Soil retention capability of Deschampsia caespitosa, Phalaris arundinacea, and Poa pratensis upon exposure to flowing water
by Curt Calvin Strobel

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Land Rehabilitation
Montana State University

Abstract:
The health of riparian systems and establishment and development of riparian vegetation depends on streambank stability. Streambank stability refers to a bank's resistance to change and its resilience after change. It is a function of the soil composition of the bank itself and the type, amount, and vigor of vegetative cover, j" "streambank rehabilitation_is best accomplished by planting_jveg.eta-tian to stabilize and protect the soil, rather than using rigid lifeless materials. Because many species of grasses establish more rapidly than woody vegetation, they may play an integral role in stabilizing streambanks.

The goal of this study was to determine the potential of tufted hairgrass (Deschampsia caespitosa (L.) Beauv.) (DECA), reed canarygrass CPhalaris arundinacea L.) (PHAR), and Kentucky bluegrass CPoa pratensis L.) (POPR) to, stabilize degraded streambanks.

A stream flow simulator (SFS) was constructed to evaluate the ability of DECA, PHAR, POPR to retain soil in the root mass while exposed to moving water.

Forty-eight samples of each species were grown in a greenhouse for 120 days after which they were tested for soil retention capability. Each species was tested at bankfull and 65% below bankfull levels.

There was no significant difference in mean soil loss between DECA and PHAR or between PHAR and POPR, however, there was a significant difference in mean soil loss between DECA and POPR. There was no significant difference in mean soil loss between samples tested at bankfull flow compared to samples tested at below bankfull flow. Complete soil loss was observed in all unvegetated controls at both water levels which occurred within five minutes after exposure to moving water.

Rapid growing rhizomatous grasses seeded immediately after peak runoff subsidence are capable of stabilizing degraded streambanks so that slower growing woody vegetation can establish. Further testing may provide a basis for selection of grass species for use in streambank rehabilitation projects.

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Curt Calvin Strobel

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

26 March 92 — Clayton B Marlow
Date — Co-chairperson, Graduate Committee

2 April 92 — Frank F. Munshower
Date — Co-chairperson, Graduate Committee

Approved for the Major Department

3/26/92 — L. C. Gagnon
Date — Head, Major Department

Approved for the College of Graduate Studies

April 8, 1992 — Henry L Parsons
Date — Graduate Dean

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ABSTRACT

The health of riparian systems and establishment and development of riparian vegetation depends on streambank stability. Streambank stability refers to a bank's resistance to change and its resilience after change. It is a function of the soil composition of the bank itself and the type, amount, and vigor of vegetative cover. Streambank rehabilitation is best accomplished by planting vegetation to stabilize and protect the soil, rather than using rigid lifeless materials. Because many species of grasses establish more rapidly than woody vegetation, they may play an integral role in stabilizing streambanks.

The goal of this study was to determine the potential of tufted hairgrass (*Deschampsia caespitosa* (L.) Beauv.) (DECA), reed canarygrass (*Phalaris arundinacea* L.) (PHAR), and Kentucky bluegrass (*Poa pratensis* L.) (POPR) to stabilize degraded streambanks.

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There was no significant difference in mean soil loss between DECA and PHAR or between PHAR and POPR, however, there was a significant difference in mean soil loss between DECA and POPR. There was no significant difference in mean soil loss between samples tested at bankfull flow compared to samples tested at below bankfull flow. Complete soil loss was observed in all unvegetated controls at both water levels which occurred within five minutes after exposure to moving water.

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INTRODUCTION

It is estimated that 70 to 90 percent of riparian ecosystems that were once present in the United States have disappeared (Council on Environmental Quality 1978). By 1988 the U.S. General Accounting Office's study of streamside management on public rangelands showed that in some states as much as 90 percent of federally managed streams were in a degraded condition. Continuing degradation has been attributed to industrial waste pollution, livestock grazing, farming, logging, mining, and urban development (Hunter 1991). As a result of disturbances to natural riparian ecosystems, an interest in conservation and rehabilitation of existing riparian zones has developed.

The health of riparian systems and establishment and development of riparian vegetation depends on streambank stability (DeBano and Heede 1987). Streambank stability refers to a bank's resistance to change and its resilience after change. It is a function of the soil composition of the bank itself and the type, amount, and vigor of vegetative cover (Bohn 1986). Rehabilitation of degraded stream channels has traditionally addressed bank stabilization through civil engineering methods and techniques. This may be due to the lack of understanding by many civil engineers of the benefits provided by established plants.

The application and effectiveness of conventional engineering structures such as gabions, revetment, riprap, flow deflectors, and flow dividers are well known. However, Kohnke and Boller (1989) noted that these static structures are inflexible when stressed, cost prohibitive, and aesthetically displeasing. In addition, they fail to remove excess streambank soil moisture and cannot heal themselves when damaged. The most important fact to consider is that the use of static structures results in an unnatural stationary stream channel. Streambank erosion, channel meandering, bank sloughing, and flooding are natural processes (Lines et al. 1978) that produce a dynamic equilibrium within the stream ecosystem.

Schiechtl (1980) suggested that we must learn to protect our environment by using nature as a working partner. Natural means of stabilization or bioengineering, places minimum reliance on mechanical or structural techniques, and maximum reliance on nature, aided by mankind (Lines et al. 1978). From this perspective, streambank rehabilitation is best accomplished by planting grasses, forbs, shrubs, and trees to stabilize and protect the soil, rather than using rigid lifeless materials. Furthermore, it is often less costly and easier to obtain immediate benefits through revegetation than through channel changes using artificial stream structures (Platts 1983). Vegetation protects streambanks by reducing the erosive energy of water, by trapping sediment that aids in maintaining the streambank, and by protecting the bank from damage by ice or debris flows and animal trampling (Platts

1983; Jackson and VanHaveren 1984). Large shrubs and trees with their extensive root systems provide bank stability. However, because many species of grasses establish more rapidly than woody vegetation, grasses may play an integral role in stabilizing streambanks. Grasses not only filter out sediment, reduce overland flow, and trap nutrients, but they also reduce the erosive effect of water long enough to allow woody vegetative species to become established.

The goal of this study was to evaluate the ability of tufted hairgrass (Deschampsia caespitosa (L.) Beauv.), reed canarygrass (Phalaris arundinacea L.), and Kentucky bluegrass (Poa pratensis L.) to retain soil in the root mass while exposed to moving water.

LITERATURE REVIEW

Riparian Degradation

Accelerated erosion in stream systems can often be attributed to removal of vegetation (Lines et al. 1978; Monsen 1983; VanHaveren and Jackson 1986) through a variety of actions including concentrated livestock grazing, timber harvesting, and road construction (VanHaveren and Jackson 1986; DeBano and Heede 1987; Clifton 1989). Further degradation occurs from ice scouring, floods, rodent tunneling, gullying (Altpeter 1944), channelization, burning or spraying, urban development, and/or upper watershed modification (Lines et al. 1978). Such disturbances result in changes in vegetative composition and reductions in vegetative cover that can directly alter the structural integrity of streambanks and floodplains (VanHaveren and Jackson 1986).

Effects of Riparian Vegetation on Streambank Stability

Riparian vegetation is critical for reducing bank erosion and lateral channel migration (Clifton 1989). Vegetation is also essential for building and maintaining stream structure conducive to productive aquatic habitat (Platts 1983). Reclaiming degraded streams to support mature riparian communities composed of woody and herbaceous plants depends on rapid vegetative stabilization of deposited sediment (Skinner et al. 1985). Riparian vegetation supplies the floodplain with this protective stabilizing blanket (Altpeter 1944;

Platts 1983; Elmore 1985). Willows, grasses, sedges, and rushes reduce streamflow velocity causing sediment deposition (Platts 1983; Elmore 1985) into the vegetative blanket thereby contributing nutrients to streambank soil and increasing plant production and vigor as well as floodplain fertility. Vegetation overhanging the streambank also helps reduce flow velocity during floods. This leads to further sediment deposition and retention in the floodplain (Hunter 1991).

