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EFFECTS OF DETERIORATED RANGE STREAMS ON TROUT

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ABSTRACT

Improper management of domestic livestock on western ranges has caused habitat degradation of trout streams in some areas. As a result, there is either less trout production or conditions have deteriorated to such a degree that the fish can not survive in the streams. To accommodate requirements of sportsmen for additional fishing opportunities and to achieve national objectives for better balance in managing resources, it is necessary for habitat degradation problems to be solved.

Livestock can alter the quality of stream habitat by damaging banks and decreasing the density of streamside vegetation. Bank damage, besides contributing to erosion and the alteration of channels, can eliminate important trout habitat associated with banks. When streamside vegetation is cropped unacceptably, erosion and sedimentation are promoted. If shading is decreased, water temperatures can elevate to levels unsuitable for trout. Sedimentation can lessen trout reproductive success and the production of aquatic insects which are the predominant food base.

Solutions to trout habitat problems will have to be achieved through inter-disciplinary efforts. Fishery biologists, watershed specialists, range management specialists, plant ecologists and other professionals must become collectively involved to perfect habitat improvement methods. It will be essential for the livestock industry to cooperatively assist land management agencies to improve stream conditions once suitable approaches are developed. ●

Introduction

With the dramatic increase in the number of sportsmen in the West, fishing pressures are intensifying and it is becoming more difficult for existing fisheries to satisfy sport fishing requirements. Consequently, resource managers are interested in improving conditions in streams to increase game fish populations for harvest by fishermen. Also, attention is being focused on accomodating environmental needs of fishery resources because of the establishment of national goals for better balance in resource management.

One opportunity for increasing fish numbers is that of improving degraded fish habitat associated with rangeland streams. Degradation has occurred in some areas because land management practices have not been implemented for protecting natural stream conditions. Resultingly, there are streams which once supported substantial game fish populations but now are marginal in quality for fish production or have deteriorated to a stage that only undesirable non-game species can survive.

Fisheries problems attributable to livestock management practices are addressed in this paper. Information is not presented with the intent of criticizing livestock use of rangelands. Instead, the purpose is to stimulate a better understanding by range specialists, range managers and stockmen about impacts of abused stream habitat conditions on fish. Once an understanding is achieved, the next step will be cooperative efforts in solving existing problems. Trout are emphasized in the paper because in the West these fish receive the most public attention.

Problem

It is recognized by fisheries specialists that livestock has been responsible for damaging some rangeland trout streams. Symptoms for a stream in an impaired condition can include bank caving and sloughing, channel straightening and widening, decreased average water depth, a high percentage of bottom area with superimposed silt and sand and limited stream shading *(Figures 1 and 2)*. Deterioration occurs because of the effects of livestock congregating along streams. The behavior is attributable to preference of animals for shade, lusher vegetation, and readily available drinking water.

As an example of how habitat alteration can collectively have an adverse impact on trout, information reported by Marcuson (n.d.) is applicable.[1] In Rock Creek in Montana, populations of brown trout (Salmo trutta) were compared in a heavily grazed zone and in a natural area. Within the natural area there were 1,880 (213 pounds) of trout per acre compared to 701 (63 pounds) per acre where heavy grazing occurred. Weightwise, the standing crop was 3.4 times greater in the natural area and fish in excess of 8 inches long were 300% more abundant than in the heavily grazed zone.

For effects of overgrazing in general, Behnke and Zarn (1976) cite it as one of the principal factors contributing to the decline of native trout in the West. Their assessment is as follows: *"Grazing livestock may destroy the vegetative cover and cave in overhanging bank, thereby eliminating the most*

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[1] For this and other information cited conceptually in the text for trout/habitat interrelationships, readers are cautioned not to assume that quantitative data apply categorically to all streams. Each stream has to be evaluated individually prior to (a) determining adverse impacts which have occurred as a result of a given land-use practice, and (b) predicting fish population responses should habitat be improved.

Figure 1. Stream habitat damaged by livestock in the Reynolds Creek watershed in southwestern Idaho. Note relatively sparse streamside vegetation, trampled banks, and channel widening. Stands of shrubs and trees near the stream are not dense as would prevail under normal conditions because replacement seedlings cannot establish. Plants in the bottomland background are predominantly curley cup gumweed (Grindelia squarrosa) considered to be an indicator of a disturbed range site. (BLM photo – Author)

