Prescribed grazing as a secondary impact in a western riparian floodplain

JAMES A. SEDGWICK AND FRITZ L. KNOPF

Abstract

The effect of late-autumn cattle grazing on plant biomass was examined in a western Great Plains cottonwood riparian zone prone to catastrophic flooding every 5–8 years. Following 1 year of pre-treatment data collection in 1982, five 16-ha pastures were grazed from 1982 to 1984 and compared to 5 control pastures within the South Platte River floodplain in northeastern Colorado. At a prescribed grazing level of 0.46 ha/AUM, riparian vegetation proved to be resilient to the impacts of grazing. We detected only a few significant treatment effects for above-ground biomass after succeeding growing seasons. Willows (*Salix* spp.) responded negatively to grazing whereas biomass of prairie cordgrass (*Spartina pectinata* Link) was greater on grazed plots. Yearly changes in above-ground biomass, especially dramatic following a severe flood in 1983, suggest that periodic, catastrophic flooding is a major perturbation to the ecosystem, and in conjunction with our results on grazing impacts, indicate that dormant-season grazing within Soil Conservation Service (SCS) guidelines is a comparatively minor impact within the floodplain. In addition, grazing impacts were probably further mitigated by a major forage supplement of cottonwood leaves which was available at the time of cattle introductions. This local forage supplement ultimately created a lighter grazing treatment than that originally prescribed.

Key Words: community composition, cottonwood leaves, flooding, late-autumn grazing, plant biomass, *Populus sargentii*

Historically, riparian zones have been abused, with the most universal degradation being cattle grazing (Leopold 1974, Platts 1979). Grazing on riparian bottomlands tends to be more exhaustive than on uplands (Roath and Krueger 1982, Platts and Nelson 1985), especially in arid regions where water, shade, and higher forage values occur near streams (Behnke 1979). Grazing cattle within riparian zones can dramatically alter vegetational structure and cover, reduce biomass, and alter plant species composition (Ames 1977, Duff 1979, Knopf and Cannon 1982, Kauffman et al. 1983, and others).

Behnke (1979) asserted that no grazing system has been shown to be effective in protecting riparian vegetation, although more recent studies suggest that light grazing levels (May and Davis 1982), late-season grazing (Kauffman et al. 1983, Sedgwick and Knopf 1987), and certain rest-rotation grazing systems (Kauffman and Krueger 1984) may prove to be acceptable management strategies for riparian zones. From 1982–85, we evaluated the impact of late fall/early winter grazing on a riparian floodplain that had not been grazed for more than 30 years. Our objective was to assess the effects of grazing on above-ground biomass of the shrub-herbaceous vegetation layer. Additionally, 2 other factors, which in combination with grazing had a potentially significant influence on the plant community—periodic severe flooding and cottonwood leaves as a forage supplement—came to light during the course of our study and are discussed relative to grazing impacts.

Materials and Methods

Research was conducted in northeastern Colorado (Logan County) on the 5,908-ha South Platte Wildlife Management Area (SPWMA). The SPWMA includes a 30-km stretch of South Platte River floodplain which was last grazed in 1951. These floodplains are periodically wet or flooded during the growing season and are unsuitable as cropland; however, moderate forage production makes them valuable as grazing lands. Historically, these areas were grazed season long or in early spring/late fall. Today, most grazing within the floodplain on private land occurs from late fall through early spring.

Elevation on the study plots ranges from 1,116 to 1,149 m. Soils along this section of the river are classified as Fluvaquents (Amen et al. 1977) that are poorly drained sediments from alluvial deposits. Surface layers 10–50 cm deep and ranging from loamy sand to clay overlay a base of sand and gravel. The riparian plant community is ca. 500–1,000 m wide and the overstory is dominated by mature, plains cottonwood (*Populus sargentii* Dode) with an average canopy cover of 25%.

