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Does Riparian Fencing Protect Stream Water Quality in Cattle-Grazed Lands?

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Abstract
Cattle degrade streams by increasing sediment, nutrient, and fecal bacteria levels. Riparian fencing is one best management practice that may protect water quality within many grazed lands. Here we surveyed the literature and summarized the responses of sediment, nutrient, and fecal indicator bacteria levels to riparian exclosure fencing in cattle-grazed lands. Overall, our review of relevant literature supports the role of riparian exclosure fencing in reducing the negative impact of cattle on water quality, particularly for sediment and fecal indicator bacteria in temperate forest and temperate grassland streams. Establishing buffer widths > 5–10 m appears to increase the likelihood of water quality improvements. Fencing may also be effective at reducing pollutant inputs during stormflows. Our survey also identified critical spatial and thematic gaps that future research programs should address. Despite cattle grazing being prevalent in 12 terrestrial biomes, our systematic search of the empirical literature identified 26 relevant studies across only three biomes. Regions with the greatest cattle populations remain largely unstudied. In addition, we identified inconsistencies in how studies reported information on regional factors, cattle management, and other metrics related to study results. We provide a list of standard parameters for future studies to consider reporting to improve cross-study comparisons of riparian fencing impacts. We also encourage future studies in semi-arid and tropical regions where cattle grazing is common.

Keywords Grazing · Water quality · Exclosure fencing · Sediment · Nutrient · Fecal

Introduction

Agricultural land use and land management practices, particularly within riparian areas, have extensive impacts on water quality (Osborne and Kovacic 1993; Wilcock et al. 2009; Chase et al. 2016). Although agricultural production is responsible for significant sediment and nutrient pollution in many streams (Pimentel et al. 1995; Allan et al. 1997; Dodds and Whiles 2004), riparian buffers can decrease non-point source pollutant loading into adjacent waterways (Lowrance et al. 1997; Aguiar et al. 2015; Pearce and Yates 2017). Several reviews have highlighted widespread effectiveness of riparian buffers in reducing agricultural contaminant loads from crop-producing fields (e.g., Mayer et al. 2007; Hoffmann et al. 2009; Zhang et al. 2010; Sweeney and Newbold 2014). However, we know less about the effectiveness of riparian buffers within cattle-grazed lands. Livestock exclosure fencing may mitigate riparian trampling, in-stream defecation, and streambed disturbance from cattle and thus decrease direct impacts of grazing while also enhancing riparian areas to decrease non-point source pollutant loading from pastures. Fencing is more likely to be used as a best management practice within grazed lands if livestock exclusion from riparian areas consistently protects water quality. However, if livestock exclusion does not protect water quality, other management practices may need to be pursued (Gillespie et al. 2007; Hadrich and Van Winkle 2013). A systematic review on the impact of riparian buffers in grazed watersheds can expand our understanding of the effectiveness of livestock exclosure

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fencing on water quality and identify gaps in current research.

Unrestricted cattle grazing often results in significant alterations to stream channels and riparian areas (Butler 2013), which generally lead to decreased water quality (Trimble 1994; Muller et al. 2016; Malan et al. 2018). Increases in sediment, nutrient (nitrogen and phosphorus), and fecal indicator bacteria (collectively referred to as "SNF" hereafter) in streams are of high concern due to negative consequences to human use as well as to aquatic communities. Cattle can increase SNF levels within streams by excreting fecal matter and urea directly into the water column when loitering within streams (Line et al. 2000; Collins and Rutherford 2004), re-suspending sediment previously deposited on the streambed (Grudzinski et al. 2018), trampling and eroding stream banks (Trimble and Mendel 1995), and increasing runoff inputs from riparian and surrounding grazed watershed areas (Hooda et al. 2000; Butler et al. 2008; Lucci et al. 2010). Cattle grazing within riparian zones, in particular, increases soil compaction and bare ground coverage, thereby increasing SNF availability and transport into streams (Grudzinski et al. 2016; Miller et al. 2016). Increased SNF concentrations can damage streambed habitat used by aquatic biota (Larsen et al. 2011; Sturt et al. 2011), promote eutrophication and increase algal biomass (Dodds et al. 2002; Scrimgeour et al. 2013), alter light availability and dissolved oxygen concentrations (Bierman et al. 1994; Stringfellow et al. 2009; Dymond et al. 2017), and decrease biotic integrity and downstream water quality (Belsky et al. 1999; Bowman et al. 2007). In addition to extensive negative environmental impacts, water contaminated with manure that is consumed by livestock can lead to detrimental health impacts and may decrease overall weight gain and calf survival, thereby decreasing potential production profits (Saker et al. 1999; Willms et al. 2002; Bremner et al. 2016). Best management practices are often encouraged and sometimes regulated in grazed lands to minimize degradation of water quality and to increase herd health.

Fencing is a best management practice for keeping cattle out of riparian and stream environments while allowing continued grazing on the landscape outside of fenced areas (e.g., Flores-Lopez et al. 2010; Sunohara et al. 2012; Bragina et al. 2017). Although excluding cattle from streams can result in increased cattle weight and milk production (Muller et al. 2016), maintenance costs and unpredictable environmental benefits can discourage implementation of exclosure fencing (Bryant et al. 2008). Previous research on the effectiveness of riparian fencing has produced contradictory results (Miller et al. 2010), thereby potentially decreasing land managers' confidence that riparian fencing is a prudent investment.

Sources, processes, and pathways that are conducive to SNF stream inputs vary by parameter (Sunohara et al. 2012). For example, a high proportion of total nitrogen is typically introduced into streams in soluble form (e.g., nitrate), whereas a majority of total phosphorus is typically introduced in particulate form (Logan 1982; Carpenter et al. 1998). Thus, a greater proportion of nitrogen, relative to phosphorus, enters streams via subsurface groundwater flow, whereas phosphorus inputs are increasingly associated with runoff inputs (e.g., Vanni et al. 2001). Surface processes (e.g., overland flow and stream bank-bed trampling) increase fecal indicator bacteria (e.g., *Escherichia coli* or fecal coliform) and sediment transport within streams. Sediment pollution may also bolster bacterial contamination as sediments increase bacteria survival and transport (Bragina et al. 2017). The effects of riparian fencing on water quality could vary by parameter and this may account for some of the discrepancy among studies.