Riparian zones altered by widened channels, frequent channel realignments, and poorly vegetated banks and floodplains can be rehabilitated by reestablishing plants in the streamside zone (VanHaveren and Jackson 1986). Such vegetation protects streambanks (Monsen 1983; Jackson and VanHaveren 1984; Beschta and Platts 1986) by reducing the erosive energy of water (Li and Shen 1973; Platts 1983) and by trapping sediment (Schumm 1963; Andrews 1982; Platts and Rinne 1985).

Importance of Sedimentation in Streambank Formation

During floods streamside vegetation reduces water velocity and allows bank building through deposition of sediment (Elmore and Beschta 1987, 1988). Because sediment deposition is normally greater on the convex (inside) sections of banks than on the concave ones, revegetation of the convex bank (usually a point bar) causes sediment deposition which builds the bank to a height closely matching that of the eroding concave bank (Altpeter 1944; Platts and Rinne 1985).

In order for the banks of a stream channel to be maintained at a constant width over time, the rate of bank material erosion must be balanced by the rate of deposition (Andrews 1982).

While rapid deposition of sediment is usually associated with low growing forms of woody vegetation, a cover of grasses often serves the same purpose (Altpeter 1944). The filtering effect of riparian vegetation is partly responsible for deposits of fertile soils on many floodplains such as mountain meadows (Swanson 1989). Sediment filtration results in aggradation of the stream bed and banks (Clifton 1989). Once riparian vegetation becomes established, it not only traps nutrient-rich sediment but also acts as a "sink" for nutrients and sediment discharged from the surrounding ecosystem (DeBano and Heede 1987). Uptake of nutrients and stabilization of the stream channel by riparian vegetation improves the quality of water leaving riparian zones (Schumm 1963; Andrews 1982; DeBano and Heede 1987).

Soils and Streambank Stability

Streambank stability is further controlled by the composition of bank material, including both vegetation and sediment (Smith 1976). Differences in soil type are likely to account for differences in inherent bank stability. Streambanks with xeric soil types, composed typically of cohesionless sand and gravel, are much less stable than banks comprised of mesic or hydric types. Hydric types, mainly composed of silt and clay, are associated with the most stable banks (Hackley 1989). Soil profiles on streambanks vary greatly. In some cases the soil material is homogeneous from the top of the bank to the bed of the stream channel. However, there can be considerable variation within the profile, representing changes in behavior and deposition by the stream throughout long periods of time. This will result in various degrees of bank stability. Heterogeneous streambank deposits may result in unstable bank conditions due to differing degrees of substrate cohesiveness. Planting vegetation on streambanks consisting of non-cohesive bank materials can provide stability and prevent bank collapse (Altpeter 1944).

Role of Roots in Stabilization

Severe bank erosion along impacted streams can be explained in part by the lack or absence of root systems which aid in soil stabilization (Groeneveld and Griepentrog 1985). The diversity of vegetative growth forms provided by trees, shrubs, sedges, forbs, and grasses, produces a variety of root networks capable of stabilizing deposited sediment. These root networks are especially important on unconsolidated alluvium (Elmore 1985; Elmore and Beschta 1987,1988). Large root masses reinforce deposited sediment by increasing tensile and shear strength of the bank soil mass (Groeneveld and Griepentrog 1985). Erosion rates drop with increases in root mass. Bank sediment deposits with 16 to 18 percent by volume of roots with a 5 cm root mat for bank protection had 20,000 times more resistance to erosion than comparable bank sediment without vegetation (Smith 1976). Dense bank vegetation reduces undercutting and helps build banks so channels typically become narrower and deeper where once they were wide and shallow (Elmore 1985; Elmore and Beschta 1987, 1988; Clifton 1989; Hunter 1991).

Establishment and Survival of Woody Species

Establishment of woody plants is critical in rehabilitating riparian areas (Volny 1984). Woody species often provide local channel stability and resistance to channel erosion allowing sedges, rushes, grasses, and forbs to establish (Elmore and Beschta 1987, 1988). Willows continue to be the woody plant of choice because they are usually readily available, easily established, grow rapidly, and provide more stability to the site than other woody plants (Monsen 1983; Schultze and Wilcox 1985). Thousands of cuttings have been established in Oregon, Washington, and California and are proving effective for bank stabilization (Lines et al. 1978; Schultze and Wilcox 1985); however, there have been failures or poor success with willow plantings. In California during 1978-80 and 1982, the leading cause of willow failure was heavy rainfall and runoff which uprooted cuttings before the plants became established. Additional causes of failure during this period resulted from subsequent drowning of cuttings planted too low in the streambed, breakage of cuttings due to high flow velocities, planting too deep, and desiccation caused by planting on sites too far from the channel (Schultze and Wilcox 1985). Douglas (1987) noted that probable factors preventing seedling establishment of Setchell willow (<u>Salix setchelliana</u> Ball) adjacent to the channel of a Canadian river were water level recession after seed dispersal, inundation, and physical removal of fine substrate by strong currents. Inundation is harmful primarily when it occurs during the growing season. Maximum tolerance of willows to inundation is forty to fifty percent of the growing season, at least for initial establishment (Gill 1970; Kozlowski 1984).

The failure of woody plant establishment could be reduced by utilizing proven planting procedures. However, methods for interplanting woody species into established grass stands or recent seedings have not been fully developed and need further study (Platts et al. 1987; Skovlin 1984).

<u>Effects of Grasses on Streambank Stability</u>

Grasses form vegetative mats on streambanks which aid in reducing bank erosion and sloughing. Grasses, like woody species, cause sediment to settle out and build up banks during overbank flow events (Platts 1983; Platts and Rinne 1985). Grasses have traditionally been used to control sheet and rill erosion on bare soil during the establishment of woody plants (Lines et al. 1978). In terms of streambank stabilization, sodforming grasses may adequately protect the banks of low gradient streams (e.g. those flowing through meadows) or ephemeral channels, but for many small streams this type of vegetation alone is inadequate to resist the erosional forces of flowing water. The fibrous root systems of grasses, once exposed to running water, can easily be washed clean of soil particles leading to rapid bank erosion. For many stream channels, root systems of woody vegetation in combination with grasses and forbs provide a better physical barrier to the effects of high velocities and turbulence by creating banks with considerable surface roughness and relative stability compared to the effects of grasses and forbs alone. The result is that channel widening and erosion at bends can be significantly reduced or eliminated (Beschta and Platts 1986) when a comination of grasses, forbs, and woody vegetation are established (Platts and Rinne 1985).

Artificial Bank Protection Structures

Artificial channel and bank protection structures can also enhance riparian development. Bank protection structures are used for armoring banks, and for deflecting or separating flows. Armors are designed to keep banks in their present location; flow deflectors are used for eliminating erosional impacts on critical banks; and separators divide streamflow into high and low energy sections with the low energy flow moving adjacent to the bank (DeBano and Heede 1987). These structures can impact riparian zones both beneficially and adversely (Altpeter 1944; Beschta and Platts 1986; DeBano and Heede 1987; Elmore and Beschta 1987, 1988).

Altpeter (1944) noted that on extreme curves it was necessary to dispense with vegetation on the outside bank and resort to rock placement to stabilize the bank. Permanent protection of the toe of the bank by rock riprap was often necessary along straight channel reaches, as well as on outside banks.

Where bank protection structures are installed, lateral incremental stream adjustments cannot occur (Beschta and Platts 1986; DeBano and Heede 1987). This locks the stream channel into a fixed location resulting in a static stream channel (Elmore and Beschta 1987, 1988). However, many biological systems are dependent on normal channel and floodplain adjustments associated with systems in dynamic equilibrium (VanHaveren and Jackson 1986). Because stream systems are dynamic (Beschta and Platts 1986), natural channel function involves a mobile bed and localized fluctuations in channel geometry about a long-term stable average (Andrews 1982; Jackson and Beschta 1982). Only after a new stream equilibrium has been reached, can new riparian plant communities establish on sediment deposits (DeBano and Heede 1987).