The average height of the floodplain, from river's edge to upland boundary, is ca. 1 m above mean river level. Although there is little large scale topographic relief across the floodplain, smaller scale features such as scour channels and intervening hummocks create a complex microtopography. The occurrence and distribution of understory floodplain vegetation is strongly influenced by microtopographic differences in flooding frequency (Bell 1974) and depth to groundwater. As a result, shrub and herbaceous plant assemblages in this floodplain are patchy and remarkably interspersed. The lowest elevation plant assemblages border intermittent meanders or temporary pools and are often dominated by prairie cordgrass (*Spartina pectinata* Link), common reed (*Phragmites australis* (Cav.) Trin. ex Steud.), witchgrass panicum (*Panicum capillare* L.), sedges (*Carex* spp.), and knotweeds (*Polygonom* spp.). Higher elevation assemblages are characterized by cheatgrass brome (*Bromus tectorum* L.), western wheatgrass (*Agropyron smithii* Rydb.), Canada wildrye (*Elymus canadensis* L.), sand dropseed (*Sporobolus cryptandrus* (Torr.) Gray), burning bush (*Kochia iranica* Bornmueller), white sweetclover (*Melilotus alba* Desr.), and sunflower (*Helianthus* spp.). Other common herbaceous species on the area include poison hemlock (*Conium maculatum* L.), American licorice (*Glycyrrhiza lepidota* Pursh), common barnyardgrass (*Echinochloa crusgalli* (L.) Beauv.), lovegrass (*Eragrostis* spp.), and goosefoot (*Chenopodium* spp.). The understory shrub layer is composed primarily of western snowberry (*Symphoricarpos occidentalis* Hook), coyote willow (*Salix exigua* Nutt.), sandbar willow (*S. interior* Rowlee), common poison-ivy (*Toxicodendron rydbergii* (Small) Greene), and Woods rose (*Rosa woodsii* Lindl.). Frost grape (*Vitis vulpina* L.) and Virginia creeper (*Parthenocissus quinquefolia* (L.) Planch.) are common vines.

We established ten 16-ha pastures in the floodplain in 1982. Five of the pastures were selected at random to serve as ungrazed

controls and the remaining 5 were grazed by cattle at a moderate rate of stocking during October and November, 1982-84. We calculated stocking rates based on an estimated annual forage production of 2,240-2,800 kg/ha, and a recommended grazing practice of removing half of the forage annually. Estimating a requirement of 454 kg/cow-calf unit/month, we calculated a stocking rate of 39 AUM's to meet grazing guidelines on the 16-ha plots. We then stocked the pastures at 90% of recommended use (35 AUM's or 0.46 ha/AUM) to assure being within guidelines for the pastures with lowest forage production.

Vegetation data were collected in September of 1982-85 prior to grazing. Pre-treatment data were collected in 1982, and from 1983 to 1985 we evaluated grazing impacts after 1, 2, and 3 seasons of grazing. Fourteen stakes in each pasture, located 100 m apart in grids of varying shapes as dictated by floodplain width, served as permanent sample points.

The criterion we used to evaluate the effects of late-fall grazing was standing biomass of grasses, forbs, and shrubs. At the end of each growing season all standing crop was harvested from within one 0.5-m^2 frame/sample point. Plots were located 5 m from permanent points in a different randomly chosen direction (NW, SW, NE, or SE) each year. Only the current year's growth of shrubs and vines was harvested. All clipped material was bagged by species in paper sacks, air-dried for ca. 60 days, and then oven-dried at 120-150° F for 8 h to assure stabilized weights. Samples were weighed to 0.01 g.

We compared standing biomass between treatments and among years using a 2-way, repeated measures analysis of variance (AOV). We compared total biomass, biomass of forage groups (grasses, forbs, and shrubs), and biomass of dominant species on the study area. The response variable was the difference in biomass, for a given species or group, between each treatment year (1983-85) and the pre-treatment year (1982). Thus, we compared biomass differences between 2 treatments (grazed and ungrazed) and among 3 years (1983-85). Samples from plots were averaged over pastures and pasture means were used in the analysis. Thus, total observations were 30 (biomass differences between 1982 [pre-treatment] and 1983, 1984, and 1985 [post-treatment] = 3 years, times 10 pastures). Duncan's multiple range test was used to separate means where appropriate.

In vitro digestible dry matter of cottonwood leaf samples was determined by the methods of Palmer et al. (1976) and a modification of techniques described by Tilley and Terry (1963). Crude protein was determined by A.O.A.C. (1965) procedures.

Results and Discussion

Standing Biomass

Of the 3 major plant groups sampled, grasses made up most of the standing biomass in 1983-85 (42.7-48.1%), and forbs ranked second (26.6-35.0%). In 1982, forbs produced most of the biomass from our clipped samples (52.5%). Shrubs contributed least to total biomass in all 4 years (14.1-26.2%). Minor components (sedges and vines) produced 0.4-5.8% of the total biomass each year. Total standing biomass was similar on control (2,077 kg/ha) and treatment (2,210 kg/ha) plots before grazing was initiated.