Numerous environmental and management factors may mediate the effectiveness of riparian fencing, including but not limited to; soils, vegetation, climate dynamics, topography, and land use history (Shukla et al. 2011). Thus, influences of riparian fencing from one region may not be indicative of impacts within others (Sunorhara et al. 2012). However, a management practice (e.g., utilization of riparian buffers and/or decreases in stocking rates) could still be effective across regions.

Fencing benefits may also vary between baseflow and stormflow. For example, during storms, riparian fencing may be less effective at mitigating cattle-grazing impacts, as overland flow from outside of fenced areas may transport water containing high SNF concentrations into streams. Thus, hydrograph timing may influence fencing impacts and we consider it here with respect to the efficacy of riparian protection from grazer impacts.

A comprehensive review of previous research on the effects of cattle exclosure fencing in riparian areas is needed by land managers and conservation scientists seeking to protect water quality in grazed lands. In this review, we survey the literature to summarize the responses of sediment, nutrient, and fecal indicator bacteria levels to riparian exclosure fencing in cattle-grazed lands. Water quality responses are also evaluated by hydrograph timing (i.e., baseflow vs stormflow) and land management practices (e.g., riparian buffer width and stocking rate). We also identify spatial and thematic gaps that can be addressed in future research. We hypothesized:

(1) Riparian fencing would be most effective at reducing sediment, phosphorus, and fecal bacteria levels.
(2) Riparian fencing would be most effective at improving water quality at baseflow.

Table 1 Summary of study characteristics

Study	Study area	Biome	Buffer width (m)	Stocking rate (AUM/ha)	Additional management change	Hydrograph timing sampled
Bragina et al. (2017)	Republic of Ireland	TBLMF	1.5 or more	NA	NA	NA (sampled bed sediment)
Bremner et al. (2016)	Nova Scotia	TBLMF	NA	NA	NA	NA
Brenner et al. (1996)	Pennsylvania	TBLMF	NA	NA	Wetland planting	NA
Flores-Lopez et al. (2010)	New York	TBLMF	5	NA	NA	Baseflow and stormflow
Georgakakos et al. (2018)	New York	TBLMF	14	21.1	NA	Baseflow
Hagedorn et al. (1999)	Virginia	TBLMF	NA	NA	Off-stream water provided	NA
Herbst et al. (2012)	California	TCF	NA	NA	NA	NA (sampled bed sediment)
Jackson et al. (2015)	Missouri	TGSS	15	4.4	Controlled burns; removed trees	NA (sampled bed sediment)
Kay et al. (2007)	United Kingdom	TBLMF	NA	6	Off-stream water provided	Baseflow and stormflow
Kay et al. (2018)	United Kingdom	TBLMF	NA	101.5	Off-stream water provided	Baseflow and stormflow
Larson et al. (2016)	Missouri	TGSS	10	1.1	Off-stream water provided	Baseflow
Laubel et al. (2003)	Denmark	TBLMF	0–4.5	NA	NA	Baseflow and stormflow
Line et al. (2000)	North Carolina	TBLMF	10–16	NA	Riparian planting; off-stream water provided; stabilized degraded stream bank; fed hay	Baseflow and stormflow
Line (2003)	North Carolina	TBLMF	10–16	NA	Riparian planting; off-stream water provided; stabilized degraded bank; fed hay	Baseflow
Miller et al. (2010)	Alberta	TGSS	40–80	0.45	Off-stream water provided	NA
Miller et al. (2018)	Alberta	TGSS		3.8	Off-stream water provided	NA (sampled bed sediment)
Muller et al. (2016)	France	TBLMF	1	NA	Off-stream water provided	NA
Olson et al. (2011)	Alberta	TGSS	NA	2.2	Moved winter bedding away from creek; off-stream water provided; implemented rotational grazing	Baseflow and stormflow
Owens et al. (1996)	Ohio	TBLMF	NA	9.5	Fed hay in winter	Stormflow
Pearce and Yates (2017)	Ontario	TBMLF	NA	NA	Manure storage; tile drainage	Baseflow
Shukla et al. (2011)	Florida	TCF	NA	3.3	Off-stream water provided	Baseflow and stormflow
Sovell et al. (2000)	Minnesota	TBLMF	NA	NA	NA	NA
Sunohara et al. (2012)	Ontario	TBLMF	3–5	15	Off-stream water provided; fed hay	Baseflow and stormflow

Table 1 (continued)

Study	Study area	Biome	Buffer width (m)	Stocking rate (AUM/ha)	Additional management change	Hydrograph timing sampled
Zaimes et al. (2008)	Iowa	1/2 TBLMF, 1/2 TGSS	NA	NA	Supplemental feed	NA (sampled bed sediment)
Zaimes and Schultz (2011a)	Iowa	1/2 TBLMF, 1/2 TGSS	NA	NA	Supplemental feed	Baseflow
Zaimes and Schultz (2011b)	Iowa	1/2 TBLMF, 1/2 TGSS	NA	NA	Supplemental feed	NA (sampled bed sediment)

Buffer width refers the distance between the stream and exclosure fencing within the riparian area

TBLMF temperate broadleaf and mixed forests, *TGSS* temperate grasslands, savannas, and shrublands, *TCF* temperate conifer forest, *NA* information not provided or not relevant, *AUM* animal unit month, *ha* hectare

Methods

We completed a Web of Science search in April 2019 to identify peer-reviewed studies that assessed the effectiveness of riparian exclosure fencing within grazed lands on stream sediment, nutrient, and fecal indicator bacteria (e.g., *E. coli*) levels. We structured the Web of Science search as follows:

> cattle or cow or ungulate or livestock or pastoral or grazing AND
>
> creek or stream or river or riparian or waterway AND
>
> fenc* or exclosure or excluded or enclosure AND
>
> nitr* or phos* or nutrient * or sediment* or water quality or *E. coli* or fecal or coliform

The "*" is used to include all word endings following each prefix, for example *nitr** will retrieve "*nitrogen, nitrate*, and/or *nitrite*". The search returned 478 studies ranging in publication date from 1927 to 2019. First, we read the title and abstract of each search result to identify potentially relevant studies. We then read all potentially relevant studies and eliminated papers where research objectives or methodologies did not address the effectiveness of riparian fencing on water quality.