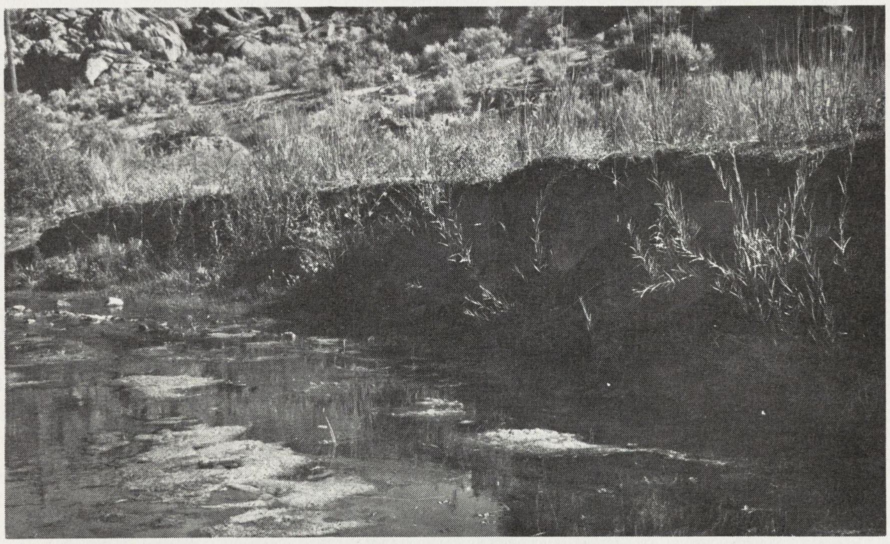

Figure 2. Streambank damage in the Reynolds Creek watershed in southwestern Idaho. The condition, which is aggravated by livestock trampling and cropping of forage, contributes to channel widening and sedimentation. After caving occurs, banks are susceptible to continuous sloughing. In addition to fish habitat being destroyed when banks are damaged, there is a gradual loss of valuable bottomland which lessens the carrying capacity for wildlife and livestock. (BLM photo – R.L. Lingenfelter)

important trout habitat. Loss of streambank vegetation leads to increased water temperatures, erosion and silting, elimination of spawning sites, and reduction of food supplies in the stream, all of which drastically degrade trout habitat." Effects of habitat damage by livestock on trout are indicated in *Figure 3* and specifics are addressed as follows.

Effects of Sediment

Sediments settle into spaces between gravel in which trout eggs are incubated. As a result, intergravel water flow is impeded and developing embryos do not receive adequate quantities of dissolved oxygen which has an adverse impact on development and survival. Also, metabolic wastes of the embryos are not flushed which contributes to higher mortality rates. As an example of how sediment can impact trout, data of Peters (1962) was plotted to indicate approximate relationships between mortality rates of incubating embryos of rainbow trout (Salmo gairdneri), different suspended sediment concentrations and intergravel water velocities *(Figure 4)*. Mortality rates exceeded 75% when sediments elevated to 200 parts per million or greater. As suspended sediment concentrations increased, intergravel water velocities decreased, indicating gravel clogging.

Bjornn (1973), after conducting field work with steelhead trout (searun rainbows), reported that for sand and gravel mixtures, less than 25% of the eggs developed to the emergent fry[2] stage when sand concentrations approximated 30%, compared to an excess of 75% emergence in concetrations less than 20%. Work of Corley (1976) also related to effects of sediment on reproductive success *(Figure 5)*. The quantity of fines (sediment less than 6.33 mm) was correlated to the approximate survival rate from the egg to fry stage in the South Fork of the Salmon River in Idaho. It was observed that survival decreased markedly when fines exceeded 20%.

It is important to maintain high reproductive success in streams because, even for normal conditions, natural mortality of trout from the egg to more advanced stages is pronounced. For an example, Shetter (1961) reported that for brook trout (Salvelinus fontinalis) mortality rates by the end of the first summer of life usually exceeded 90% of the original number of eggs laid.

Effects of Removal of Streamside Vegetation

Streamside vegetation in hot, arid areas typical of much of the western rangeland is extremely important for shading and maintaining tolerable temperature regimes. It is recognized by aquatic specialists that vegetation is an extremely important factor influencing summer water temperatures. For an

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[2] Emergent fry are fish which hatch in spawning gravel and then exit from the gravel to enter the stream for completion of other life stages.

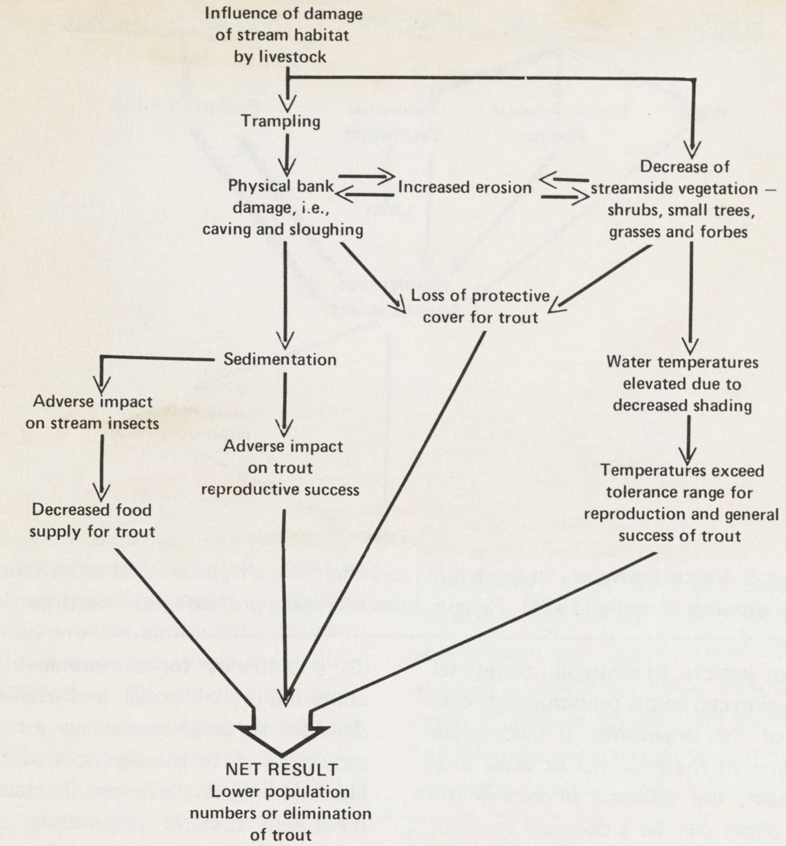

Figure 3. Conceptualized flow chart of adverse effects of stream habitat damage by livestock on trout populations.

