and livestock herding was annually implemented at 67% of units. Over the 10-year period, the range in average reported annual managerial effort invested in each practice increased from maintenance of off-stream nutritional supplementation (0–8 days/year) to fence maintenance (0–25 days/year) to livestock herding (0–30 days/year) (Table 1).

We identified 190 benthic invertebrate taxa representing 10 classes, 17 orders, and 64 families in the pool of samples collected across the 46 study sites. Taxonomic classes observed across the 46 study sites were *Actinopterygii*, *Arachnida*, *Entognatha*, *Gastropoda*, *Insecta*, *Malacostraca*, *Maxillopoda*, *Oligochaeta*, *Ostracoda*, and *Turbellaria*. Orders and families observed are detailed in Table 2. Mean values for key metrics of invertebrate richness and composition are presented in Table 3. We found no significant relationships of any of the three benthic invertebrate richness metrics with grazing-management unit stocking rate ($P \geq 0.45$ in all cases), and no significant relationships of any of the three richness metrics with implementation (yes/no) of off-stream nutritional supplements ($P \geq 0.23$ in all cases), fence maintenance ($P \geq 0.22$ in all cases), or livestock herding ($P \geq 0.34$ in all cases) (regression results not shown). We found significant positive relationships of all three richness metrics with managerial efforts (days/year) to implement off-stream nutritional supplements ($P \leq 0.008$ in all cases) and to maintain fences ($P \leq 0.017$ in all cases) (Table 4, Fig. 2). Livestock herding effort was positively, but not significantly, associated with richness metrics ($P \geq 0.2$ in all cases).

Table 2. Summary of benthic invertebrate taxa by order and family collected across 46 study sites on US Forest Service grazing allotments and privately owned pastures in north-eastern and central California

Values in parentheses are the number of benthic invertebrates found in each family

Taxonomic order	Taxonomic families
Trichoptera	Hydropsychidae (551), Glossosomatidae (357), Philopotamidae (265), Brachycentridae (224), Rhyacophilidae (170), Lepidostomatidae (155), Apataniidae (140), Hydroptilidae (121), Limnephilidae (74), Sericostomatidae (55), Psychomyiidae (37), Uenoidae (10), Helicopsychidae (3), Polycentropodidae (3)
Diptera	Chironomidae (7610), Simuliidae (4197), Tipulidae (132), Ceratopogonidae (98), Empididae (11), Psychodidae (11), Tabanidae (5), Ptychopteridae (4), Ephydridae (3), Dixidae (2)
Plecoptera	Nemouridae (1467), Chloroperlidae (465), Perlodidae (227), Peltoperlidae (128), Perlidae (77), Leuctridae (23), Pteronarcyidae (16), Capniidae (2)
Coleoptera	Elmidae (2478), Dytiscidae (198), Psephenidae (178), Haliplidae (46), Hydrophilidae (19), Hydraenidae (8), Helophoridae (3), Staphylinidae (2)
Ephemeroptera	Baetidae (3273), Ephemerellidae (1179), Leptophlebiidae (987), Heptageniidae (633), Leptohyphidae (268), Ameletidae (43), Isonychiidae (3)
Odonata	Coenagrionidae (343), Gomphidae (10), Libellulidae (7)
Amphipoda	Hyalellidae (194), Gammaridae (60)
Megaloptera	Sialidae (18), Corydalidae (2)
Heteroptera	Naucoridae (3), Gerridae (2)
Trombidiformes	Unidentified family (318)
Harpacticoida	Unidentified family (58)
Isopoda	Asellidae (20)
Neotaenioglossa	Hydrobiidae (12)
Collembola	Unidentified family (7)
Basommatophora	Ancylidae (3)
Cyclopoida	Unidentified family (1)
Lepidoptera	Unidentified family (1)

Table 3. Summary of benthic invertebrate metrics calculated for samples collected across 46 study sites on US Forest Service grazing allotments and privately owned pastures in north-eastern and central California

Metric	Mean ± s.e.
Total abundance	200.4 ± 35.6
Total number of taxa	17.7 ± 5.9
Number of EPT (Ephemeroptera + Plecoptera + Trichoptera) taxa	8.4 ± 4.9
EPT richness index ((total no. of taxa/no. of EPT taxa) × 100)	43.9 ± 16.2
Relative abundance of EPT ((EPT abundance/total abundance) × 100)	34.5 ± 20.9
Number of intolerant taxa	4.6 ± 3.4
Relative abundance of intolerant taxa ((intolerant taxa abundance/total abundance) × 100)	15.3 ± 15.8

Table 4. Negative binomial regression results for models testing relationships between macroinvertebrate community richness metrics and managerial effort (person-days/year) to implement livestock distribution practices