The change in total biomass (from 1982 biomass) was similar for the 2 treatments ($P = 0.15$) but differed across years ($P<0.01$) (Fig. 1). The interaction between years and treatments was not significant ($P = 0.19$). For both control and treated pastures, total biomass declined in 1983 following a severe flood in the spring of that year, increased sharply in 1984, and declined again in 1985. The biomass change in 1984 was significantly greater than change in 1983 or 1985 (Fig. 1).

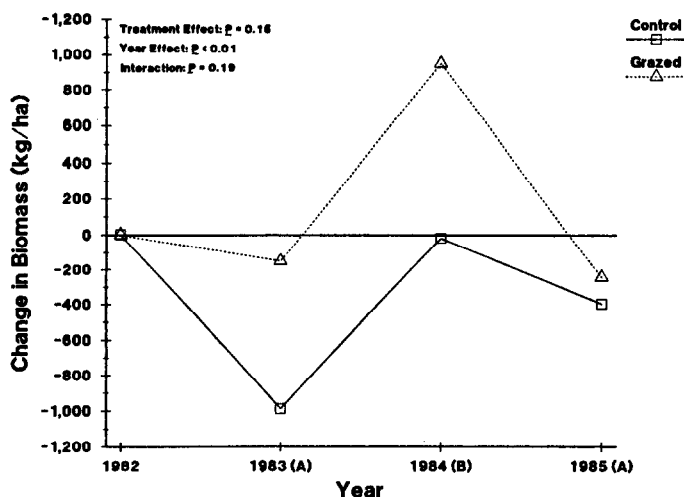Similarly, for all grasses combined (Fig. 2), and for all forbs

Fig. 1. Change in total biomass (kg/ha) in 1983-85 from pre-treatment levels (1982) for control and grazed pastures, South Platte Wildlife Management Area, Colorado. Years sharing the same letter are not significantly different at $P = 0.05$).

Fig. 2. Change in grass biomass (kg/ha) in 1983-85 from pre-treatment levels (1982) for control and grazed pastures, South Platte Wildlife Management Area, Colorado. Years sharing the same letter are not significantly different at $P = 0.05$).
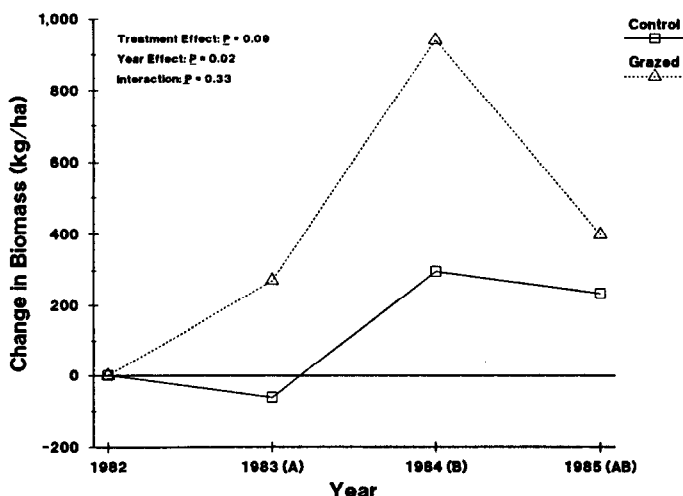
combined (Fig. 3), there was no significant treatment effect but there was a significant year effect. That is, the change in biomass (from 1982 biomass) was similar for grazed and control treatments for both grasses ($P = 0.09$) and forbs ($P = 0.21$), but differed across years (grasses: $P = 0.02$; forbs: $P = 0.02$). For both grasses and forbs, the change in biomass was greater in 1984 than in 1983 (Figs. 2, 3), as both grass and forb biomass increased sharply in 1984, 1 year post-flood. For all shrubs combined, there was neither a significant treatment ($P = 0.09$) nor year ($P = 0.24$) effect for change in biomass from pre-treatment (1982) levels (Fig. 4). Years by treatments interactions were not significant for grasses, forbs, or shrubs (Figs. 2-4).