We included studies that examined fencing impacts on SNF levels in the water column as well as from the streambed (e.g., Larson et al. 2016; Chase et al. 2017). Studies which included extensive stream restoration were not included in the review, as the impact of fencing on water quality could not be isolated. However, we did include studies that in addition to riparian fencing, replanted degraded riparian zones, or stabilized stream banks in local areas degraded by extensive trampling, as these changes should occur naturally upon cattle exclusion. Lastly, we also included studies that provided off-stream water sources or access points to streams, as this is a necessary management practice when exclosure fencing is applied. The location of each study was mapped by state in the United States, province in Canada, and country in Europe. Study biomes were grouped according to Olson et al. (2001). Details on each study including location, grazing management, environmental characteristics, and sampling design are summarized in Table 1 and Online Resources 1–3.

We grouped responses of specific water quality parameters (e.g., nitrate and fecal coliform) across the studies into four broader parameter categories: sediment, phosphorus, nitrogen, and fecal indicator bacteria. The *Sediment* category contained total suspended sediment and deposited streambed sediment. The *Phosphorus* category included,

Table 2 Summary of study results

Study	Sediment	Phosphorus	Nitrogen	Fecal Indicator Bacteria	Basis for result classification
Bragina et al. (2017)				*E. coli* = ** (−5.3); Other coliform = X	Overall summary statistic
Brenner et al. (1996)		PO_4 = **		Fecal coliforms = ** (−2.5); Fecal streptococci = X (0.0)	Improvement percentages
Bremner et al. (2016)	TSS = * (−0.14)	TP, SRP = X	TKN, NH_3, NO_3 = X	*E. coli* = * (−0.81)	Summary statistic within each site
Flores-Lopez et al. (2010)		SRP = ** (−0.36)	NO_3 = X		Overall summary statistic
Georgakakos et al. (2018)	TSS = ** (−2.6)	TP = ** (−0.30); SRP = X (1.1)			Overall summary statistic
Hagedorn et al. (1999)				Fecal coliforms = ** (1.0); Fecal streptococci = X	Frequency of significant results
Herbst et al. (2012)	Bed Fines = X (−0.42)				Overall summary statistic
Jackson et al. (2015)	Bed Fines = ** (−0.34)				Overall summary statistic
Kay et al. (2007)				Total coliforms, presumptive *E. coli*, presumptive enterococci = ** (−0.09, −0.07, −0.13)	Overall summary statistics for high flow conditions
Kay et al. (2018)				*E. coli*, intestinal enterococci = ** (−3.9, −3.7)	Based on magnitude of change. No statistical comparison provided
Larson et al. (2016)		TP = * (−0.29), SRP = X	TN, NO_3 = * (−0.85, −0.36)	*E.coli* = * (−2.6)	Overall summary statistic
Laubel et al. (2003)	TSS = **	TP = *			Strength of correlations between bank erosion and TSS, TP transport
Line et al. (2000)	TSS = ** (−1.40)	TP = ** (−1.2)	NO_3 = X (−0.26); TKN = ** (−1.2)		Overall summary statistic
Line 2003	TSS = ** (−1.89)			Fecal coliforms, Enterococci = ** (−2.3, −1.5)	Overall summary statistic
Miller et al. (2010)	TSS = * (0.05)	TP, SRP = * (TP = 0.08)	TN, NH_4 = X; NO_3, = * (TN = 0.0)	*E. coli* = * (0.08); Fecal coliforms = X	Frequency of significant results
Miller et al. (2018)	Bed Fines = ** (−0.3)				Frequency of significant results
Muller et al. (2016)		PO_4 = X (0.52)	NH_4, NO_3 = X (0.02, 0.50)		Overall summary statistic
Olson et al. (2011)	TSS = X (0.059)	TP, TDP = X (0.0, 0.0); PP = ** (−0.058)	TN = ** (−0.19); NO_3, NH_4 = X (0.068, −0.11)	*E. coli* = X (−1.35)	Overall summary statistic
Owens et al. (1996)	TSS = **				Improvement percentages
Pearce and Yates (2017)		TP, TDP, SRP = X	TN, NO_3, NO_2, NH_4 = X		Overall summary statistic
Shukla et al. (2011)		TP = **	TN = *		Frequency of significant results
Sovell et al. (2000)	Bed Fines = X	Orthophos-phate = X	NO_3, Ammonia = X	Fecal coliforms = X	Based on authors' interpretation of PCA; Summary statistic

Table 2 (continued)

Study	Sediment	Phosphorus	Nitrogen	Fecal Indicator Bacteria	Basis for result classification
Sunohara et al. (2012)		TP = X (−0.10); DRP = ** (−0.38)	NO_3 = X (0.017); NH_4 = ** (−0.21)	Total coliforms, *E. coli*, *Enterococcus* spp. = ** (−0.10, 0.06, −0.16); Fecal coliforms = X (−0.43)	Overall summary statistic
Zaimes et al. (2008)		TP = * (−0.11)			Frequency of significant results
Zaimes and Schultz (2011a)	TSS = * (−1.1)	TP, DP = * (0.23, −0.72)			Frequency of significant results
Zaimes and Schultz (2011b)	Bed Fines = * (−1.2)				Frequency of significant results

"Summary statistic" refers to an overall statistical test that determined if water quality was improved due to fencing. Response ratios are shown in parentheses. The lower the response ratio value, the greater the effect of fencing on the measured parameter. "**" signifies "majority improvement," "*" signifies "minority improvement," and "X" signifies "no improvement"

Parameters examined include *TSS* total suspended sediment, *PO_4* phosphate, *TP* total phosphorus, *SRP* soluble reactive phosphorus, *TDP* total dissolved phosphorus, *PP* particulate phosphorus, *DRP* dissolved reactive phosphorus, *DP* dissolved phosphorus, *TKN* total Kjeldahl nitrogen, *NH_3* ammonium, *NO_3* nitrate, TN total nitrogen, *NH_4* ammonia

total phosphorus, particulate phosphorus, soluble reactive phosphorus, and total phosphorus deposited within the streambed. The *Nitrogen* category included, total nitrogen, nitrate, nitrite, ammonium, ammonia, and total Kjeldahl nitrogen. The *Fecal indicator bacteria* category included, *E. coli*, fecal coliform, fecal enterococci, fecal strep, and total coliforms (Table 2). Combining parameters into broader categories may mask some water quality responses to fencing (e.g., potential differences in particulate vs. dissolved nutrient responses). However, given the limited number of studies for some specific parameters, we could not draw conclusions at finer levels.