Artificial bank protection structures do not directly obstruct the channel, but by deflecting or separating streamflow may affect nearby riparian sites. Channel aggradation induced by structures may be of such magnitude that existing riparian areas become buried (DeBano and Heede

1987) due to a wide range of flow and sediment transport conditions (Beschta and Platts 1986). Therefore, if an in-stream structure is large enough to cause deposition of most of a streams sediment load, erosion in downstream riparian zones may result because the sediment-free water has sufficient energy to pick up new sediment (Beschta and Platts 1986; DeBano and Heede 1987). Beschta and Platts (1986) noted that since bank protection structures are not dynamic, major rock works such as spur-dikes, revetment, and riprap, which are relatively permanent features, are undesirable for enhancing fish and wildlife habitat.

It is important to avoid excessive rigidity in rehabilitating stream-riparian systems in order to allow the biological processes associated with dynamic equilibrium to proceed (VanHaveren and Jackson 1986). Structures are often installed in streams where they are not needed because we rarely allow several years of vegetation recovery before identifying where in-stream structures will provide the greatest value. Installing permanent instream structures in rangeland riparian areas without changing vegetation management is counterproductive in the long run. Therefore, spending large amounts of money to build instream structures will seldom solve riparian problems and may only allow managers to sidestep difficult decisions (Elmore and Beschta 1987, 1988).

Maintenance of instream structures is necessary to insure that they continue to function properly. Frequently, once instream structures are funded and built, additional funding for annual or periodic maintenance is lacking (Beschta and Platts 1986). Therefore, it is usually easier and less costly to rehabilitate streambanks using vegetation than by using artificial stream structures (Platts 1983, 1985).

Riparian Research Needs

Little research has been initiated to reclaim streams and riparian zones to promote subsurface water storage, control non-point source pollution, and answer questions related to water rights and demand for new supplies downstream (Skinner et al. 1985). Methods and techniques for revegetating to achieve high establishment rates and reduced plant mortality have not been adequately evaluated (Platts et al. 1987; Skovlin 1984). In addition, evaluation of individual plant species to provide optimal streambank stability requires further study (Lines et al. 1978). There are adequate investigations and reports in the literature that demonstrate the importance of maintaining quality riparian habitats, but, these investigations fail to address the methods necessary to rehabilitate riparian environments. Future study should be directed toward determining how to rehabilitate these habitats once they are degraded (Platts and Rinne 1985).

The goal of this study was to evaluate the ability of tufted hairgrass, reed canarygrass, and Kentucky bluegrass to retain soil in the root mass while exposed to moving water. This research will begin to address the question of which grasses are adequate for the needs of the riparian rehabilitation scientist.

MATERIALS AND METHODS

Controlled Environment

This investigation was carried out in a controlled environment in the Montana State University Plant Growth Center. Daytime temperatures were held relatively constant at 21°C while night time temperatures were maintained at 13°C. These temperatures fall within the minimum and maximum temperature range for sustained shoot (16°C to 24°C) and root (10°C to 16°C) growth (Beard 1973). Supplemental lighting was used to insure a constant day length of fourteen hours, the average day length for Bozeman, Montana from April through September (Caprio et al. 1990).

Species Description

Deschampsia caespitosa (L.) Beauv. (Hitchcock 1971) (Appendix A) is a native species better adapted to high elevations typically ranging from 1900 to 3000 meters (Youngblood et al. 1985). It is a densely tufted, shallow rooted perennial reproducing primarily from seed (Best et al. 1971). Stands are common on poorly drained soils and in areas that are seasonally flooded (Hansen et al. 1988). Some researchers believe that many tufted hairgrass community types at high elevations have been replaced by Kentucky bluegrass (Franklin and Dyrness 1973; Hansen et al. 1988; Padgett 1981). The change in community dominance type from tufted hairgrass to Kentucky bluegrass has been attributed to poor grazing practices (Franklin and Dyrness 1973).

Phalaris arundinacea L. (Hitchcock 1971) (Appendix A) is a robust, native, deep-rooted species (Evans 1946; Booth and Rumely 1989) that spreads by short vigorous rhizomes. Stands typically occur on wet or poorly drained soils (Alberta Agriculture 1981). It tolerates more water during the growing season than any other cultivated grass yet withstands short summer droughts. New seedings are successfully established in early spring. This species is also easily established by pressing sod or stems with "joints" into wet soil (Hafenrichter et al. 1968).

Poa pratensis L. (Hitchcock 1971) (Appendix A) is an introduced, shallow-rooted, rhizomatous species (Weaver 1920; Evans 1946; Youngblood et al. 1985) that is a major community type at low to mid-elevations throughout the Rocky Mountain region. It is a common component of the driest riparian communities (Hansen et al. 1988) typically found on well drained, fine textured soils (Youngblood et al. 1985).

These species were selected for testing in this study based on their different rooting forms. They are also common components of riparian communities in southwestern Montana.

Procedures

A stream flow simulator (SFS) shown in Figure 1, was constructed to evaluate the ability of tufted hairgrass, reed

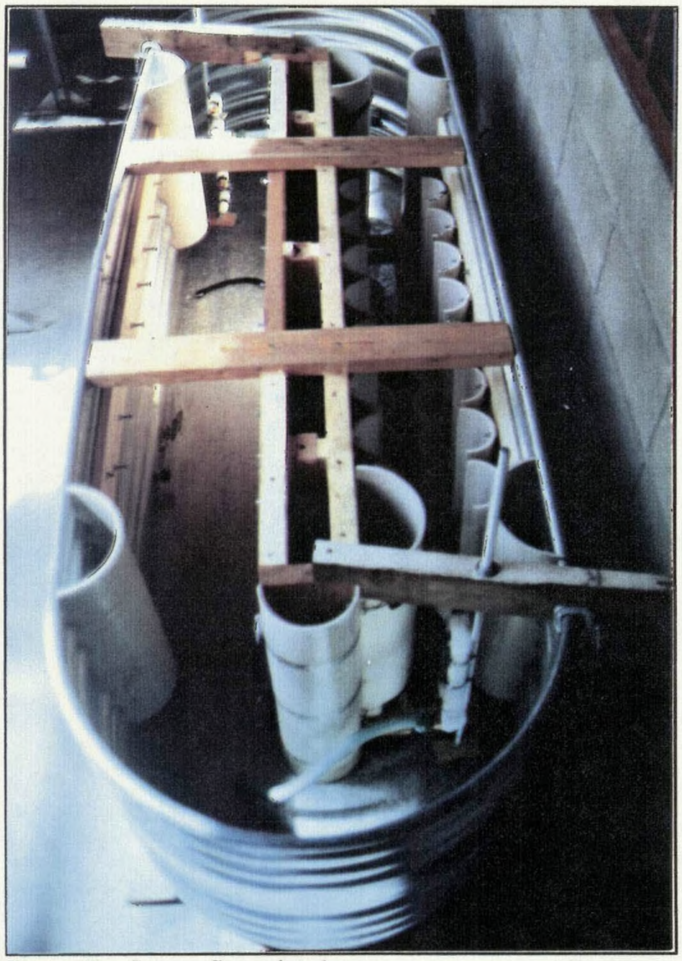

Figure 1. Streamflow simulator.

canarygrass, and Kentucky bluegrass to retain soil in the root mass while exposed to moving water. Artificial streamflow was created by attaching two 0.5 horsepower submersible pumps to a jet system (Figure 2). The system was calibrated to achieve a maximum flow rate of 0.97 m/sec which is the magnitude of flushing flow recommended by Wesche et al. (1986). Flow velocity was measured with a Montedoro-Whitney Corporation PVM-2 portable velocity meter. Velocities in the SFS ranged from 0.43 m/sec measured at the slowest end of the channel to 0.98 m/sec measured at the fastest end of the channel.

Each species was tested at bankfull and at 35% of bankfull levels. These flow levels were chosen because bankfull flow is the discharge stage that is the dominant control for changes in stream channel morphology (Petts and

Foster 1985), and below bankfull is the discharge stage which appears to have little influence on bank stability (Andrews 1982).

Trial Streamflow Simulator Run

Ten soil/root cores (10 x 38 cm) of each species were collected from representative stands found at three field locations. Tufted hairgrass samples were gathered from the bank of the Taylor Fork of the Gallatin River, 22.5 km south of Big Sky, Montana, reed canarygrass from the bank of Rocky Creek 1.6 km north of Bozeman, Montana, and Kentucky bluegrass from the bank of Cottonwood Creek 24.1 km north of Ennis, Montana.