We examined the effects of grazing on individual species of grasses, forbs, and shrubs which were both common and at least marginally palatable to cattle (U.S. Forest Serv. 1937, Stubbendieck et al. 1981). This included 8 grasses, 4 forbs, and 3 shrubs. Evidence of a grazing effect was significant only for prairie cord-
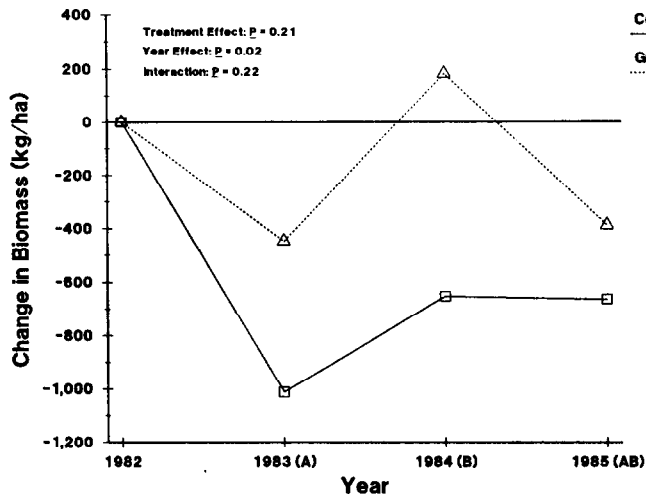
Fig. 3. Change in forb biomass (kg/ha) in 1983–85 from pre-treatment levels (1982) for control and grazed pastures, South Platte Wildlife Management Area, Colorado. Years sharing the same letter are not significantly different at $P = 0.05$).
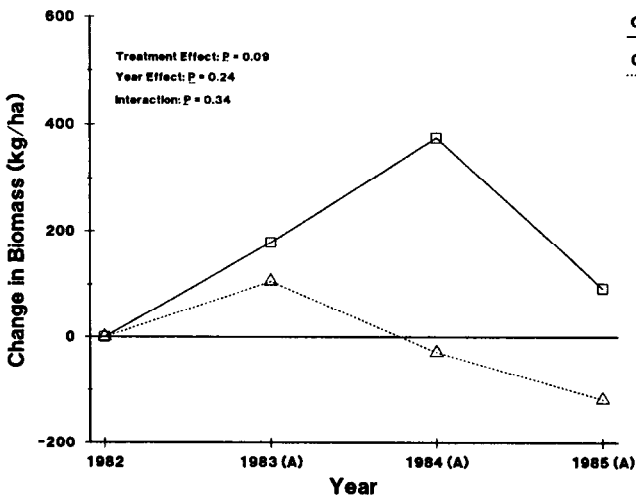

Fig. 4. Change in shrub biomass (kg/ha) in 1983–85 from pre-treatment levels (1982) for control and grazed pastures, South Platte Wildlife Management Area, Colorado. Years sharing the same letter are not significantly different at $P = 0.05$).
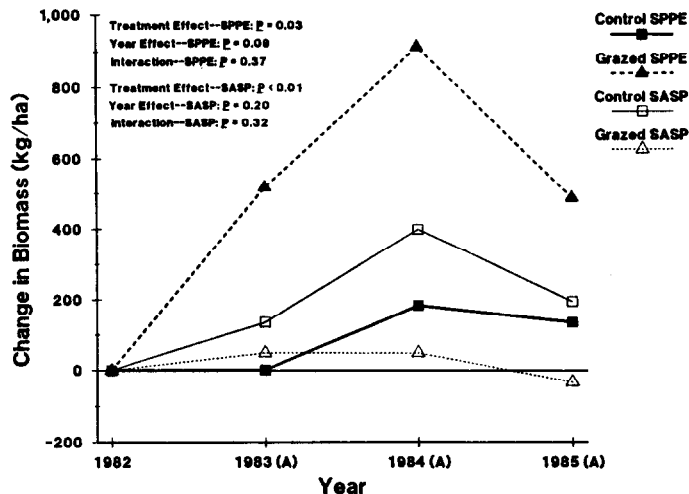

Fig. 5. Change in prairie cordgrass (SPPE) and willow (SASP) biomass (kg/ha) in 1983–85 from pre-treatment levels (1982) for control and grazed pastures, South Platte Wildlife Management Area, Colorado. Years sharing the same letter are not significantly different at $P = 0.05$).

grass ($P = 0.03$) and willows (*S. exigua* and *S. interior* combined) ($P<0.01$). The change in biomass from 1982 biomass was greater on grazed pastures for prairie cordgrass and was less on grazed pastures for willows; thus cordgrass responded positively to grazing and willows responded negatively (Fig. 5). A significant year effect was found for cheatgrass ($P = 0.01$), witchgrass panicum ($P<0.01$), sand dropseed ($P = 0.03$), and sunflower ($P = 0.03$). The change in biomass was less in 1983 than in either 1984 or 1985 for cheatgrass, was greater in 1983 than either 1984 or 1985 for witchgrass panicum, was greater in 1985 than either 1983 or 1984 for sand dropseed, and was greater in 1984 than in 1983 for sunflower. The interactions of years × treatments were insignificant for all individual species ($P>0.05$).