We classified fencing impact on each specific parameter from each study as generating either a "majority improvement", "minority improvement", or "no improvement" based on the frequency of statistically significant improvements. We classified the result as generating a "majority improvement" if a majority (>50%) of sites and/or temporal periods in a study showed significant improvements for a water quality parameter. We classified fencing as generating a "minority improvement" if statistical tests showed a significant improvement in half or less of the analyses (<50 but >0%). We determined that fencing resulted in "no improvement" if no statistically significant differences were detected due to fencing (0% of statistical tests detected significant water quality improvements). Often, authors provided a summary statistic for each water quality parameter, and thus classifying study results was straightforward. Some studies contained separate statistical tests between sites (upstream vs. downstream or for each stream) and times (seasons or years). Two studies (Brenner et al. 1996; Kay et al. 2018) did not use statistics to assess fencing impacts. For these studies, the authors reported the amount of change in pollutant levels before and after fencing installation. We based our classification on the authors' description and conclusion of fencing impacts.

In addition to the categorical classification, we also calculated response ratios (RRs) (Hedges et al. 1999). RRs allow for comparisons of change in water quality parameters between studies (e.g., Ecke et al. 2017) and have previously been used to examine impacts of agricultural practices (e.g., Zuber and Villamil 2016). We calculated RRs from mean values that studies reported in tables or extracted values with Engauge Digitizer software (https://markummitchell.github.io/engauge-digitizer/) when publications only provided data in figures. RRs for each water quality parameter were calculated as: ln(mean treatment value/mean control value) (Hedges et al. 1999). In order to interpret all the endpoints uniformly, we needed to change the direction in some inequalities (from RR > 0 to RR < 0) so that a negative response ratio indicates an improvement in water quality (decrease in undesirable conditions in fenced treatment relative to grazed treatment). For this adjustment, we multiplied the RR by −1. Study results were classified into the qualitative categories and had RRs calculated, as some studies did not provide sufficient data to calculate RRs. The categorical classification allows for an examination in frequency of improvements, while RRs describe the degree of change in a water quality parameter. We examined statistical relationships between potential covariates (e.g., buffer width) and water quality responses when enough data was available. We calculated a midpoint value when authors reported a range for a covariate (e.g., 40–80 m buffer width in Miller et al. 2010). Spearman's rank correlation for non-parametric data was used to assess relationships between water quality variables and covariates as the data were not normally distributed. Due to limited reporting of many covariates (e.g., cattle breed, manure application, presence–absence of concentrated animal feeding operation) or uniformity across studies (e.g., grazing season), the relationships with covariates on impact to water quality could not be determined beyond stocking rates and riparian buffer widths.

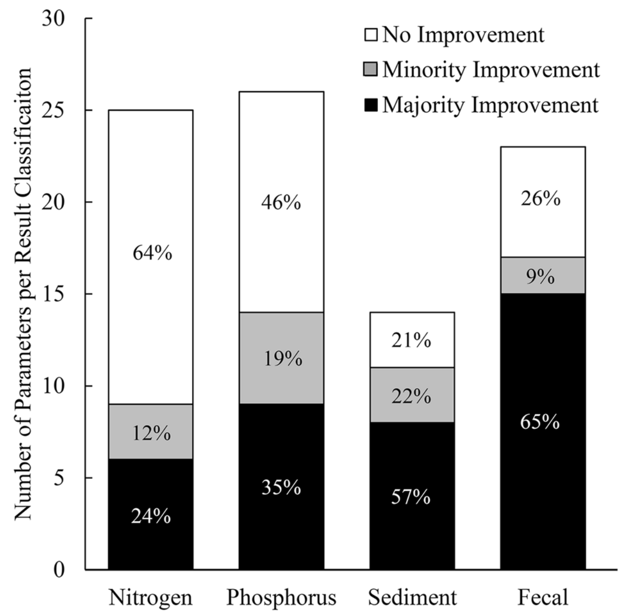

Fig. 1 Summary of study findings grouped by parameter type

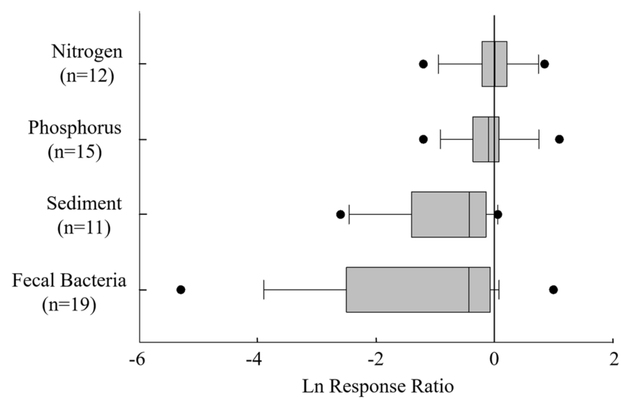

Fig. 2 Response ratios of water quality parameters to riparian exclosure fencing from studies that provided sufficient data for calculation. Lines inside of each box represent median values, outside box boundaries represent 25 and 75th percentiles. Error bars represent 10 and 90th percentiles. Points outside of errors bars represent 5 and 95th percentiles

Results

Twenty-six of the 478 studies identified in the literature search were relevant to this review (Table 1). Fencing impacts were empirically assessed through a variety of study designs, including, pre–post treatment comparisons (e.g., Line et al. 2000), upstream to downstream treatment comparisons (e.g., Miller et al. 2010), and whole watershed comparisons (e.g., Larson et al. 2016). Studies also varied widely in sampling frequency and duration. Some studies sampled multiple times per week (e.g., Kay et al. 2007), while others sampled several times per year (e.g., Zaimes and Schultz 2011a). Most studies sampled water between weekly and monthly time scales. The duration of studies varied from <1 year (e.g., Herbst et al. 2012) to over a decade (e.g., Owens et al. 1996).

Twenty of the 26 studies examined impacts of fencing on water quality (i.e., within the water column) and six studies measured fencing impacts on parameters related to streambed characteristics (Table 1). Of the 26 studies, the impact of fencing on sediment was examined in 14 studies, phosphorus in 16 studies, nitrogen in 11 studies, and fecal indicator bacteria in 12 studies (Table 2). Fifteen studies examined impacts of fencing on more than one of these parameter categories and four studies examined impacts on all four parameter categories. In total, our review examined fencing impacts on 88 water quality responses (mean of 3.4 parameters per study) (Table 2).