Nine samples of each field collected species were weighed and then placed in 10 x 38 cm PVC containers (Figure 3), loaded into the SFS (Figure 4), and tested at the bankfull level. Samples were exposed to moving water in the SFS for twenty-four hours. Observations were made hourly by carefully feeling the soil column of each sample to detect soil loss. Visual examination was impossible due to the turbidity of the water. After twenty-four hours the cores were cautiously removed and allowed to drain for one hour before being oven dried at 75°C for twenty-four hours. Soil loss from each core was calculated as:

soil loss = initial dry wt. - final dry wt.

expressed in grams.

The data collected from the trial run were used for a

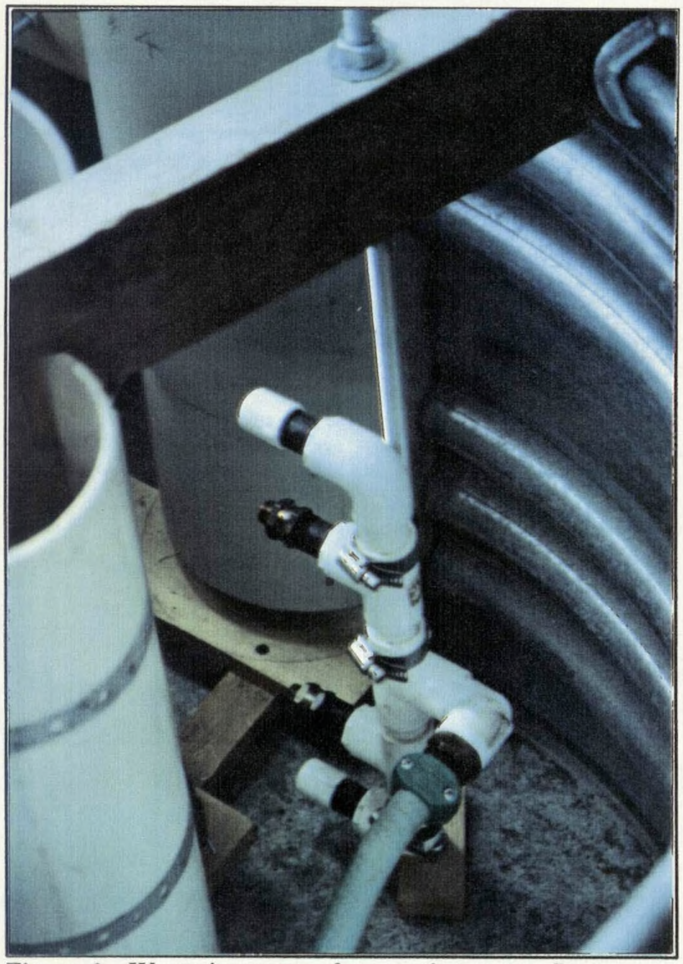

Figure 2. Water jet system for creating streamflow.

sample adequacy test. Sample adequacy was determined from the following formula:

$$n = (t\sigma/L\mu)^2$$

where: n = adequate sample size
t = Student's t-distribution
σ = standard deviation
L = allowable error
μ = sample population mean

The calculated sample size for each of the three species is shown in Table 1. The results of a simple statistical analysis are summarized in Table 2.

From the data shown in Table 1 it was decided that 48 samples of each species would be a feasible number for use in the experimental simulation runs. It was decided to determine if 48 samples of each species were in fact sufficient for the experiment.

Table 1. Calculated sample size for individual species.

Species[a]	t	σ	L	μ	calculated n
tufted hairgrass	1.645	12.35	0.10	30.04	45
reed canarygrass	1.645	3.71	0.10	23.44	7
Kentucky bluegrass	1.645	7.03	0.10	12.22	121

[a]9 samples of each species collected in the field

Sample adequacy was checked using the following formula:

$$n = (t\sigma_{\epsilon}/L\mu)^2$$

where:
- n = adequate sample size
- t = Student's t-distribution
- σ_{ϵ} = pooled variance
- L = allowable error
- μ = population mean

For this data:

$$n = (1.645 \cdot 5/0.10 \cdot 20)^2 = 17$$

The pooled variance of the nine individual field collected samples of each species was used in this equation to generate an unbiased estimation of the adequate sample size needed for the actual experiment.

Table 2. Percent soil loss from core samples, trial SFS run.

Species	n	Mean % soil loss	std
tufted hairgrass	9	30.4	± 12.4
reed canarygrass	9	23.4	± 3.7
Kentucky bluegrass	9	12.2	± 7.0

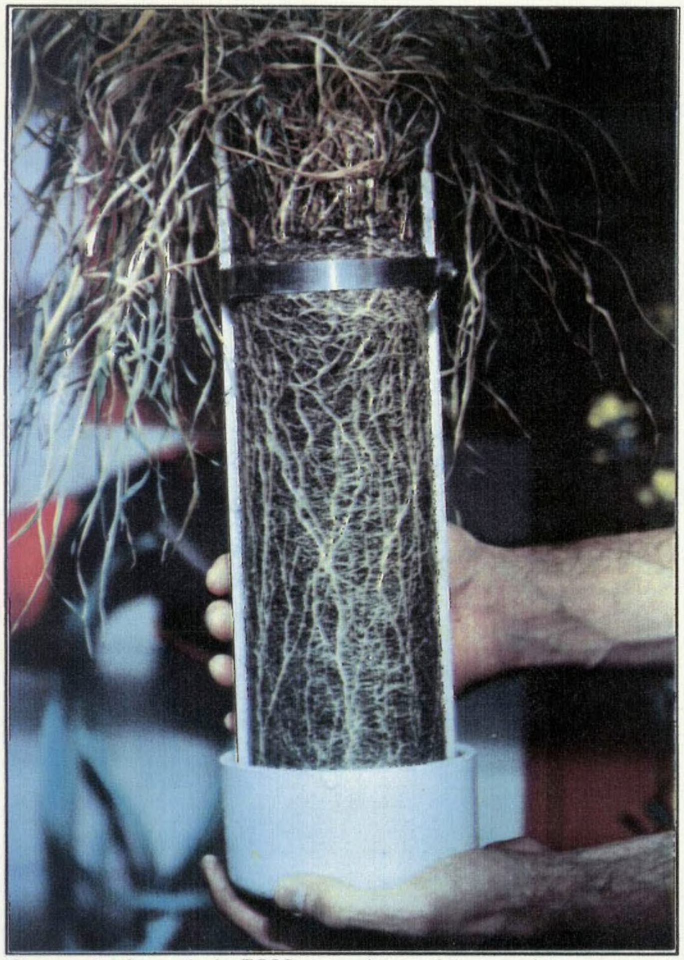

Figure 3. Sample in PVC container with one side removed.

This second test of sample adequacy indicated that at least seventeen samples of each species were needed to detect a 10% difference in soil loss with 90% confidence. Fourty-eight samples of each species were more than adequate for evaluating soil loss in this study.

Greenhouse Soil Construction

Ten soil cores of each species were originally collected but only nine were used in the trial SFS run. Soil texture was determined in the additional core sample of each species. The hydrometer method described by Gee and Bauder (1986) was utilized in this analysis. With this information an appropriate soil mix was constructed for the greenhouse container seedings.

Based on a particle size analysis of core samples from each species, a 5:1 ratio of silt loam to sand was chosen as the soil mix. One-hundred-forty-four PVC containers (10 x 38


Figure 4. Positioning of samples in streamflow simulator.

cm) were filled with soil mix. Six containers were selected at random and the soil from each was oven dried and weighed. A mean oven dry weight of 1540 g was calculated from the six samples and used as an estimate of initial soil weight for each of the one-hundred-forty-four containers. Prior to seeding, the containers were watered daily for one week to settle the soil. Each of the three grass species were planted separately in forty-eight containers. Seeding rates were the mid-range suggested by Bermant (1990) and are shown in Table 3. Twenty-four unseeded containers were used as controls.