Riparian ecosystems have been reported to be sensitive to most grazing practices tested to date. Yearlong and spring-summer grazing have been shown to be especially detrimental (Kauffman and Krueger 1984). Most studies report decreases in herbaceous biomass due to grazing or increases due to cessation of grazing (Pond 1961, McLean et al. 1963, Gunderson 1968, Duff 1979). Species composition changes can also occur with grazing (Dobson 1973, Hayes 1978). Under prescribed, late-season grazing in the South Platte River floodplain, however, herbaceous and shrub communities appeared to be resilient to grazing by cattle, at least during the initial 3 years after cattle grazing began, following 31 years rest.

Although utilization created a "fenceline contrast" at the perimeter of grazed plots immediately after treatments, standing biomass differences after the succeeding growing season were minor. We detected only a few significant differences between treatments for biomass change from 1982 pre-treatment levels. Only biomass of willows decreased with grazing whereas prairie cordgrass increased on grazed plots. Similarly, Kauffman et al. (1983) reported few changes in standing biomass or in species composition in exclosed vs. late-season-grazed riparian areas in Oregon with a forest canopy.

We attribute the resilience to grazing by cattle in this riparian floodplain to 2 primary factors: (1) initially grazing at proper ("prescribed") levels, and (2) grazing late in the year during the dormant season of plant phenology. SCS grazing prescriptions recommended removing half of the forage annually, which is generally considered to be a "moderate" level of utilization. The correspondence of "proper" or "moderate" grazing with 50% utilization is corroborated by Hayes (1978), for example, who found no differences in riparian species composition when utilization was <60%. Similarly, Platts (1982) found that utilization rates of 65% or more resulted in habitat alteration whereas insignificant alteration occurred with utilization rates of <25%. Furthermore, in our study the season of use and intensity of grazing were strictly controlled. This is untrue of many grazing programs, wherein a riparian zone is included as part of a larger allotment and the utilization of riparian vegetation often exceeds forage use on the upland (Gillen et al. 1985).

Secondly, our program was designed to allow grazing late in the year. Light-to-moderate late-autumn grazing has been proposed as the most acceptable management strategy for riparian zones (Bowers et al. 1979, Platts 1981), and field tests have been promising to date (Kauffman et al. 1983). River levels are low at this time of year and floodplains are relatively dry, thus minimizing the effects of trampling and soil compaction (Rauzi and Hanson 1966).

We caution that even a 4-year study (3 years of grazing) is a relatively short-term study of grazing impacts, and longer term

grazing effects may alter the composition, density, and demography of this forest ecosystem. Cottonwood seedling survival is low on the SPWMA (Sedgwick and Knopf 1989), and because seedlings occurred in so few of our samples, we were unable to assess grazing impacts on cottonwood seedling biomass or composition. Glinski (1977), however, found grazing by cattle reduced cottonwood establishment along Sonoita Creek, Arizona, and predicted that the future width of the riparian zone would be reduced by nearly 60%. Similarly, Kauffman et al. (1983) believed that grazing had a negative impact on woody-dominated community succession patterns and structural diversity. Longer term (>3 years) dormant-season grazing may also result in some herbaceous community changes.

The results of our study reflect the impacts of a late-season grazing strategy. The tolerance of this forested riparian community to grazing probably would not apply under season-long grazing. For example, Crouch (1978) studied the impacts of year-long grazing on the SPWMA and reported dramatic differences in grass, forb, and shrub cover and height between grazed areas and those protected from grazing for 7 to 25 years. Season-long grazing has generally been found by others to be detrimental to the integrity of riparian ecosystems (Ames 1977, Behnke 1979, Duff 1979, Knopf et al. 1988). Our findings support earlier speculations (Knopf and Cannon 1982) that the riparian ecosystems we studied will tolerate moderate late-season grazing without changes in standing biomass.

Cottonwood Leaves as Forage

An additional factor contributing to the resilience of the vegetation and likely mitigating the impacts of the grazing introduction was the availability of cottonwood leaves as a forage resource. We observed cattle foraging on cottonwood leaves during the course of our study, often in apparent preference to grasses, forbs, and shrubs. While we have no utilization estimates for cottonwood leaves, scattered observations over the 4 years of the study suggest more than minimal use of this resource. Cattle selected fallen leaves "almost to the exclusion of the herbaceous or understory vegetation" in the Flint Hills of Kansas (Forwood and Owensby 1985). Whereas fallen leaves have been largely ignored as a forage source for cattle and wild ungulates (Hobbs et al. 1981, Forwood and Owensby 1985) the forest canopy, with its annual leaf fall, produces a potentially valuable forage supplement during the late-season. Hobbs et al. (1981) reported that 32–36% of the bites of tame elk in an aspen (*Populus tremuloides* Michx.) habitat type in winter were (fallen) aspen leaves.