From the 26 reviewed studies, 22 (85%) found that fencing was associated with either a "majority improvement" or "minority improvement" in at least one water quality parameter (Table 2). Fencing decreased sediment in 11 of 14 (79%) analyses (8 of 14 majority improvement), phosphorus in 14 of 26 (54%) analyses (9 of 26 majority improvement), nitrogen in 9 of 25 (36%) analyses (6 of 25 majority improvement), and fecal indicator bacteria in 17 of 23 (74%) analyses (15 of 23 majority improvement) (Fig. 1). The largest responses in water quality to riparian exclosure fencing occurred for fecal bacteria (mean RR = −1.25) and sediment parameters (mean RR = −0.84), while nutrients experienced the smallest responses (mean RRs ≥ −0.11) (Fig. 2).

Three of the 12 terrestrial biomes with a mean population density of >1 cattle/km^2 (Fig. 3) had studies related to the

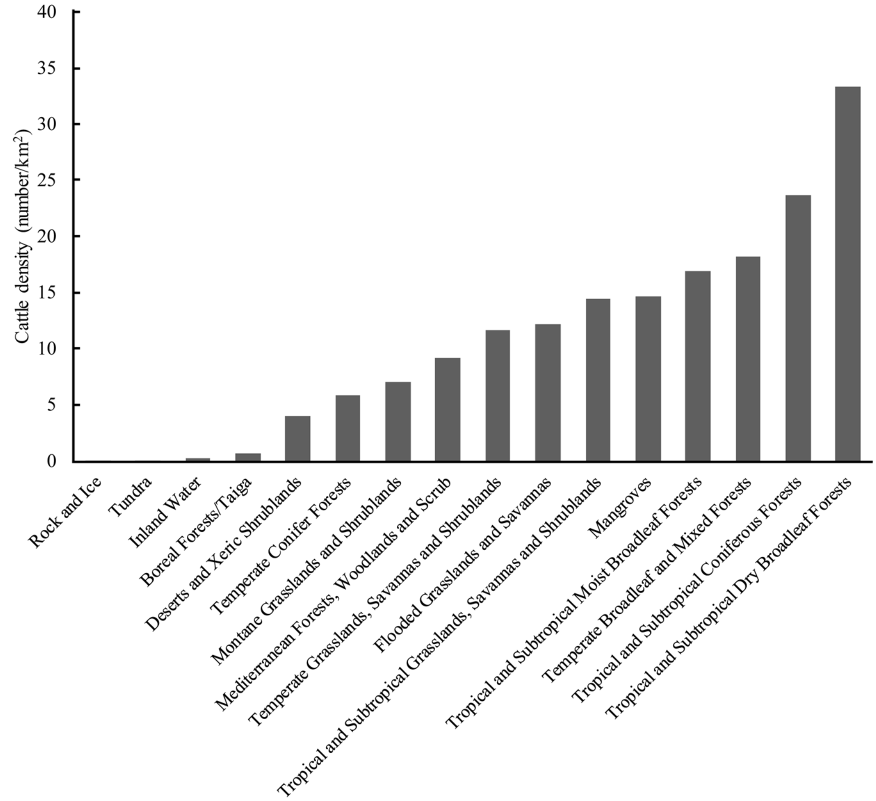

Fig. 3 Mean cattle population density within each biome. Biome boundaries were determined following Olson et al. (2001). Global cattle density data were obtained from the Food and Agricultural Organization of the United Nations (FAO 2014). Cattle densities were calculated by biome within ArcGIS (version 13.3)

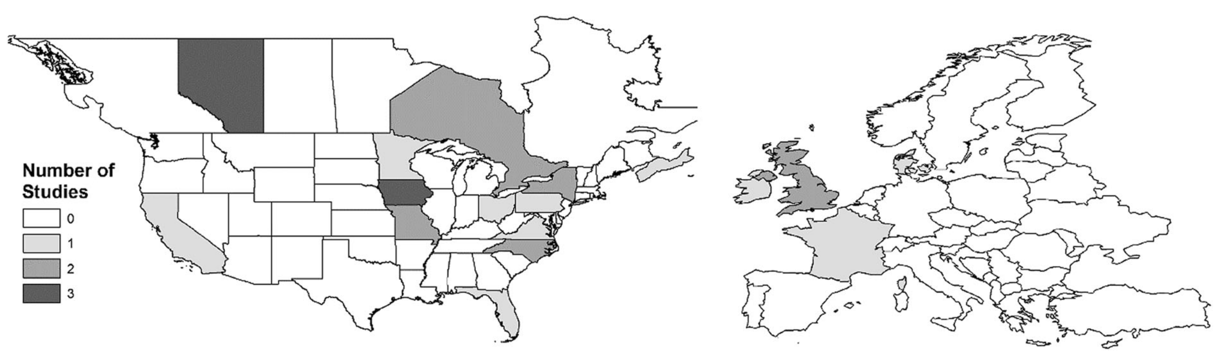

Fig. 4 Spatial distribution of reviewed studies. No studies were identified outside of North America and Western Europe

impact of fencing, including temperate broadleaf and mixed forests (17 studies), temperate grasslands, savannas and shrublands (seven studies), and temperate conifer forests (two studies). Only five studies have been completed outside of North America and these were in Western Europe (France, United Kingdom (two), Denmark, and Ireland) (Fig. 4 and Table 1). Iowa and Alberta had the most published studies with three each (Fig. 4).

Eight of 20 studies sampled the water column during both baseflow and stormflow conditions. Four studies compared the impacts of fencing during baseflow and stormflow conditions with separate statistical analyses (Kay et al. 2007, 2018; Olson et al. 2011; Sunohara et al. 2012), while four did not separate analyses by hydrograph timing (Line et al. 2000; Laubel et al. 2003; Flores-Lopez et al. 2010; Shukla et al. 2011). Between the four studies that separated statistical analyses by hydrograph timing, 21 water quality responses to fencing were tested (Table 3). Ten (48%) of these experienced a "majority improvement" during stormflows (phosphorus $n = 3$; fecal $n = 7$) and none had "minority improvements" (Table 3). The four studies that did not separate statistical analyses by hydrograph timing, examined 10 water quality responses and found a "majority improvement" for six (60%; sediment $n = 2$; phosphorus $n = 3$; nitrogen $n = 1$), a "minority improvement" for two (20%; phosphorus $n = 1$; nitrogen $n = 1$), and