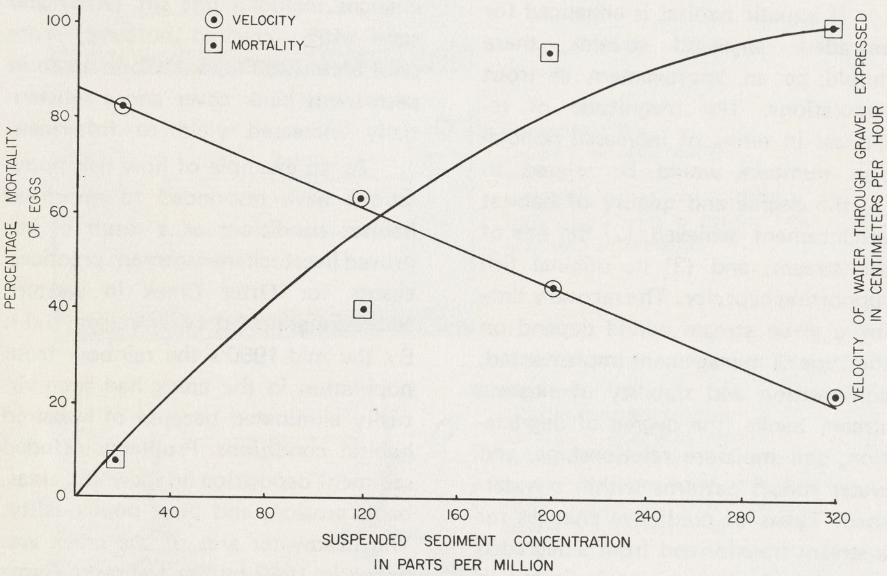

Figure 4. Relationships between suspended sediment concentrations, apparent intergravel water velocity, and mortality rates of rainbow trout eggs in Bluewater Creek, Montana. For the velocity plot, measurements were taken within the gravel. As suspended sediment concentrations increased, velocity decreased indicating sediment deposition and clogging of spaces between gravel. Data plotted from Peters (1962).

example, monitoring was conducted in the Needle Branch of the Alsea River in Oregon and the highest recorded temperature was 61°F. After logging and removal of streamside vegetation, the maximum temperature exceeded 85°F. In succeeding years, temperatures lowered as vegetation became

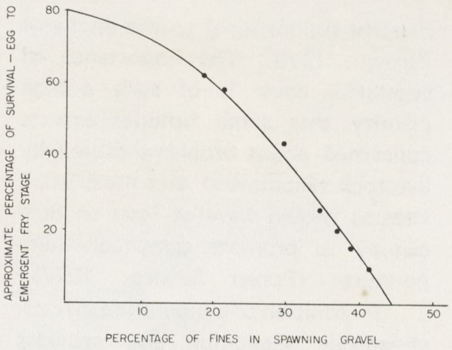

Figure 5. Survival of chinook salmon (Oncorhynchus tshawtscha) from the egg to fry stage in the South Fork of the Salmon River in Idaho correlated with fines (material smaller than 6.33 mm) in gravel (Corley, 1976). Salmon and trout have relatively similar environmental requirements.

re-established (Lantz, 1971).

In general, for trout to be successful, temperatures should not exceed the mid-60's during summer periods. For critical phases of the life cycle, such as spawning and hatching, they are less tolerant *(Table 1).* High temperatures can be lethal to trout directly, making them susceptible to diseases because of stress, inhibit reproductive success, and adversely affect spawning migrations (Lantz, 1971).

Water temperatures in small streams associated with rangelands commonly exceed 80°F when vegetation is eliminated by grazing, and conditions for trout become unsuitable. These streams are particularly susceptible to harmful temperature alterations because, as in logged areas, potential for change is directly proportional to the amount of stream surface area which becomes exposed and in-

Species	Optimum Range °F	Spawning Range °F	Hatching Range °F
Rainbow	54-66	36-68	55**
Brook*	47-52	38-45	39-54
Brown*	39-70	50**	36-52
Steelhead	45-58	39-49	50***
Cutthroat****	49-55	43-63	40-55

*Fall spawners.
**Optimum temperature.
***Preferred temperature.
****Salmo clarki.

Table 1. Temperature data for trout (Bell, 1973).

directly proportional to the discharge (Brown, 1973). The importance of vegetative cover is of such a high priority that some fisheries experts concerned about problems caused by livestock recommend that most small streams should have at least an 80% canopy to promote acceptable temperatures (Forest Service, 1977).[3]

In addition to temperature effects, streamside vegetation also provides cover for trout. This fact has been recognized for some time by fishermen and students of animal behavior. One investigator (Boussu, 1954) documented influences of cover in a small stream in South Dakota within experimental sections. The placement of cut brush by hand to simulate natural willow cover in density and position resulted in an increase of trout poundage by 258.1%.

Impacts of Habitat Alteration on Food Organisms

Trout in streams are normally dependent on aquatic invertebrates (predominantly insects) for nourishment, which has been verified by Griffith (1974), who examined stomach contents of brook and cutthroat trout in four Idaho streams. He reported that members of five aquatic insect orders comprised approximately 92% of the number of organisms eaten by the fish.