Richness metric	Managerial effort	Coefficient	*P*-value	Intercept
Total taxa	Off-stream nutritional supplements	0.060	0.005	2.74
	Pasture fence maintenance	0.017	0.014	2.76
	Livestock herding	0.005	0.221	2.81
EPT taxa	Off-stream nutritional supplements	0.112	0.005	1.87
	Pasture fence maintenance	0.030	0.017	1.92
	Livestock herding	0.007	0.395	2.04
Intolerant taxa	Off-stream nutritional supplements	0.131	0.008	1.21
	Pasture fence maintenance	0.037	0.012	1.25
	Livestock herding	0.012	0.202	1.37

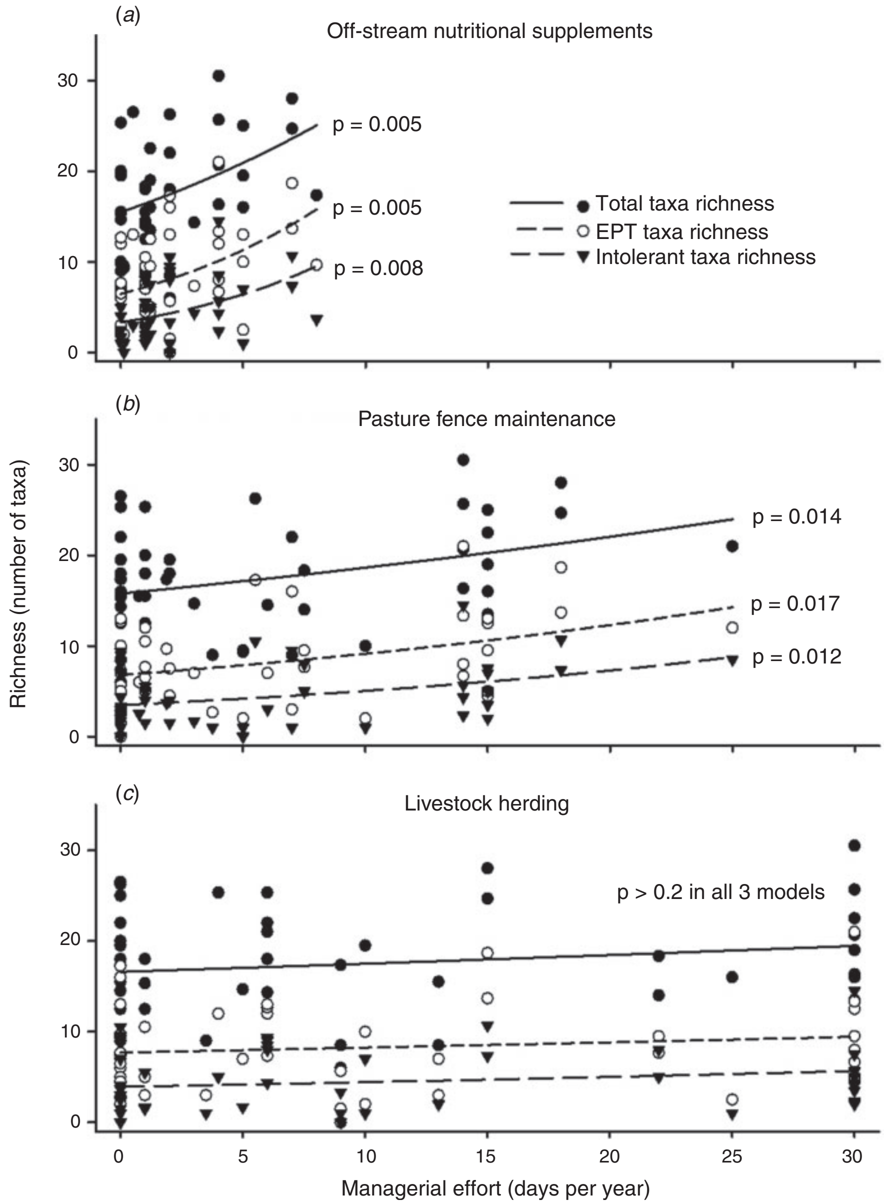

Fig. 2. Significant relationships of taxa richness metrics with effort-days for three livestock distribution management practices at riparian mountain meadows and adjacent stream reaches ($n = 46$) on US Forest Service grazing allotments and privately owned pastures in north-eastern and central California. EPT, Ephemeroptera + Plecoptera + Trichoptera. Symbols are observed data, and lines are model predictions.