<u>Growing Period</u>

Seedlings of each species emerged within one week. Plants were watered every other day to simulate high soil moisture conditions typically associated with natural streambanks. The plants were grown for ninety days. This

Table 3. Seeding rates used for container grown grasses.

Species	Pounds PLS[a]/ha
tufted hairgrass	4.4
reed canarygrass	15.4
Kentucky bluegrass	6.6
unseeded controls	0.0

[a] PLS= pure live seed

growth period was based on the growing season from peak spring runoff subsidence to the first fall frost in this region (Caprio et al. 1990). At the end of the ninety day period the plants were transferred to a vernalization room. Here they were subjected to a sixty day cold treatment to induce dormancy experienced under winter conditions. Day length during this period remained constant at ten hours simulating the average daylight period in Bozeman, Montana from October through March (Caprio et al. 1990). Daytime temperature was held at 4°C. Night time temperature was reduced to 1°C. At the end of the vernalization period the plants were returned to the greenhouse for thirty days, the estimated growing period from the last spring frost to the beginning of peak runoff. Temperatures ranged from 13°C at night to 21°C during the day. At the end of this period the plants were tested for soil retention ability in the streamflow simulator.

Streamflow Simulator Runs

For each simulator run eight samples of each species were randomly assigned to positions in the streamflow simulator. Runs lasted for a period of twenty-four hours. To account for any effect of velocity on soil loss, streamflow velocity was recorded at the fast and slow ends of the artificial channel on both sides of the SFS. Measurements were taken at hourly intervals for the first seven hours and one hour prior to conclusion of the run. At the end of each run rooted samples were carefully removed and allowed to drain for one hour. After draining, the soil/root column was sectioned into three 10 cm increments. All sections were oven dried at 75°C. The portion of the column below 30 cm was not affected during treatment due to the bottom cap on each container but soil weight in the cap was needed for determination of total soil loss. After weighing, plant roots were extracted according to Böhm (1936) and Schuurman and Goedewaagen (1965), dried at 40°C for 24 hours, and weighed. The initial intent was to analyze soil loss in each of the 10 cm sections to determine if there were differences in soil loss with increasing soil depth. However, during calculations of soil loss for each section, it was discovered that negative soil loss values were recorded. This implied that some sections were heavier after exposure to moving water in the SFS than they were before exposure. One explanation for this is that during the container filling process, two separate cart loads of soil were needed to fill all 144 containers. The difference in soil moisture content between the two cart loads may have been sufficient to cause the estimated 1540 g container soil weight to be much lower than the actual weight. Therefore, it was necessary to standardize the weight for the containerized soil in order to record positive soil losses for each sample. Standardization was accomplished by calculating the volume of soil occupying a container and dividing by the average bulk density of the soil mix.

The calculation is:

grams soil = volume container soil (cm^3)/bulk density (cm^3)

Average bulk density (Brady 1974) for components of soil mix are:

silt loam = 1.30 g/cm^3; sand = 1.50 g/cm^3

Greenhouse soil mix = 5:1 (silt loam:sand) = 80%:20%

Weight of greenhouse soil mix calculated as:

soil volume = 2676.3 cm^3

$$\text{silt loam} = \frac{2676.3\ cm^3}{1.30\ g/cm^3} \times (.80) = 1647\ g$$

$$\text{sand} = \frac{2676.3\ cm^3}{1.50\ g/cm^3} \times (.20) = \underline{357\ g}$$

$$2004\ g$$

Thus, the standardized initial soil weight per container was 2004 g. Soil loss from each soil/root column section was then calculated by:

Soil loss = 2004 g - final wt. - total root wt.

expressed in grams on a dry weight basis. This resulted in soil loss values for the entire soil/root column rather than for 10 cm sections and, as such, represent relative rather than absolute soil loss.

Root Elongation Measurements

In addition to the 144 samples used in the SFS runs, twelve samples of each species were grown in half sections of the PVC containers covered with 0.30 cm plexiglass (Figure 5). The objective was to record depth of root penetration of each species over time with the assumption that the species exhibiting the most rapid root elongation would provide more rapid soil stabilization. Beginning five days after emergence, root elongation measurements were recorded weekly for eight weeks. Measurements were terminated at the end of the eighth week when the roots of reed canarygrass reached the bottom of the container.

Experimental Design

A completely randomized design with three replications at each water level was used to detect a 10% difference in soil loss between species with 90% confidence. Data were statistically analyzed using multiple regression analysis (Lund 1988) to detect changes in the dependent variable, soil loss, as a function of changes in the independent variables; water level, species, root mass, and velocity. Potential interactions between species and water level, species and root mass, and species and velocity were also analyzed for effects on soil loss as were replications within each water level. Since species ($P=0.035$) was the only significant variable affecting soil loss, comparisons among treatment means were made using analysis of variance (Lund 1988).

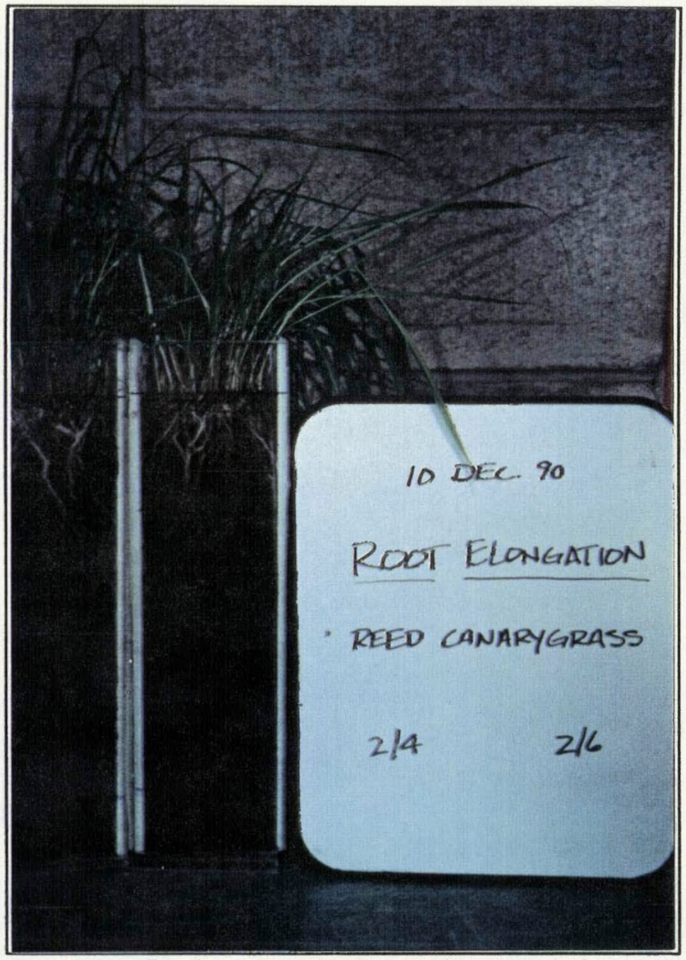

Figure 5. Plexiglass covered container used for root elongation measurements.

RESULTS AND DISCUSSION

Results

Based on the trial SFS run, it was hypothesized that tufted hairgrass would retain less soil in the root mass when subjected to flowing water than both reed canarygrass and Kentucky bluegrass, and that Kentucky bluegrass would retain less soil than reed canarygrass. In the actual experiment there was no significant difference in mean soil loss between tufted hairgrass and reed canarygrass or between reed canarygrass and Kentucky bluegrass, however, there was a significant difference in mean soil loss between tufted hairgrass and Kentucky bluegrass. Comparisons of treatment means for soil loss among species are shown in Table 4, while the results of the analysis of variance are shown in Appendix B.

It was hypothesized that there would be less soil retention in those samples tested at the high (bankfull) water level than those tested at the low (below bankfull) water level. There was no significant difference in mean soil loss between samples tested at bankfull flow compared to samples tested at below bankfull flow. Comparison of treatment means between the water levels is shown in Table 5. The results of the analysis of variance on water levels are printed in Appendix B. Complete soil loss was observed in all unvegetated controls at both water levels within five minutes after exposure to moving water.

Table 4. Comparison of treatment means for soil loss among species.