The nutritive value of cottonwood leaves is apparently quite high relative to that of other understory vegetation. Cottonwood leaves in late October averaged higher crude protein (4.9 vs. 2.5%) and in vitro dry matter digestibility (57 vs 39%) than understory vegetation during late October in Kansas (Forwood and Owensby 1985). As a followup to our study, we assessed the nutritive value of cottonwood leaves and their potential value as ruminant forage by collecting samples of newly fallen leaves at 8 randomly selected stakes in late October 1987. We found that cottonwood leaves on the SPWMA averaged 5.3 ± 0.2% crude protein and 55 ± 0.8% digestibility shortly after leaf fall in late October.

In addition to high nutritive value, the biomass of the litter fall of the overstory canopy can be a substantial supplement to the herbaceous biomass. Based on leaf litterfall values for 3 different deciduous forest types in Minnesota (Grigal and McColl 1975), New Hampshire (Gosz et al. 1972), and Sweden (Nihlgard 1972), and for an Illinois bottomland forest (Brown and Peterson 1983), and adjusted for the mean basal area of cottonwoods on our study area ($x = 3.5$ m²/ha; Sedgwick and Knopf 1986), we conservatively estimate that leaf litterfall contributed 285–397 kg/ha or 13–18% of the forage production (2,240 kg/ha) on the SPWMA. Assuming utilization was similar for "typical" forage (grasses, forbs, and shrubs) and cottonwood leaf forage, then grazing pressure on the "typical" forage species was reduced correspondingly. It seems likely that the cottonwood forage supplement effectively reduced our grazing intensity below the prescription, and we conclude that cottonwood floodplains are more resilient to grazing in late autumn due to this major forage supplement of leaves.

Flooding as a Disruptive Influence

Severe overbank flooding inundated the study area in the spring of 1983, after 1 season of grazing. Flooding began in early May and continued into early July, totally immersing >95% of the floodplain (Knopf and Sedgwick 1987). The study area was flooded to a depth of about 1 m for >30 days. In contrast, no overbank flooding occurred in 1984, and in 1982 and 1985 normal runoff resulted in waters receding into the river channels by late May. As a consequence of the 1983 flood, litter, seeds, and plant biomass were silted over or flushed downstream and sections of the floodplain were scoured clean. Irregular aggradation and degradation raised and lowered sections of the floodplain in the study area by >30 cm in some areas.

Such severe floods can have a disruptive influence on plant community structure (White 1979). In our study, we found significant year effects for change in biomass from pretreatment levels for total grass, total forb, and total plant biomass, as well as for 4 individual species. These dramatic year-to-year shifts in biomass, coinciding with sharp contrasts in spring runoff, suggest the dominance of environmental factors and point to floods as a disruptive influence.

Chaneton et al. (1988) noted that severe flooding of an Argentinean floodplain was an overriding environmental determinant of plant species composition compared to grazing. In that study, forb cover decreased, whereas the cover of some grasses and sedges increased in response to severe flooding. Similarly, in our study, biomass estimates of all 4 major forbs decreased on the control area while those of 5 of 8 grasses increased between non-flood (1982) and flood (1983) years. In contrast, when comparing changes in biomass between 1984 and 1985 (years without significant flooding), 3 of those 4 forbs increased, and 6 of the 8 grasses decreased. A test of the association of the changes (+ or –) in biomass of forbs and grasses between (a) 2 years which differed in flood dynamics (1982 and 1983) and (b) 2 years which did not differ in flood dynamics (1984 and 1985) was significant (Fisher's Exact Test, $P = 0.03$). Furthermore, when the ranks of percent changes in biomass were compared for the 8 grasses and 4 forbs between (a) 1982 and 1983, and (b) 1984 and 1985 (as above), the correlation was significantly negative (Spearman correlation = –0.65, $P = 0.02$) indicating that changes in biomass generally reversed themselves. That is, biomass of plant species which increased from 1982 to 1983 (no flood to flood), generally decreased from 1984 to 1985 (flood to no flood) and vice versa. These reversals in biomass change on the control area are indicative of the disruptive influence of flooding. Periodic, major floods and the magnitude of flood impacts on plant biomass and species composition relegate proper grazing to a relatively minor perturbation. Such floods are historically typical within riparian communities along major rivers of the western Great Plains, and still occur on the South Platte River due to the absence of in-stream impoundments (Knopf and Scott 1990).