As indicated in *Figure 3,* sedimentation can adversely impace insects (Cordone and Kelley, 1961). Sediments are harmful because of abrasive actions and interference with functioning of respiratory organs. Also, the material settles over the most productive substrates, such as rubble, which reduces quality insect-producing habitat. The net result of sedimentation to insects can be the lessening of their diversity and production which impairs the food quality of a stream for trout.

Even if sedimentation does not become a problem, it is possible that removal of bankside vegetation can have an adverse impact on food supplies of insects. In a stream, terrestrial vegetation can be the predominant food base of the organisms. If inadequate amounts of material fall or wash into a stream, the ultimate impact in the food chain can be a decrease in trout production *(Figure 6).*

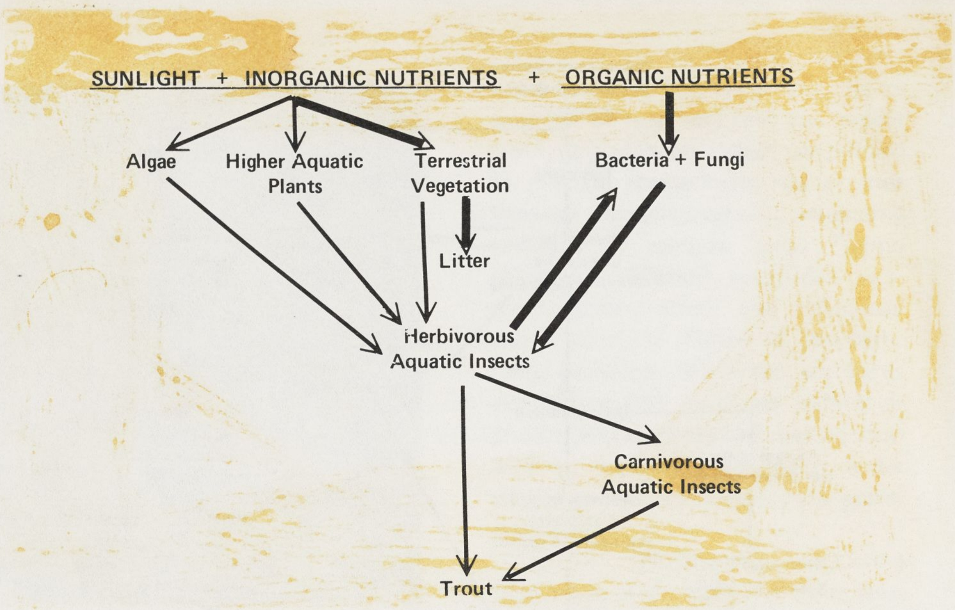

Figure 6. Major pathways in the trout food chain for a hypothetical range stream. Modified after Mundie (1974). Heavier lines represent greater rates of energy flow.

Types of Responses for Streams Managed for Improved Habitat

If aquatic habitat is enhanced for degraded rangeland streams, there should be an improvement in trout populations. The magnitude of responses in terms of increased population numbers would be related to (1) the degree and quality of habitat enhancement achieved; (2) the size of the stream; and (3) its original fish supporting capacity. The recovery time for a given stream would depend on the type of management implemented, composition and stability of existing stream banks, the degree of degradation, soil moisture relationships, and water runoff patterns within a watershed. Types of predictive changes for a stream transformed from a degraded to a recovered phase are indicated in *Figure 7.* After recovery, improved conditions for fish success would be associated with (1) a deeper average water depth; (2) flushing of silt and sand to expose gravel, rubble, and rock bottoms; (3) stabilized banks due to vegetative cover; (4) bank undercutting to provide additional cover for fish; (5) overhanging vegetation to shade and keep water temperatures lower during hot periods of the year; and (6) a tendency for a more favorable combination of pools and riffles to develop. As an example how a stream can respond to management, data of Hunt (1971) is pertinent. The author measured specific parameters of a brook trout stream during a three-year period after stream improvements were installed. Responses as indicated in *Figure 8* were recorded. Important changes included less silt (70%) and sand (40%) covered bottoms, more pool areas (289%), a 416% increase in permanent bank cover and a substantially increased yield to fishermen.

As an example of how fish populations have responded to improved habitat conditions as a result of improved livestock management practices, events for Otter Creek in western Nebraska are cited (Van Velson, n.d.). By the mid-1950's the rainbow trout population in the creek had been virtually eliminated because of impaired habitat conditions. Problems included sediment deposition on spawning areas, bank erosion, and poor pool quality. The headwater area of the creek was leased in 1969 by the Nebraska Game and Parks Commission and fenced to exclude livestock. Within three years, the stream had improved to become a major rainbow trout producer. The author reported that: *"The average width of the stream was decreased in places. Untrampled stream banks quickly stabilized, providing protection and resting areas for trout. More important, however, fencing protected the watershed from flooding, and sand*

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[3] The group authoring the report was comprised of aquatic specialists from the academic community and Federal agencies attending a livestock and wildlife-fisheries workshop. A report in preparation will be published by the Forest Service, Pacific Southwest Forest and Range and Experiment Station.

no longer drifted in to smother the gravel beds. Deep pools and clean gravel quickly appeared in the stabilized stream. The water temperature during the critical summer months was reduced 2 to 5 degrees, another benefit of stream fencing." In 1975, it was estimated that in approximately two miles of the fenced creek with 3.34 surface acres of water, 20,419 smolts (young migratory fish) were produced (Van Velson, personal communication, December 22, 1977).