Species	n	Mean soil loss	std	T grouping[a]
tufted hairgrass	48	458.5 g	± 39.4	B
reed canarygrass	48	441.0 g	± 41.5	AB
Kentucky bluegrass	48	426.2 g	± 88.4	A
Control	24	2004.0 g	± 0.0	C

$\alpha = 0.10$
[a]LSD=19.88

Table 5. Comparison of treatment means for soil loss between water levels.

Water level	n[a]	Mean soil loss	std	T grouping[b]
High	72	434.6 g	± 59.3	A
Low	72	449.2 g	± 63.8	A

$\alpha = 0.10$
[a] 24 samples of each species
[b]LSD=16.44

It was also hypothesized that differences in root mass among the three species would control the amount of soil loss. There were significant differences in mean root mass between tufted hairgrass and both reed canarygrass and Kentucky bluegrass, but no significant difference in mean root mass between reed canarygrass and Kentucky bluegrass. Table 6 shows a comparison of treatment means for root mass among species, while the results of the analysis of variance are presented in Appendix B. The complete data set is shown in Appendix B.

Table 6. Comparison of treatment means for root mass among species.

Species	n	Mean root mass	std	T grouping[a]
tufted hairgrass	48	3715.0 g/m^3	± 1084	A
reed canarygrass	48	8556.0 g/m^3	± 2532	B
Kentucky bluegrass	48	8593.0 g/m^3	± 2587	B

$\alpha = 0.10$
[a]LSD=713.40

Even though the maximum flow velocity in the streamflow simulator was recorded at 0.98 m/s, from the full multiple regression model (Appendix B) it was determined that velocities of 0.98 m/s or less had no significant effect on soil loss. From the data shown in Appendix B it was concluded that the interaction between species and root mass, and replication E2 (replication 1 within low water level) had a significant effect on soil loss. However, analysis of the reduced multiple regression model (Appendix B) led to the conclusion that the interaction between species and root mass was insignificant. From analysis of variance on water level, replication within water level, and species (Table 7) it was concluded that species had the only significant effect on soil loss.

Reed canarygrass exhibited the fastest root elongation rate penetrating to a mean depth of 34.3 cm, the bottom of the container, in eight weeks. Root elongation in Kentucky bluegrass was second fastest with a mean penetration depth of

Table 7. Analysis of variance on water level, replication within water level, and species.

Source	DF	MS	F value	P value
Water level	1	7598.0	1.12	0.3502
Rep./water level[a]	4	6801.9	1.90	0.1138
Species	2	12544.2	3.51	0.0328

[a]MS used as error term

20.7 cm after eight weeks while roots of tufted hairgrass reached a mean depth of 14.7 cm over the same time period. Weekly root elongation rates are depicted in Figure 6.

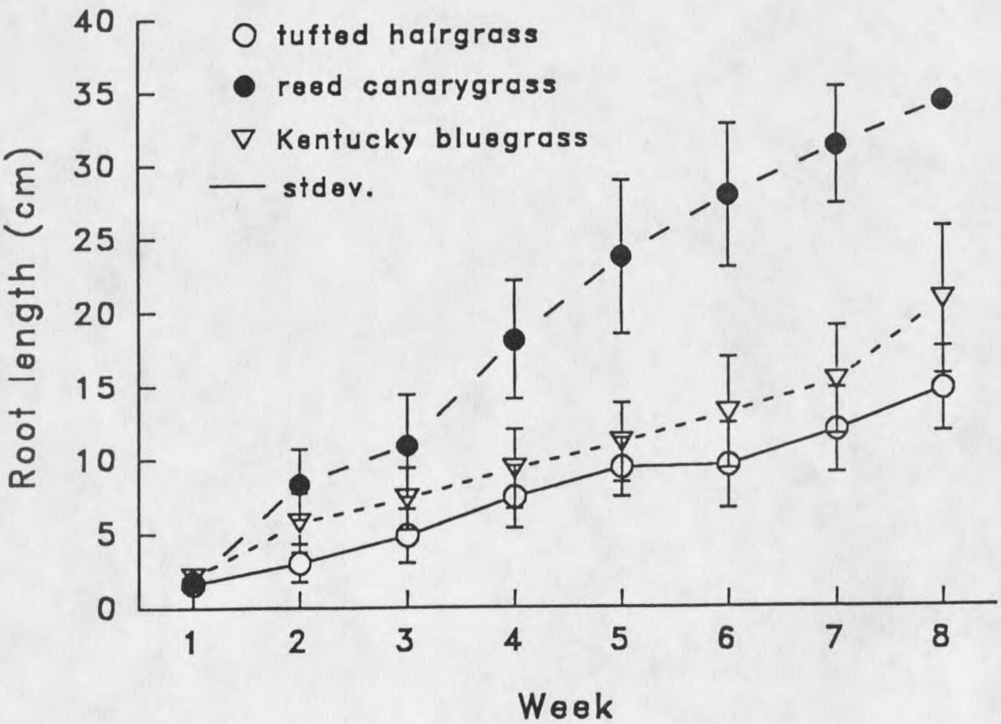

Figure 6. Weekly mean root penetration depths of tufted hairgrass, reed canarygrass, and Kentucky bluegrass.

Discussion

Complete soil loss from each of the twenty-four unvegetated controls supports the argument set forth by numerous investigators (Alpeter 1944, Monsen 1983, Jackson and VanHaveren 1984, Platts and Rinne 1985, Skinner et al. 1985, and Beschta and Platts 1986) that streamside vegetation protects streambanks. Because there was a significant difference in soil loss between tufted hairgrass and Kentucky bluegrass (Table 4), there is sufficient evidence to reject the null hypothesis that no differences in soil loss would be observed among species. Because there was no significant difference in soil loss between bankfull (high) and below bankfull (low) flows (Table 5), there is insufficient evidence to accept the alternate hypothesis that differences in soil loss between water levels would occur. Smith (1976) noted a decrease in bank erosion rate with an increase in root mass. There is evidence from this study which supports the conclusion set forth by Smith in that tufted hairgrass had a significantly smaller root mass than Kentucky bluegrass (Table 6) and lost a significantly greater amount of soil (Table 4). Platts (1983) and Elmore (1985) reported that vegetation reduces streamflow velocity thereby reducing erosion.

In this study, although flow velocity in the laminar boundary layer between water and the soil/root column was impossible to measure, the data support the inference that there was no significant effect ($p>0.10$) of velocity on soil loss. Beschta and Platts (1986) stated that sod-forming grasses adequately protect the banks of low gradient streams and ephemeral channels, but for many small streams this type of vegetation alone is inadequate to resist the erosional forces of running water. Grass root systems, once exposed to flowing water, can easily be washed clean of soil particles, leading to rapid bank erosion. Unfortunately, Beschta and Platts (1986) provide no definition of what they consider to be a "small stream". Reed canarygrass and Kentucky bluegrass, both sod-forming grasses used in this study, lost only 22% and 21% soil by weight respectively after a 120 day growing period. The results from this study indicate that Kentucky bluegrass will stabilize streambank soils better than tufted hairgrass along streams with flow velocities of 0.98 m/s or less after a 120 day growing period. Although all three species exhibited soil loss resistance to flow velocities of 0.98 m/s or less, further testing at increased velocities is necessary to determine velocity tolerance levels at which grass root systems are no longer capable of binding the soil as argued by Beschta and Platts (1986).

Elmore and Beschta (1987,1988) contend that woody species provide local channel stability and resistance to erosion so sedges, rushes, grasses, and forbs can establish. Figure 6 depicts the penetration of reed canarygrass roots to a depth of over 30 cm in 56 days. This could lead to the hypothesis that vigorous, deep rooted, rhizomatous grasses provide improved channel stability so slower growing woody vegetation can establish. Rapid root penetration by fast growing grass species such as reed canarygrass may provide rapid bank stabilization along degraded streams, compared to slower growing species like tufted hairgrass.