Table 3 Comparison of study findings between baseflow and stormflow conditions with statistical analyses separated by hydrograph timing

Study	Significant impact grouped by hydrograph timing			
	Baseflow	Stormflow	Both	Neither
Kay et al. (2007)		Total coliforms, presumptive *E. coli*, presumptive enterococci		
Kay et al. (2018)			*E. coli*, intestinal enterococci	
Olson et al. (2011)	TN	TP	PP	NO_3, NH_4, *E. coli*, TDP, TSS
Sunohara et al. (2012)	TP, Fecal coliforms, *E. coli*		DRP, Total coliforms, *Enterococcus* spp.	NH_4, NO_3

Within these four studies, all improvements were "majority improvements"

Table 4 Comparison of findings from studies with combined statistical analyses for baseflow and stormflow conditions

Study	Sediment	Phosphorus	Nitrogen
Flores-Lopez et al. (2010)		SRP = **	NO_3 = X
Laubel et al. (2003)	TSS = **	TP = *	
Line et al. (2000)	TSS = **	TP = **	NO_3 = X; TKN = **
Shukla et al. (2011)		TP = **	TN = *

No data were available for fecal indicator bacteria. "**" signifies "majority improvement," "*" signifies "minority improvement," and "X" signifies "no improvement"

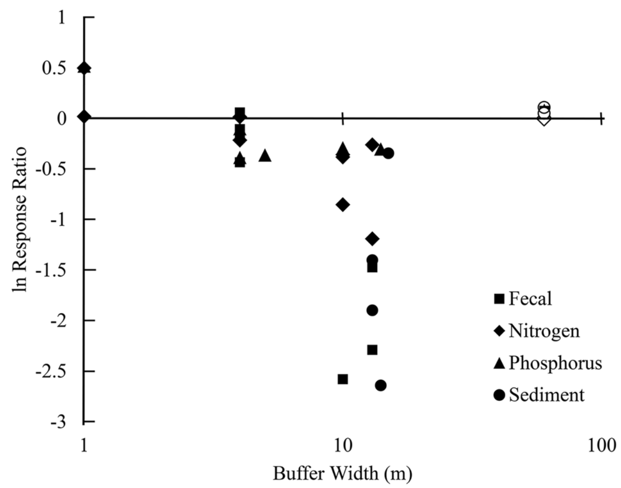

Fig. 5 Relationship between buffer widths and ln response ratios. The open symbols are outliers from a study with a disproportionally large buffer width (Miller et al. 2010). The relationship is statistically significant ($r_s = -0.71$, $p < 0.0001$) when we excluded outliers. The excluded observations had exceptionally high influence and leverage (diagnostic statistics, SAS 9.0 PROC REG) on the relationship because of high riparian buffer width (60 m) relative to the other studies (1–15 m)

"no improvement" for two (20%; nitrogen $n = 2$) (Table 4). Between the eight studies and 31 analyses that contained stormflow sampling, 18 (58%) water quality responses were found to be improved due to fencing. When we exclude nitrogen parameters, improvements occurred in 16 of 22 (73%) remaining water quality responses. Fifteen of these 16 (94%) were "majority improvements".

There are many potential environmental and management related covariates that could influence riparian exclosure fencing impacts on water quality (Table 1, Online Resources 1–3). Studies were inconsistent in reporting some of these characteristics. For example, buffer widths which can be critical for stream health (e.g., Mayer and Canfiled 2008; Yongping et al. 2009), stocking rates (number of cattle head × animal unit equivalents × months/ha; USDA 2009), and sufficient data to calculate RRs were all only reported within five of 26 studies (Miller et al. 2010; Sunohara et al. 2012; Jackson et al. 2015; Larson et al. 2016; Georgakakos et al. 2018). We examined the relationship between RRs across all parameters and riparian buffer width ($n = 31$) and stocking rates ($n = 34$). Overall, a greater riparian width appears to improve water quality ($r_s = -0.71$, $p < 0.0001$) and buffer effectiveness had a marked increase at some sites, but not all, when buffer widths were between 5 and 10 m. With the stepped increase in RR was a concomitant increase in RR variation (Fig. 5). There was no relationship between stocking rate and water quality improvement ($r_s = -0.26$, $p = 0.13$).

Discussion

Most studies (85%) reported reductions of SNF levels in streams due to fencing (Table 2). Establishing buffer widths > 5–10 m appears to increase the likelihood of water quality improvements (Fig. 5). This range of widths is consistent with previous studies documenting buffer widths effective at removal of suspended sediment, fecal bacteria, and nutrients (see review by Fischer and Fischenich 2000, Zhang et al. 2010). With increased buffer widths, the area over which environmental heterogeneity (e.g., slope and vegetation cover) increases and may explain why there was a stepped increase in variation of water quality improvements between 5 and 10 m. The impact of fencing varies by parameter category. Overall, fencing was most effective at decreasing fecal indicator bacteria and sediment parameters followed by phosphorus and nitrogen parameters (Figs 1 and 2). In some study sites, fencing reduced sediment (Zaimes and Schultz 2011a; Georgakakos et al. 2018) and fecal indicator bacteria by 90% (Bremner et al. 2016; Hagedorn et al. 1999; Larson et al. 2016) and > 50% lower levels were common (Owens et al. 1996; Line et al. 2000; Line 2003; Kay et al. 2007). These patterns were not surprising as fencing cattle out of streams will decrease direct defecation and associated fecal inputs into the water column (Kauffman and Krueger 1984; Sherer et al. 1992; Davies-Colley et al. 2004). Eliminating stream bank trampling decreases bank erosion, while mitigating streambed trampling decreases sediment resuspension and downstream transport (Line 2003; Wilson and Everard 2018).

Overall, studies reported less phosphorus reduction, but several studies reported 50–80% lower phosphorus concentrations in response to fencing (e.g., Brenner et al. 1996; Line et al. 2000; Georgakakos et al. 2018; Zaimes et al. 2008). Nitrogen levels benefited the least frequently from fencing and levels were generally only 5–30% lower in studies that identified statistically significant differences (e.g., Olson et al. 2011; Sunohara et al. 2012; Larson et al. 2016). Mayer et al. (2007) found that riparian buffers over 50 m were consistently more effective at removing nitrogen than buffers 0–25 m. We suggest that nutrient concentrations experienced lower frequencies of water quality improvement, in part, due to nitrogen's high solubility and thus greater potential for groundwater inputs or transport during surface runoff (Pitt et al. 1999; Rech et al. 2018). Cattle may alter soil microbial processes leading to losses of gaseous nitrogen forms (N_2O and N_2), but this could vary widely depending upon soil moisture, carbon availability, and soil texture, thus we are not able to further speculate on this. In general, studies found more frequent "no improvement" responses for dissolved nutrients (nitrate and soluble phosphorus) than for total or particulate nutrients (e.g., Line et al. 2000; Olson et al. 2011; Larson et al. 2016; Georgakakos et al. 2018).