Present Range Manager Predicaments

A range specialist seemingly has only three choices for management of a degraded stream. He can elect to (1) permit existing grazing conditions to prevail and risk additional deterioration; (2) manage exclusively for quality fish habitat; or (3) implement a livestock management program which will help improve the stream to some acceptable level of trout production.

Because of intensified public demands for better balance in approaches for resource management, it will become increasingly unlikely that option one (continue with existing conditions) will be a viable alternative. Therefore, there must be a trend toward achieving improved fish habitat. This poses a formidable problem because range specialists have failed to be active in developing innovative methodologies for improving habitat damaged by livestock. When a management decision is made to improve habitat conditions for fish, the only recourse at the present time is usually that of fencing a degraded stream to exclude livestock. Although habitat usually recovers dramatically after fencing, it is quite expensive and has other drawbacks. It is estimated that fencing costs for both sides of a stream per linear mile range from $2,000 to $6,000, and there are annual maintenance costs thereafter for repairing normal and intentionally-caused damage. Expenses such as those for services of a fence rider during the grazing season can also be incurred in some areas. Additionally, if a stream is fenced and the carrying capacity of the range outside the stream zone cannot support additional grazing, the stocking density for livestock has to be reduced or range improvement projects designed to provide additional forage have to be implemented.

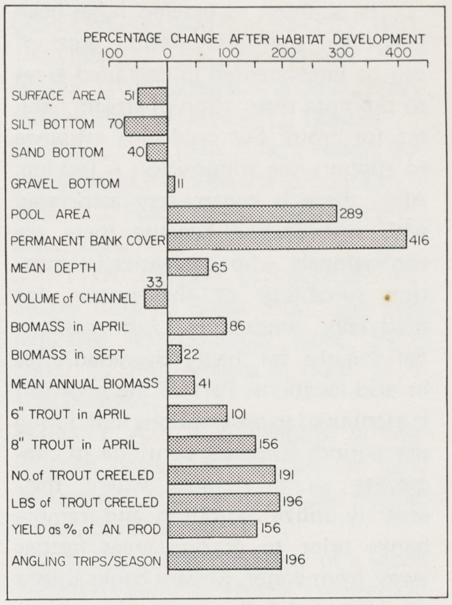
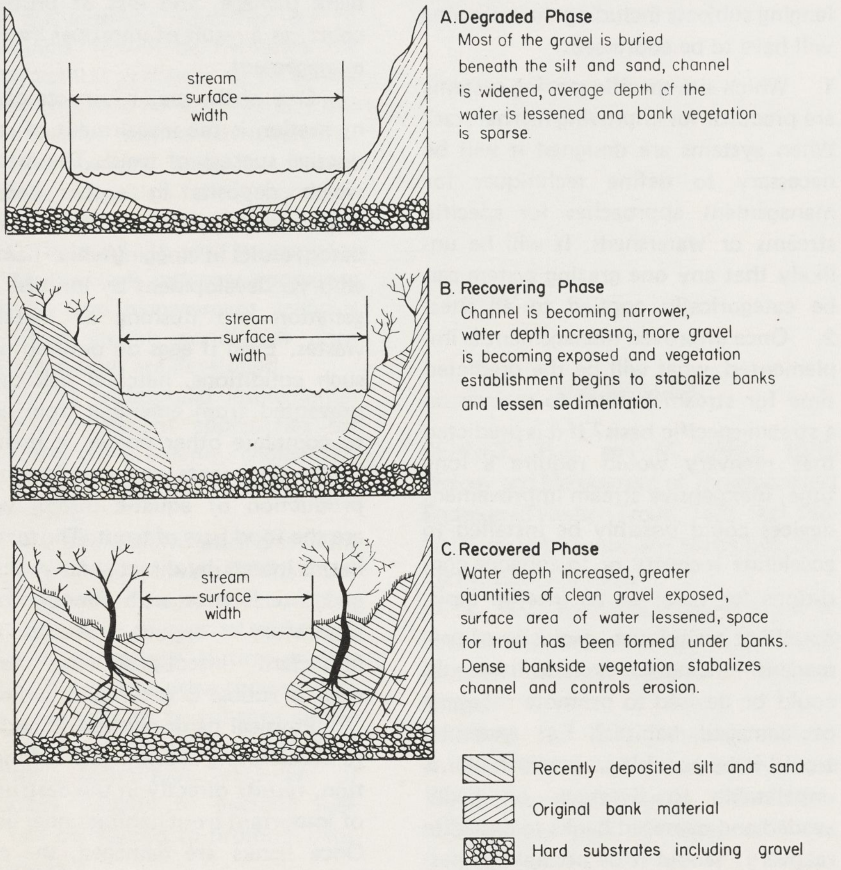


Figure 8. Recorded changes in characteristics in a 1.5 mile segment of Lawrence Creek in Wisconsin during a three-year period after stream improvement devices were installed (Hunt, 1971). For a range stream recovering naturally after management is implemented to improve habitat, it is predicted that similar types of changes would occur. Time requirements and magnitudes of change would vary on a stream-specific basis. Biomass changes in the figure refer to brook trout.