Literature Cited

Amen, A.E., D.L. Anderson, T.J. Hughs, and T.J. Weber. 1977. Soil survey of Logan County, Colorado. USDA-SCS and Colo. Agr. Exp. Sta., Washington, DC.

Ames, C.R. 1977. Wildlife conflicts in riparian management: grazing, p. 49-51. *In:* R.R. Johnson and D.A. Jones (tech. coords.), Importance, preservation and management of riparian habitat. USDA Forest Serv. Gen. Tech. Rep. RM-43.

A.O.A.C. 1965. Official methods of analysis, 10th ed. Assoc. Off. Agr. Chemists, Washington, DC.

Behnke, R.J. 1979. Values and protection of riparian ecosystems, p. 164-167. In: G.A. Swanson (tech. coord.), The mitigation symposium: national workshop in mitigating losses of fish and wildlife habitats. USDA Forest Serv. Gen. Tech. Rep. RM-65.

Bell, D.T. 1974. Studies on the ecology of a streamside forest: composition and distribution of vegetation beneath the tree canopy. Bull. Torrey Bot. Club 101:14-20.

Bowers, W.B., B. Hosford, A. Oakley, and C. Bond. 1979. Wildlife habitats in managed rangelands—the Great Basin of southeastern Oregon. Native Trout. USDA Forest Serv. Gen. Tech. Rep. PNW-84.

Brown, S., and D.L. Peterson. 1983. Structural characteristics and biomass production of two Illinois bottomland forests. Amer. Midl. Natur. 110:107-117.

Chaneton, E.J., J.M. Facelli, and R.J.C. Leon. 1988. Floristic changes induced by flooding on grazed and ungrazed lowland grasslands in Argentina. J. Range Manage. 41:495-499.

Crouch, G.L. 1978. Effect of protection from livestock on bottomland wildlife habitat in northeastern Colorado, p. 118-125. In: W.D. Graul and S.J. Bissell (tech. coords.), Lowland river and stream habitat in Colorado: a symposium. Colo. Chap. Wildl. Soc. and Colo. Audubon Counc., Greeley, Colo.

Dobson, A.T. 1973. Changes in the structure of a riparian community as the result of grazing. Proc. New Zealand Ecol. Soc. 20:58-64.

Duff, D.A. 1979. Riparian habitat recovery on Big Creek, Rich County, Utah, p. 91-92. In: Proc., forum-grazing and riparian/stream ecosystems. Trout Unlimited, Inc., Denver, Colo.

Forwood, J.R., and C.E. Owensby. 1985. Nutritive value of tree leaves in the Kansas Flint Hills. J. Range Manage. 38:61-64.

Gillen, R.L., W.C. Kreuger, and R.F. Miller. 1985. Cattle use of riparian meadows in the Blue Mountains of northeastern Oregon. J. Range Manage. 38:205-209.

Glinski, R.L. 1977. Regeneration and distribution of sycamore and cottonwood trees along Sonoita Creek, Santa Cruz County, Arizona, p. 116-123. In: R.R. Johnson and D.A. Jones (tech. coords.), Importance, preservation and management of riparian habitat: a symposium. USDA Forest Serv. Gen. Tech. Rep. RM-43.

Gosz, J.R., G.E. Likens, and F.H. Bormann. 1972. Nutrient content of litter fall on the Hubbard Brook experimental forest, New Hampshire. Ecology 53:769-784.

Grigal, D.F., and J.G. McColl. 1975. Litter fall after wildfire in virgin forests of northeastern Minnesota. Can. J. Forest. Res. 5:655-661.

Gunderson, D.R. 1968. Floodplain use related to stream morphology and fish populations. J. Wildl. Manage. 32:507-514.

Hayes, F.A. 1978. Streambank and meadow condition in relation to livestock grazing in mountain meadows of central Idaho. M.S. Thesis, Univ. Idaho.

Hobbs, N.T., D.L. Baker, J.E. Ellis, and D.M. Swift. 1981. Composition and quality of elk winter diets in Colorado. J. Wildl. Manage. 45:156-171.

Kauffman, J.B., and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications...a review. J. Range Manage. 37:430-438.

Kauffman, J.B., W.C. Krueger, and M. Vavra. 1983. Effects of late season cattle grazing on riparian plant communities. J. Range Manage. 36:685-691.