SUMMARY AND CONCLUSIONS

Based on the results from this study, rapid growing rhizomatous grasses seeded immediately after peak runoff may provide rapid stabilization to degraded streambanks so that slower growing woody vegetation can establish. Kentucky bluegrass provided quicker and superior soil stabilization in this experiment than tufted hairgrass. It might also be hypothesized that sodforming grass species such as Kentucky bluegrass have a greater capability to stabilize soil material than bunchgrasses such as tufted hairgrass. However, testing of species in the field is needed to adequately support these results. It may be inferred from this study that soil stabilizing differences do exist among grass species. Further testing may provide a basis for selection of grass species, according to rooting characteristics (i.e. root mass and root length density), for use in streambank rehabilitation projects.

Riparian ecosystem functions and processes are not fully understood. This lack of knowledge results in a need for further study and investigation of these systems. Additional testing of individual grass species for bank stabilizing capability as well as their ability to filter out sediment delivered by instream and overland flow will aid land managers and rehabilitation specialists in selecting the best species to provide rapid stabilization of degraded streambanks. Manning et al. (1989) suggested root length density may be better related to soil binding ability than root mass. This is because root length density is a measurement of root surface area. It is believed that greater root surface area equates to greater soil binding capacity. The ability to measure root length density in this study was not possible, therefore root mass was the measured variable used to compare soil binding ability between species. This factor may have limited the capacity to fully analyze differences in soil loss in terms of rooting structure. However, although there was no statistical correlation between root mass and soil loss ($R^2 < 0.10$) there were significant differences in both root mass and soil loss between tufted hairgrass and Kentucky bluegrass. Therefore, it might be hypothesized that root mass does have an effect on the ability of grass plants to retain soil when exposed to moving water. Comparisons of root length density and root mass in relation to soil binding ability require further investigation to determine the differences in soil binding ability between species with many thick roots such as reed canarygrass and those with numerous fine roots like Kentucky bluegrass. Further research opportunities include comparisons of root penetration rates between grasses and woody species to determine which growth form possesses the greatest ability for rapid soil stabilization. Velocity tolerance levels of established grasses to determine their bank stabilizing threshold also require investigation. In addition, improved planting techniques for establishing woody vegetation on streambanks already stabilized with grasses and improved methods for evaluating soil loss in a controlled environment also warrant further attention.

RECOMMENDATIONS

The following recommendations are based on observations and results from this study. These suggestions are not intended to be used in a "cookbook" approach to rehabilitating degraded streambanks. Each individual stream reclamation project should be carefully evaluated to determine the applicability of these recommendations to that particular stream.

Use of tufted hairgrass is recommended on high altitude (> 1900 m) streambanks of low gradient (< 1.5 %), low velocity (< 0.45 m/s) streams. Use of tufted hairgrass at elevations below 1900 m is not suggested because of its inability to compete with species in this region, e.g. Kentucky bluegrass. However, tufted hairgrass has proven to be very tolerant of acidic soil and ponded water and may prove beneficial on a site specific basis at lower elevations. Seed sources of tufted hairgrass are presently very poor therefore, vegetative establishment should be considered.

Kentucky bluegrass and reed canarygrass should be considered for use in streambank reclamation projects where stream gradients do not exceed 4.0 % and velocities are below 0.98 m/s. Soil moisture of each project site must be considered when using Kentucky bluegrass and reed canarygrass. Reed canarygrass withstands wet and poorly drained soils but will perform well in moderately drained soils. Kentucky bluegrass prefers well drained, fine textured soils but will tolerate short periods of high soil moisture.

Due to its rapid growth, native reed canarygrass is recommended when fast soil stabilization is required. However, the competitive nature of this grass may not be desirable when species diversity is critical. Reed canarygrass has been observed to suppress the growth of other plants where it has become established. Kentucky bluegrass provides moderate to rapid soil stabilization and may be more desirable than reed canarygrass when a diverse groundcover is one of the revegetation goals. Kentucky bluegrass is an introduced species which limits its use when native plants are desired. Seed of both species is commercially available and both are known to grow well from vegetative propagation.

The selective use of grasses for streambank stabilization is in its infancy. Therefore, it is important that land management professionals adequately evaluate each project site thoroughly before selecting which grass species to use in a rehabilitation effort.

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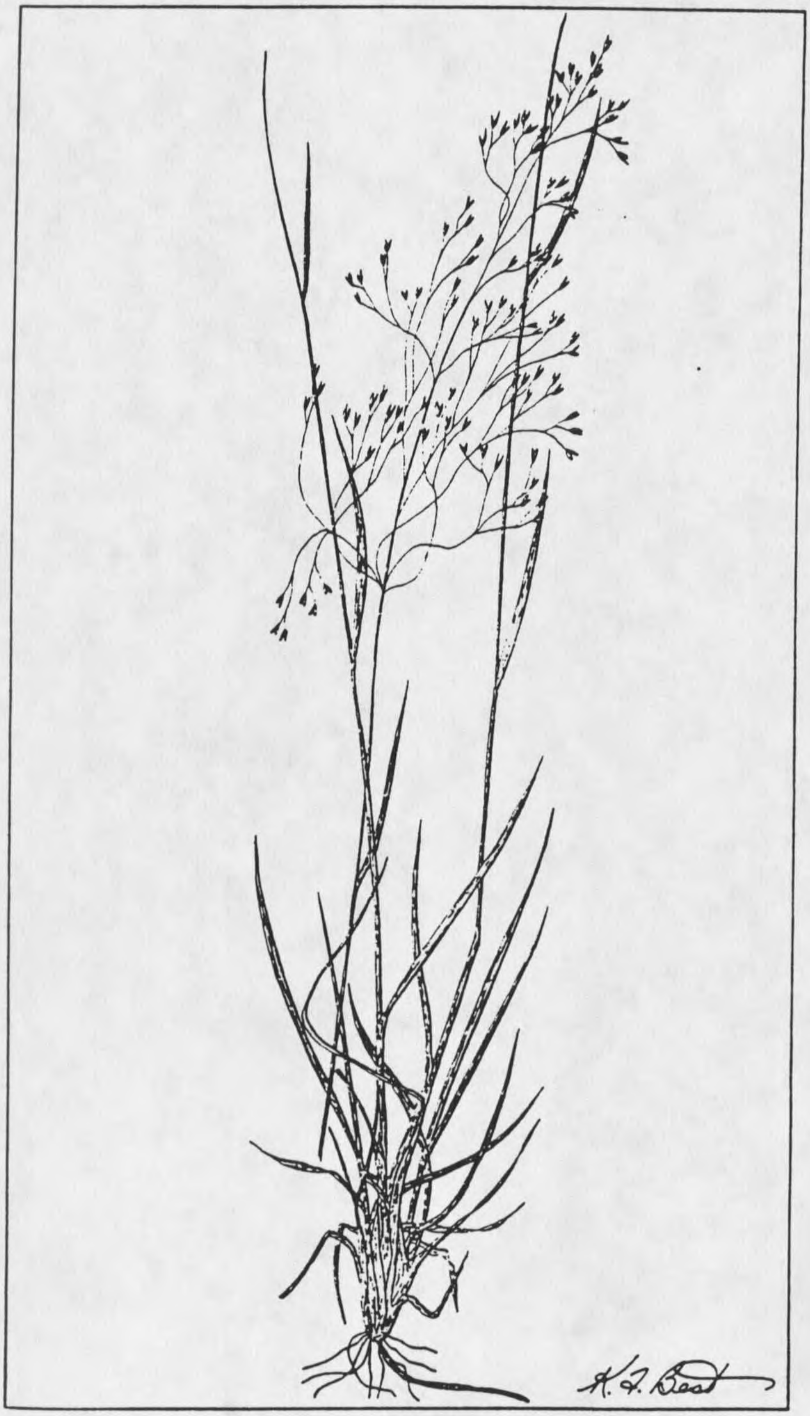
APPENDICES

APPENDIX A

DRAWINGS OF THE THREE GRASS SPECIES

Figure 7. Deschampsia caespitosa (L.) Beauv.



Best 1971

Figure 8. Phalaris arundinacea L.

Best 1971

Figure 9. Poa pratensis L.

Best 1971

APPENDIX B

DATA TABLES

Table 8. Full multiple regression model with soil loss as the dependent variable.