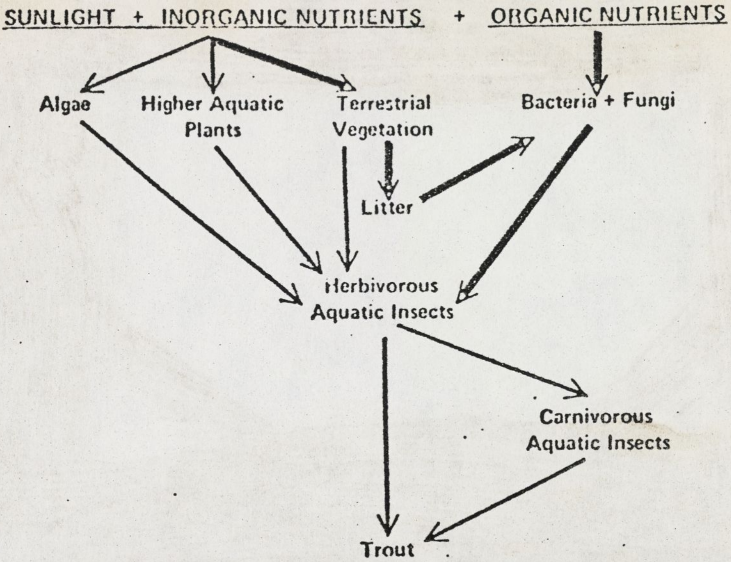
A. Degraded Phase
Most of the gravel is buried beneath the silt and sand, channel is widened, average depth of the water is lessened and bank vegetation is sparse.

B. Recovering Phase
Channel is becoming narrower, water depth increasing, more gravel is becoming exposed and vegetation establishment begins to stabalize banks and lessen sedimentation.

C. Recovered Phase
Water depth increased, greater quantities of clean gravel exposed, surface area of water lessened, space for trout has been formed under banks. Dense bankside vegetation stabalizes channel and controls erosion.

Figure 7. Conceptualized cross-sectional characteristics of a stream in degraded, recovering, and recovered habitat phases. For the Phase A condition, pool and riffle relationships are disrupted. In effect, there is a tendency for channeling to occur which lessens the abundance and quality of pools. For Phase C, note undercut areas which eventually develop. If bank damage occurs, this important trout cover is usually the first to be destroyed and years can be required for natural restoration. Adapted from White and Brynildson (1967).

The loss of forage for livestock can often be inconsequential, however, because it is usually necessary to fence off only a few feet along each stream bank. Also, there is evidence that as habitat recovers and water depth increases as a result of channel narrowing, the water table elevates. As a result of water table improvement, vegetation production can be stimulated adjacent to a fenced area to compensate for forage not accessible for grazing. Additionally, stream bank stabilization lessens erosive losses of productive bottomland which has a long term benefit of retaining grazing land for livestock.


SUNLIGHT + INORGANIC NUTRIENTS
+
ORGANIC NUTRIENTS
Algae
Higher Aquatic Plants
Terrestrial Vegetation
Bacteria + Fungi
Litter
Herbivorous Aquatic Insects
Carnivorous Aquatic Insects
Trout

In addition to fencing, it has been advocated that rest-rotation grazing[4] can be implemented in degraded areas to promote restoration of stream habitat for trout, but published evidence to support the supposition is lacking. Also, there is controversy associated with rest-rotation because there are professionals who categorically question suitability of the system for materially improving fish habitat, particularly for badly degraded sites in arid locations. Part of the negatism is attributed to observations that during use periods livestock continue to congregate along streams where they severely utilize vegetation, and trample banks prior to grazing areas farther away from water. Stream banks thusly are impacted each year grazing occurs and recovery is impeded.

Besides the concern that rest-rotation will not permit desirable streamside vegetation conditions to develop, it is suspected trout habitat under banks *(Figure 7)* will also not be restored. Habitat associated with banks is important for trout as was documented by Boussu (1954). He destroyed submerged pockets and then recorded responses of fish. In an altered area, the decrease of the standing crop poundage was approximately 33.3%. Biomass in a control section increased 19.7% during the same sampling period.

In respect to effects of grazing per se on areas associated with water, Hormay (1970) commented about the subject. The following quotations were in a personal communication dated September 1, 1976: *"Vegetation in certain areas, such as meadows and drainage ways, are invariably closely utilized under any stocking rate or system of grazing. Such use may be detrimental to wildlife, esthetic or recreational or other values. Where this is the case, about the only way to preserve values is to fence the area off from grazing. Reducing livestock or adjusting the grazing season usually will not solve such a problem."*

For option two (manage exclusively for quality fish habitats), a trend to maintain all stream habitat in this condition is improbable. However, for some streams this approach should be exercised, i.e., for those (1) supporting rare, endangered, or sensitive species; (2) which provide uniquely favorable habitat necessary for an important phase of a life cycle of a preferred species, such as steelhead trout; (3) with an existing or a potential fishery in geographic areas where sport fishing opportunities are limited; (4) designated for special activities such as ecological research and public water supply; (5) which presently are vitally important for maintaining sport fishing recreational demands; and (6) quality streams which currently are underfished but are projected to receive heavier use in future years.

Solutions for Stream Habitat Degradation Problems

Before it will be possible to implement widespread management to solve stream habitat problems, challenging subjects including the following will have to be addressed:

1. Which site-specific grazing systems are practical for improving fish habitat? When systems are designed it will be necessary to define techniques for management approaches for specific streams or watersheds. It will be unlikely that any one grazing system can be categorically applied to all sites.
2. Once improved management is implemented, what will be the predicted time for stream habitat to recover on a stream-specific basis? If it is predicted that recovery would require a long time, inexpensive stream improvement devices could possibly be installed to accelerate recovery or to enhance conditions for trout on an interim basis.
3. What techniques, exclusive of permanent fencing to exclude livestock, could be devised to promote recovery of damaged habitat? For example, would it be possible to establish plants unpalatable to livestock on badly eroded and damaged banks to expedite recovery? Would it be possible to perfect inexpensive, easily maintained devices to prevent livestock access to localized damaged areas of a stream to hasten habitat improvement?