Knopf, F.L., and R.W. Cannon. 1982. Structural resilience of a willow riparian community to changes in grazing practices, p. 198-207. In: J.M. Peek and P.D. Dalke (eds), Wildlife-livestock relationships symposium: Proc. 10. Univ. Idaho, Forest., Wildl. and Range Exp. Sta., Moscow.

Knopf, F.L., and M.L. Scott. 1990. Altered flows and created landscapes in the Platte River headquarters, 1840-1990, p. 52-74. In: J.M. Sweeney (ed.), Management of dynamic ecosystems. The Wildlife Soc., West Lafayette, Ind.

Knopf, F.L., and J.A. Sedgwick. 1987. Latent population responses of summer birds to a catastrophic, climatological event. Condor 89:869-873.

Knopf, F.L., J.A. Sedgwick, and R.W. Cannon. 1988. Guild structure of a riparian avifauna relative to seasonal cattle grazing. J. Wildl. Manage. 52:280-290.

Leopold, A.S. 1974. Ecosystem deterioration under multiple use, p. 96-98. In: Proc. Wild trout management symposium. U.S. Fish and Wildl. Serv. and Trout Unlimited, Denver, Colo.

May, B., and B. Davis. 1982. Practices for livestock grazing and aquatic habitat protection on western rangelands, p. 271-278. In: Wildlife-livestock relationships symposium: Proc. 10. Univ. Idaho Forest, Wildl. and Range Exp. Sta., Moscow.

McLean, A., H.H. Nicholson, and A.L. Van Ryswyk. 1963. Growth, productivity and chemical composition of a sub-alpine meadow in interior British Columbia. J. Range Manage. 16:235-240.

Nihlgard, B. 1972. Plant biomass, primary production and distribution of chemical elements in a beech and a planted spruce forest in south Sweden. Oikos 23:69-81.

Palmer, W.L., R.L. Cowan, and A.P. Ammann. 1976. Effect of inoculum source on in vitro digestion of deer foods. J. Wildl. Manage. 40:301-307.

Platts, W.S. 1979. Livestock grazing and riparian/stream ecosystems-an overview, p. 39-45. In: O.B. Cope (ed.), Proc. forum-grazing and riparian/stream ecosystems. Trout Unlimited, Inc., Denver, Colo.

Platts, W.S. 1981. Effects of livestock grazing. In: W.R. Meehan (tech. ed.), Influence of forest and rangeland management on anadramous fish habitat in western North America. USDA Forest Serv. Gen. Tech. Rep. PNW-124.

Platts, W.S. 1982. Sheep and cattle grazing strategies on riparian-stream environments. p. 251-270. In: J.M. Peek and P.D. Dalke (eds.), Wildlife-livestock relations symposium: Proc. 10. Univ. Idaho, For. Wildl, and Range Exp. Sta., Moscow.

Platts, W.S., and R.L. Nelson. 1985. Streamside and upland vegetation use by cattle. Rangelands 7:5-7.

Pond, F.W. 1961. Effect of three intensities of clipping on the density and production of meadow vegetation. J. Range Manage. 14:34-38.

Rauzi, F., and C.L. Hanson. 1966. Water intake and runoff as affected by intensity of grazing. J. Range Manage. 19:351-356.

Roath, L.R., and W.C. Krueger. 1982. Cattle grazing and behavior on a forested range. J. Range Manage. 35:332-338.

Sedgwick, J.A., and F.L. Knopf. 1986. Cavity-nesting birds and the cavity-tree resource in plains cottonwood bottomlands. J. Wildl. Manage. 50:247-252.

Sedgwick, J.A., and F.L. Knopf. 1987. Breeding bird response to cattle grazing of a cottonwood bottomland. J. Wildl. Manage. 51:230-237.

Sedgwick, J.A., and F.L. Knopf. 1989. Demography, regeneration, and future projections for a bottomland community, p. 249-266. In: R.R. Sharitz and J.W. Gibbons (eds), Freshwater wetlands and wildlife. DOE Symposium Series No. 61, USDOE Office of Scientific and Tech. Inf., Oak Ridge, Tenn.

Stubbendieck, J., S.L. Hatch, and K.J. Kjar. 1981. North American range plants. Univ. Nebraska Press, Lincoln.

Tilley, J.M.A., and R.A. Terry. 1963. A two stage technique for the in vitro digestion of forage crops. J. Brit. Grassl. Soc. 18:104-111.

U.S. Forest Serv. 1937. Range plant handbook. U.S. Government Printing Office, Washington, DC.

White, P.S. 1979. Pattern, process, and natural disturbance in vegetation. Bot. Rev. 45:229-299.