We cannot conclude that similar fencing impacts will carry over into biomes that are not well studied. The three temperate biomes with peer-reviewed research on riparian exclosure fencing all contain moderate temperatures and precipitation compared to some other biomes that support extensive grazed landscapes. Impacts of fencing may vary in semi-arid and/or tropical environments due to different hydrologic regimes, runoff patterns, soils and underlying geology, vegetation composition, and riparian-grazing pressure (Dodds et al. 2015). For example, fencing may have greater implications in semi-arid environments, where natural shading is rare outside of riparian zones (e.g., open rangelands in the United States), if cattle are increasingly drawn to stream environments especially during drought conditions (e.g., Allred et al. 2013). In tropical environments with higher discharge and groundwater exchange, impacts of fencing may also vary. Impacts may potentially be greater as fencing may effectively filter inputs associated with frequent runoff events, or to the contrary, fencing may generate less meaningful benefits if discharge sufficiently dilutes grazing impacts (e.g., during in-stream trampling and defecation). Variable hydrologic regimes, such as that occurring in seasonally flooded tropical regions (e.g., the Pantanal in South America), may reset riparian areas between grazing seasons thereby creating unique management dynamics.

Although this review reveals that fencing is promising for improving water quality (Figs 1 and 2), there are many critical spatial gaps where we could find no studies on regions with extensive cattle populations. For example, over half of all studies were completed within the United States (Table 1 and Fig. 4), which represents only 7% of the global cattle population (Robinson et al. 2014; USDA 2018). None of the studies identified by our review were completed in Texas, Nebraska, or Kansas (Fig. 4), the three states with the highest number of cattle within the United States (USDA 2018). No studies examined the efficacy of riparian fencing to mitigate cattle-grazing impacts in Asia or in the southern hemisphere (i.e., South America, Australia, or Africa), which include the three countries with the highest cattle populations (Brazil, India, and China) and are mostly grazed by zebu rather than taurine cattle breeds. Future studies may be particularly beneficial if they are completed in the tropics, which contain three of the four biomes with the highest cattle populations (Fig. 3). In these biomes, moderate to high cattle populations are widespread in the South American, African, and Asian Tropical and Subtropical Moist Broadleaf Forests; South American and Indian Tropical and Subtropical Dry Broadleaf Forests; and South American and African Temperate Grassland, Savanna, and Shrublands. Outside of the tropics, extensive cattle populations also inhabit India's Desert and Xeric Shrubland biome (Fig. 6). Stream responses to riparian fencing also may vary around the world due to unique cattle breeds (i.e., species), each with different physiological requirements, thermal responses, and behavioral characteristics (Zhang et al. 2012; Melletti and Burton 2014; Garrick and Ruvinsky 2015; Gantner et al. 2017). The reviewed studies rarely reported cattle breeds (Online resource 2). Since the biomes and cattle breeds studied thus far do not overlap with the biomes and cattle breeds in many of the understudied regions, it is currently not possible to determine if riparian exclosure fencing would be more or less effective relative to regions examined in this review.

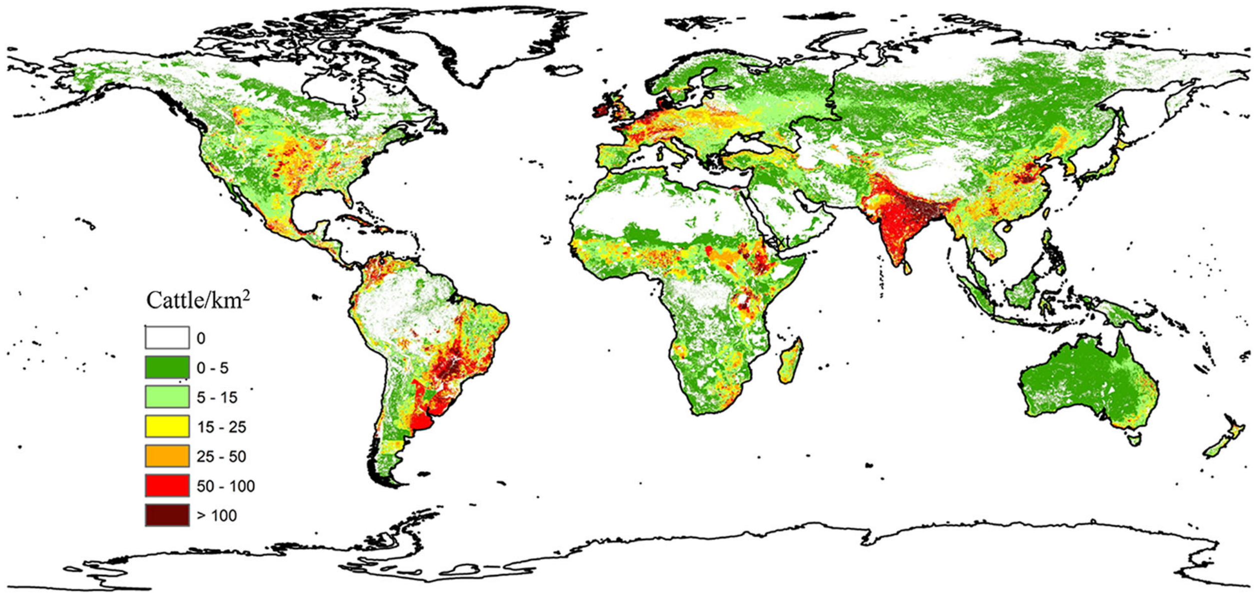

Fig. 6 Global cattle population per kilometer square. Data obtained from the Food and Agricultural Organization of the United Nations (FAO 2014)

Fencing benefits are not always confined to baseflow conditions (Tables 3 and 4). Over half (58%) of the stormflow water quality responses improved with fencing and these were mostly (89%) "majority improvements" (Tables 3 and 4). The four studies that separated baseflow and stormflow analyses found that over half (62%) of the water quality responses were comparably lower with fencing under both baseflow and stormflow conditions (Table 3). As with baseflow, nitrogen parameters appear to benefit the least frequently during storm events (Tables 3 and 4). Although it is promising to see that fencing can benefit water quality during storms, currently there are an insufficient number of studies for us to determine the generality of positive impacts. We identify this as a major gap in the literature. It is important to increase our understanding of fencing impacts during storm events as climate change is altering the timing and amount of precipitation in some heavily grazed regions (e.g., the South American mid-latitudes; African tropics; Australia; India) and this could intensify runoff inputs (IPCC 2014).