Similar questions to those above, in addition to one about how different classes of livestock affect aquatic habitat, have been posed by Platts and Meehan (1977).

Prior to stream habitat problems being solved it will be necessary for research efforts to be expanded. The research must include meaningful evaluations of grazing systems on streams for a variety of site characteristics.

Where it is not practical to improve degraded habitat for the benefit of fish, mitigative approaches could possibly be developed. For example, it might be desirable to manage some watersheds to achieve water quality objectives instead of striving to markedly improve fish habitat. Then reservoirs could be established in these areas to be managed exclusively for fish production and sport fishing.

Conclusions

Trout are adversely affected by factors such as sedimentation, physical bank damage, and loss of protective cover, as a result of improper livestock management.

One of the major impacts of sedimentation is the impairment of reproductive success of trout. The material which deposits in spaces between spawning gravel where eggs are incubated results in clogging which hampers embryo development by limiting oxygenation and flushing of metabolic wastes. Even if eggs do develop under such conditions, hatched fish can be prevented from emerging from gravel to complete other phases of their life cycle. Sediments can also lessen the production of aquatic insects which are the food base of trout. The material harms insects by direct abrasive actions and interference with functioning of respiratory organs. Additionally, important insect-producing habitat such as rubble is covered by sediment.

Physical bank damage, in addition to accelerating erosion and sedimentation, results directly in the destruction of important trout habitat under banks. Once banks are damaged, the cross-sectional profile of a stream is altered which results in poorer quality trout habitat. Water depth is decreased because of channel widening, favorable pool and riffle relationships are disrupted and there is a tendency for greater quantities of sediments to

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[4] Rest-rotation can have various patterns. For a given pasture in a three-pasture system, there could be early spring use the first year, grazing from seed-ripe time until fall the second, and resting the third year.

deposit which decreases the abundance of suitable spawning gravel.

When streamside vegetation is eliminated by livestock, besides losing protective overhead cover for trout, there is less shading. In hot, arid areas common for much of the range lands, shade loss can result in water temperatures rising to levels unsuitable for trout.

Presently, range managers are at a disadvantage because, other than the fencing of streams to exclude livestock, there are few known practical practices which can be implemented to improve or maintain quality habitat for trout. Because of the establishment of national goals for achieving better balance in resource management and to accomodate needs of sports fishermen, the habitat of some range streams will be managed exclusively for quality trout production. For other streams, programs will be implemented to improve habitat to some acceptable level of trout production.

If management objectives are to be achieved, it will be necessary for existing habitat problems to be solved through inter-disciplinary efforts. Fishery biologists, watershed specialists, range management specialists, plant ecologists, individuals from other disciplines and management will have to get actively involved. It will be essential that the livestock industry cooperate and assist land management agencies to improve stream conditions once viable techniques are developed.

Prior to widespread implementation of management approaches for solving habitat problems, challenging subjects including the following must be addressed:

- Which site-specific grazing systems are practical for improving fish habitat?
- If livestock management approaches can promote improvement of habitat but extended periods of time are required, what can be done to accelerate habitat recovery or to enhance conditions for trout on an interim basis?
- Which techniques, exclusive of permanent fencing to exclude livestock can be devised to promote habitat recovery?

The success of programs designed to restore habitat quality of trout streams will depend on cooperative efforts. Once habitat is improved results can be mutually beneficial to sports fishermen and the livestock industry. ● ● ●

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Basic Hydrologic Studies for Assessing Impacts of Flow Diversions on Riparian Vegetation: Examples from Streams of the Eastern Sierra Nevada, California, USA

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ABSTRACT / As the number of proposals to divert streamflow for power production has increased in recent years, interest has grown in predicting the impacts of flow reductions on riparian vegetation. Because the extent and density of riparian vegetation depend largely on local geomorphic and hydrologic setting, site-specific geomorphic and hydrologic information is needed. This article describes methods for collecting relevant hydrologic data, and reports the results of such studies on seven stream reaches proposed for hydroelectric development in the eastern Sierra Nevada, California, USA. The methods described are: (a) preparing geomorphic maps from aerial photographs, (b) using well level records to evaluate the influence of streamflow on the riparian water table, (c) taking synoptic flow measurements to identify gaining and losing reaches, and (d) analyzing flow records from an upstream–downstream pair of gages to document seasonal variations in downstream flow losses. In the eastern Sierra Nevada, the geomorphic influences on hydrology and riparian vegetation were pronounced. For example, in a large, U-shaped glacial valley, the width of the riparian strip was highly variable along the study reach and was related to geomorphic controls, whereas the study reaches on alluvial fan deposits had relatively uniform geomorphology and riparian strip width. Flow losses of 20% were typical over reaches on alluvial fans. In a mountain valley, however, one stream gained up to 275% from geomorphically controlled groundwater contributions.

A growing awareness of the importance of riparian vegetation as an ecological resource, coupled with a recent proliferation of proposals to divert water from streams, especially for small hydroelectric development, has generated interest in developing methods for predicting the impacts of flow reductions on riparian vegetation.