Discussion

Our results indicate that riparian health, assessed by using benthic invertebrate metrics, can be enhanced with effective implementation of livestock distribution practices to reduce livestock utilisation in sensitive areas. These findings are congruent with overall conclusions from previous research, noting that there is significant site-specific variability in effectiveness. In a review of the effectiveness of a broad set of rangeland riparian conservation practices, George *et al.* (2011) found that there is a general consensus of benefit associated with distributional practices, but that there was substantial variability in effectiveness reported among studies. Others have more recently reached similar conclusions in reviews of off-stream livestock drinking-water developments (Malan *et al.* 2018) and riparian fencing (O'Callaghan *et al.* 2019) to improve riparian health. Our regression analyses predicted increases in total taxa richness of 53%, 13%, and 4% associated with 1 week (7 person-days) per year of investment in managerial effort for off-stream nutritional supplementation, fence maintenance, and livestock herding, respectively (Fig. 2). The predicted increases for the same practices were 119%, 23%, and 5% for EPT richness. The riparian health returns on managerial effort for off-stream nutritional supplementation and fence maintenance were significant and are worthy of managerial consideration, whereas livestock herding was not significantly correlated with invertebrate richness metrics (Table 4).

We found no significant riparian health benefits associated with livestock distributional practice implementation when we tested the simple question 'do you conduct this practice, yes or no?' in a binary approach common to the scientific study of practice effectiveness. Based on this approach alone, we would have concluded that distributional practices do not improve riparian health, which is counter-intuitive. However, when we consider implementation as a gradient of management effort, we conclude that improvements in riparian health are dependent upon the manager implementing practices effectively at the site level (Table 4, Fig. 2). This finding of returns on wise investments is intuitive, and we propose that it explains at least a portion of the variation in practice effectiveness observed on the ground (e.g. Johnson *et al.* 2016) and across the scientific literature (e.g. Malan *et al.* 2018) when implementation is considered binary, rather than as a gradient. Understanding the linkages between ecological systems and human use and management is key to enhancing both agricultural and environmental outcomes (McAllister 2012; Roche *et al.* 2015*b*); however, the human dimension (e.g. behaviour, attitudes, knowledge, values) of natural resources is too often treated separately from the ecological system, or even completely lacking in environmental assessments (Briske *et al.* 2011; Roche *et al.* 2015*a*). Our results indicate that failing to (*a*) consider managerial behaviours and (*b*) prescribe adequate managerial investment when developing livestock-management strategies to improve riparian health can undermine the value of capital investments in physical tools such as fencing, supplements, and feeding stations.

Finally, we found no significant relationship between stocking rate and riparian health. There is clear evidence in the literature that stocking rate (i.e. grazing pressure) is a primary determinant of environmental and economic outcomes in general (e.g. Briske *et al.* 2011; Byrnes *et al.* 2018), an overall relationship with which we agree. However, it is important to note that these expansive, mountainous grazing units have relatively low stocking densities (mean 49 ha/cow) and stocking rates with more-than-adequate forage at all times during the grazing season. These are scenarios where livestock preferences and landscape elements create uneven use and resulting impacts across grazing units (e.g. Creamer *et al.* 2019). In similar grazing units across the region, Freitas *et al.* (2014) found that implementation of distributional improvements without reduced stocking rates resulted in restoration rates of riparian plant communities equal to rates observed on units with grazing removed. Those authors reported $<5\%$ bare ground in riparian areas, indicative that riparian conservation goals were being met within the grazing management units. In the region, Oles *et al.* (2017) found that overall stocking rates of grazing-management units were not significantly related to riparian plant community restoration rates, but that increased site-specific grazing pressure was negatively correlated with restoration rates. Those authors made the case that reductions in overall stocking rates across this region – which had a policy-driven 30% reduction in AUMs in the decade prior– would not lead to continued site-specific improvements in riparian health. Our results are in agreement with these findings.

Conclusion

We recommend that grazing distributional practices should be employed to safeguard riparian health across large, complex landscapes. However, implementation of these practices alone does not assure the desired response. We found that the greatest riparian health returns to investment of managerial effort were associated with off-stream nutritional supplementation followed by fence maintenance. Livestock herding was not significantly correlated with invertebrate richness metrics. The potential benefits are dependent on expenditure of adaptive management effort tailored to site-specific livestock, ecological, and physical conditions. We found limited implementation of off-stream water sources within this cross-section of management units, and suggest there is opportunity to improve livestock distribution and riparian health with private and public incentive investments to increase implementation of this practice. Finally, these distribution practices should be implemented in conjunction with other grazing best management practices across the management unit, such as appropriate stocking rate (e.g. Briske *et al.* 2011; Hunt *et al.* 2014), appropriate seasons of use (e.g. Jones *et al.* 2009), and monitoring indicators of successful implementation and resource response (e.g. Clary and Webster 1990).

Conflicts of interest

The authors declare no conflicts of interest.

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