Fencing alone may be insufficient for meaningful water quality improvements in watersheds with extensive pollutant inputs upstream of fenced pastures. Several of the studies examined in this review had study reaches that drained landuses other than grazed land (e.g., row crop agriculture; Hagedorn et al. 1999; Laubel et al. 2003), which can produce higher SNF pollution compared to grazed landscapes (e.g., Dodds and Whiles 2004). Concentrated animal feeding operations (CAFOs) commonly use manure spreading on pasture as a management practice which can impact water quality. Although in some studies located within headwater streams it was obvious that CAFOs were not present, there may have been CAFOs upstream of treatment areas in studies conducted in larger watersheds. If CAFOs are present and manure spreading is occurring, the effectiveness of riparian exclosure fencing may be negligible in improving water quality. If meaningful improvements in water quality adjacent to and downstream of fenced pastures are a goal, upstream land use impacts, including the presence of CAFOs, should be evaluated prior to implementation of fencing. In general, when only a small fraction of stream length is fenced within a predominantly cattle-grazed watersheds, watershed level benefits from fencing may be minimal.

Most studies included in this review were short-term and studies which had a before–after study design generally began collecting the "after" samples immediately following cattle exclusion. Only Muller et al. (2016), Georgakakos et al. (2018), and Kay et al. (2018) allowed for a transitional period of >1 year. An instantaneous impact of riparian fencing, if effectively installed, is the elimination of cattle from stream channels (Larson et al. 2016), which should reduce several grazing impacts including, resuspension of streambed material, direct bank erosion from trampling, and in-stream defecation. However, riparian environments may take extended time to respond and recover after implementing fencing (Belsky et al. 1999). For example, the development of mature riparian vegetation can take up to several decades (Linhart and Whelan 1980). As riparian zones mature and subsurface root networks become more established following cattle exclusion, greater decreases in sediment, nutrient, and fecal bacteria inputs are expected.

During storms, additional benefits following longer riparian transition periods may occur if increased biomass creates a more efficient filter during runoff. Long-term studies are needed with respect to riparian fencing. Although sometimes stream restoration efforts attempt "quick-fixes", restoring some streams and their riparian areas may take decades or longer (Wohl et al. 2005). In addition, very few studies monitor long-term ecosystem response following restoration (Bernhardt et al. 2005), thus current studies may not be capturing the full long-term benefits of riparian exclosure fencing. With a better understanding of long-term ecosystem benefits from these restoration efforts, returns on financial investments can be better assessed (Palmer and Filoso 2009).

Designing Future Studies on Riparian Fencing

Many variables can interact with riparian exclosure fencing and impact water quality. In this study, we were unable to evaluate the impact of many potential covariates due to inconsistent reporting of study designs and statistical outputs. The lack of studies in most biomes with cattle-grazing calls for coordinated research across biomes to determine if and how water quality will respond to riparian exclosure fencing at baseflow and stormflow conditions within other biomes. In addition, some studies only provided sufficient statistical details when parameters had a significant response to riparian exclosure fencing (see RRs in Table 2). To improve our understanding of interactions between environmental and management related drivers of water quality, we recommend that future studies include the following details so that cross-study comparisons may be more effectively made: (1) fencing buffer widths, (2) stocking rates, (3) details on when cattle enter and exit pastures (i.e., grazing seasons), (4) additional cattle management details including potential decreases in cattle stocking following installation of riparian exclosure fencing, (5) type of cattle operation (beef, dairy or both), (6) information on geology, (7) soils, (8) vegetation (native or cultivated), (9) upstream land use (including presence of CAFOs), (10) cattle breeds, (11) if and how much manure or fertilizer is applied to pastures, and (12) coordinates for sampling locations. We do not contend that future study designs need to incorporate all of these factors into their designs, it would require an impossible number of study units and organization for all these factors. However, by including such information in field studies, future meta-analyses can begin to identify the key factors that interact most strongly with riparian fencing to affect water quality. Such findings will allow researchers to design field experiments that incorporate key factors to further advance scientific understanding. With comprehensive study descriptions in future research, the relative impacts and interactions of numerous environmental and management driven variables may be determined.

Conclusions

We summarized 26 studies that examined the impact of riparian exclosure fencing on water column or streambed sediment, nutrient, and fecal indicator bacteria levels. Across the three studied temperate biomes, represented by the available literature, most studies identified positive impacts generated by riparian exclosure fencing. Exclosure fencing appears to be particularly effective in reducing sediment and fecal bacteria. We also note that riparian fencing might protect sensitive riparian plant species and vital wildlife habitat in addition to water quality effects. Our review also revealed critical gaps in research on the effectiveness or riparian fencing. In particular, in the United States and on a global scale, areas with the greatest cattle populations remain largely unstudied. With research conducted in so few biomes, it is currently not feasible to compare fencing impacts across biomes. Response of other grazing livestock (e.g., sheep, goats, water buffalo, and bison) and wildlife to riparian fencing is also a large unknown. Future studies conducted over decadal time scales or at previously studied sites may determine if fencing benefits are compounded with time after riparian zones largely recover from cattle grazing. We also currently, lack data to determine how riparian fencing varies across spatial scales (e.g., local vs. watershed).

In sum, riparian fencing practice appears to have great potential for allowing successful livestock production while decreasing negative environmental impacts. It is important to note that riparian exclosure fencing is not the only stream restoration approach that can be implemented in grazed lands to improve environmental condition. Patch-burn grazing, shade structures, alternative water sources, and decreases in stocking rates can be utilized alongside riparian exclosure fencing to further reduce stream degradation. Additional studies are needed to evaluate the economic value added from improved water quality and producer costs of installing riparian exclosure fencing.

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