Independent variable	DF	P value
Root mass	1	0.6802
Velocity	1	0.1425
W	1	0.2657
S1	1	0.9355
S2	1	0.1453
E1	1	0.1850
E2	1	0.0855
E3	1	0.4985
E4	1	0.2061
WS1	1	0.9931
WS2	1	0.4039
S1V	1	0.8182
S2V	1	0.8775
S1RM	1	0.9373
S2RM	1	0.0881

Explanation of symbols:

W= water level

S1, S2= species

E1, E2, E3, E4= replications within water level

WS1, WS2= interaction between water level and species

S1V, S2V= interaction between species and velocity

S1RM, S2RM= interaction between species and root mass

Table 9. Reduced multiple regression model with soil loss as the dependent variable.

Independent variable	DF	P value
Root mass	1	0.7584
W	1	0.2180
S1	1	0.6041
S2	1	0.0426
E1	1	0.1695
E2	1	0.0824
E3	1	0.5149
E4	1	0.1960
S1RM	1	0.9896
S2RM	1	0.1192

Explanation of symbols:

W= water level

S1,S2= species

E1, E2, E3, E4= replications within water level

S1RM, S2RM= interaction between species and root mass

Table 10. Analysis of variance on soil loss among species.

Source	DF	MS	F-value	P-value
Treatment	2	12540		
Error	141	3698		
			3.392	0.0198

SE=8.78

Table 11. Analysis of variance on soil loss between water levels.

Source	DF	MS	F-value	P-value
Treatment	1	7598		
Error	142	3795		
			2.002	0.1388

SE=7.26

Table 12. Analysis of variance on root mass among species.

Source	DF	MS	F-value	P-value
Treatment	2	377800000		
Error	141	4670000		
			79.37	0.000

SE=314.90

Table 13. Data set for replication 1, high water level.

Sample[a]	Root weight (g)	Root mass (g/m^3)	Final soil weight (g)	Soil loss (g)
111	7.85	2933	1552	452
117	12.02	4492	1521	483
123	11.75	4391	1527	477
131	10.07	3763	1563	441
133	9.40	3513	1605	399
137	5.76	2152	1529	475
139	11.14	4163	1622	382
141	8.55	3195	1573	431
203	19.15	7156	1542	462
209	21.88	8176	1555	449
219	23.05	8614	1604	400
225	21.29	7956	1653	351
229	28.35	10594	1593	411
237	27.43	10250	1578	426
241	22.55	8427	1557	447
243	14.58	5448	1549	455
309	23.86	8916	1658	346
315	19.84	7414	1689	315
325	26.57	9929	1556	448
327	10.88	4066	1652	352
329	21.78	8139	1639	365
331	19.58	7317	1500	504
333	11.97	4473	1666	338
343	24.89	9301	1517	487

[a]100's=tufted hairgrass, 200's=reed canarygrass, 300's=Kentucky bluegrass

Table 14. Data set for replication 2, high water level.

Sample[a]	Root weight (g)	Root mass (g/m^3)	Final soil weight (g)	Soil loss (g)
103	12.57	4697	1549	455
105	18.70	6988	1494	510
109	10.75	4017	1511	493
113	11.54	4312	1518	486
115	4.41	1648	1547	457
125	9.68	3617	1571	433
129	17.37	6491	1569	435
147	5.41	2022	1591	413
201	21.36	7982	1566	438
207	30.29	11319	1521	483
211	18.58	6943	1534	470
215	22.68	8475	1554	450
217	27.91	10430	1570	434
233	11.64	4350	1564	440
235	11.97	4473	1575	429
239	27.44	10254	1593	411
301	21.59	8068	1561	443
305	33.77	12620	1542	462
311	21.34	7975	1675	329
317	27.96	10448	1696	308
321	23.06	8617	1571	433
339	13.82	5164	1686	318
345	25.63	9578	1714	290
347	17.93	6700	1495	509

[a]100's=tufted hairgrass, 200's=reed canarygrass, 300's=Kentucky bluegrass

Table 15. Data set for replication 3, high water level.

Sample[a]	Root weight (g)	Root mass (g/m^3)	Final soil weight (g)	soil loss (g)
101	9.94	3714	1488	516
107	12.87	4809	1498	506
119	8.99	3359	1523	481
121	11.03	4122	1560	444
127	9.56	3572	1564	440
135	9.74	3640	1561	443
143	12.35	4615	1613	391
145	6.62	2474	1602	402
205	34.29	12814	1530	474
213	31.84	11898	1620	384
221	35.27	13180	1586	418
223	34.57	12919	1580	424
227	25.56	9552	1565	439
231	16.92	6323	1466	538
245	17.97	6715	1554	450
247	25.94	9694	1570	434
303	18.91	7067	1547	457
307	16.83	6289	1607	397
313	30.38	11353	1516	488
319	17.00	6353	1538	466
323	34.09	12739	1726	278
335	23.91	8935	1444	560
337	33.82	12638	1478	526
341	22.39	8367	1492	512

[a]100's=tufted hairgrass, 200's=reed canarygrass, 300's=Kentucky bluegrass

Table 16. Data set for replication 1, low water level.

Sample[a]	Root weight (g)	Root mass (g/m^3)	Final soil weight (g)	Soil loss (g)
102	7.19	2687	1449	555
110	9.95	3718	1516	488
114	10.08	3767	1546	458
128	11.61	4339	1507	497
130	7.68	2870	1541	463
132	10.03	3748	1514	490
134	11.98	4477	1547	457
144	11.79	4406	1590	414
202	26.67	9966	1600	404
206	22.16	8281	1500	504
210	37.98	14193	1547	457
228	22.56	8430	1617	387
236	18.46	6898	1609	395
240	13.80	5157	1545	459
242	22.00	8221	1558	446
248	19.35	7231	1523	481
304	28.90	10800	1520	484
316	28.47	10639	1500	504
322	29.71	11102	1494	510
328	21.42	8004	1483	521
330	42.76	15979	1600	404
332	21.58	8064	1441	563
338	17.34	6480	1488	516
348	18.15	6783	1646	358

[a]100's=tufted hairgrass, 200's=reed canarygrass, 300's=Kentucky bluegrass

Table 17. Data set for replication 2, low water level.

Sample[a]	Root weight (g)	Root mass (g/m^3)	Final soil weight (g)	Soil loss (g)
108	7.22	2698	1493	511
112	6.42	2399	1534	470
118	8.32	3109	1529	475
120	7.52	2810	1601	403
122	9.52	3558	1591	413
138	9.10	3401	1561	443
140	14.28	5336	1579	425
146	4.40	1644	1526	478
204	23.15	8651	1508	496
208	19.19	7171	1541	463
212	15.50	5792	1519	485
214	25.90	9677	1542	462
216	36.84	13767	1654	350
222	27.62	10321	1611	393
224	19.65	7343	1657	347
246	12.02	4492	1545	459
306	21.71	8113	1521	483
312	16.07	6005	1670	334
314	13.67	5108	1703	301
324	20.12	7519	1661	343
336	13.64	5097	1575	429
340	26.88	10045	1463	541
342	18.47	6902	1620	384
346	23.20	8670	1496	508

[a]100's=tufted hairgrass, 200's=reed canarygrass, 300's=Kentucky bluegrass

Table 18. Data set for replication 3, low water level.

Sample[a]	Root weight (g)	Root mass (g/m^3)	Final soil weight (g)	Soil loss (g)
104	7.79	2911	1473	531
106	8.10	3027	1483	521
116	12.25	4578	1538	466
124	9.35	3494	1602	402
126	11.38	4253	1533	471
136	13.06	4880	1535	469
142	7.39	2762	1570	434
148	12.74	4761	1575	429
218	16.87	6304	1517	487
220	30.50	11398	1576	428
226	27.10	10127	1618	386
230	24.36	9103	1549	455
232	16.55	6185	1527	477
234	14.53	5430	1495	509
238	16.97	6342	1532	472
244	16.81	6282	1557	447
302	18.68	6981	1507	497
308	17.95	6708	1657	347
310	21.06	7870	1531	473
318	21.56	8057	1738	266
320	37.67	14077	1650	354
326	22.42	8378	1677	327
334	37.18	13894	1492	512
344	23.33	8718	1436	568

[a]100's=tufted hairgrass, 200's=reed canarygrass, 300's=Kentucky bluegrass