Stream diversions for hydroelectric development can adversely affect riparian vegetation in several ways. Reduced flows in a diverted reach may (a) lower the stream water surface below levels that mosses and other submerged and emergent plant species depend on for direct contact or spray, (b) lower the alluvial water table below the rooting depths of some species, and (c) reduce annual high flows that otherwise could recharge bank sediments with moisture. Moreover, natural periodic flood scour may be reduced, especially by large hydroelectric development. As a result, the area, density, composition, and species diversity of riparian vegetation can be affected by altering streamflow. While the dependence of riparian vegetation on streamflow is generally accepted, efforts to quantify this dependence have been plagued with difficulties.

A general model for predicting the impacts of future diversions has proved elusive because of differences in species composition and in geomorphic and hydrologic setting from locality to locality. The autecology of component species in large part determines how a given riparian community is likely to respond to changes in groundwater levels or other changes in water availability. The geomorphic and hydrologic characteristics of a site determine how streamflow reductions are likely to affect water availability for riparian plants. It was toward defining this hydrologic link between streamflow and water availability that the methods described in this article were applied.

This article describes the types of hydrologic data relevant to impact analysis, and reports the results of our analysis of these data in a region where riparian

KEY WORDS: Hydroelectric impacts; Eastern Sierra Nevada; Riparian vegetation

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vegetation is of exceptional ecological and recreational importance.

Hydrologic Data Requirements

Many kinds of data can be useful in understanding local hydrologic conditions; none should be dismissed without some consideration. The following types of information proved especially useful in the case study reported here and should be of similar value elsewhere.

Geomorphic Setting

Overall geomorphic setting, which provides an indication of substrate and groundwater conditions, is often the best basis for classifying sites. Geomorphic features within a reach may also be responsible for local variations in water table that potentially affect riparian vegetation. In some cases, geomorphic information is available from published maps and reports. In general, however, simple geomorphic maps can be based on aerial photographs.

Response of Groundwater Levels to Flow Fluctuations

Determining the effect of streamflow changes on the availability of water for riparian plants requires an understanding of the nature of the interactions between streamflow and the alluvial water table. Where groundwater levels in the riparian zone can be measured (either by observing existing wells or by installing piezometers), their fluctuations can be compared with changes in streamflow as an indication of the degree to which the two are interrelated.

Gaining versus Losing Reaches

One of the most fundamental determinations to be made is whether a stream reach is *gaining* water from groundwater (a gaining reach), *losing* water to groundwater (a losing reach), or *in equilibrium* with respect to groundwater. There is general agreement that losing reaches are more sensitive to flow reductions than are gaining reaches: The shallow water table in a losing reach is probably dependent upon flow whereas, in a gaining reach, riparian vegetation may be supported by inflowing groundwater (Risser and others 1984). Patterns of gaining or losing along a stream may vary seasonally; for a given point in time, they can be detected by comparing simultaneous measurements of flow at more than one site along the stream.

Study Area

In the Owens River basin, California, USA, license applications are pending for small (<5 MW) hydroelectric projects that would divert water into penstocks from seven streams in the eastern Sierra Nevada, Mono and Inyo counties. Because of the need to assess the possible cumulative impacts of many small projects clustered in a single river basin, the Federal Energy Regulatory Commission (FERC) has recently applied the Cluster Impact Assessment Procedure (CIAP) in the Owens River basin (FERC 1985 and 1986). The case study described here was conducted as part of the CIAP in the Owens River basin.

The Inyo National Forest, which includes the study stream reaches, is one of the premier recreational areas in the United States by virtue of its extraordinary scenery and proximity to major metropolitan areas. In 1983, the Inyo National Forest received over 7 million recreational visits, more than Glacier, Yellowstone, and Grand Canyon national parks combined. Recreational use is heavily concentrated in riparian zones, both for directly water-related activities such as angling, and because of the cooler microclimate, aesthetic qualities, and more abundant wildlife in the riparian zones (FERC 1986).

The proposed new hydroelectric projects would divert water from reaches with a total combined length of 29 km on seven streams (Table 1 and Figure 1) and have generating capacities ranging from 950 kW to 4200 kW. They would operate in a run-of-the-river mode, that is, the projects would use available streamflow in excess of required minimum flow releases; the seasonal distribution of flow would not be affected by storage of water, because impoundments would be small, designed only to divert flow. The shortest proposed diversion (or bypassed reach) is less than 2 km (McGee Creek), and the longest is about 9 km (Pine Greek). All diverted reaches are steep, with average slopes ranging from 7% (Pine Creek) to 17% (Horton Creek), as measured from US Geological Survey (USGS) 15′ topographic maps. Drainage areas for the study streams range from 21 km^2 (Tinemaha Creek) to 98 km^2 (Pine Creek); average flows range from 0.12 m^3/s (Red Mountain Creek) to 1.3 m^3/s (Pine Creek) (Table 1). Recording stream gages are maintained on all affected streams by the Los Angeles Department of Water and Power (LADWP) (Figure 1).

Topographic relief is remarkably high along the eastern front of the Sierra Nevada (on the western side of the basin), with peaks over 4000 m in elevation rising above the floor of the Owens Valley (below 2000 m in elevation) less than 30 km away. Precipitation occurs primarily as snow at higher elevations. East of the Sierra crest, the Owens Valley lies in a rain shadow; the town of Bishop receives 145 mm of precipitation annually (California Department of Water

Table 1. Streamflow characteristics of study streams and study reaches.

Stream	Gage elevation[a] (m above msl)	Drainage area[a] (km²)	Average flow[b] (m³/s)	Study reach[c]		Synoptic flow measurements[d]			
						August 1985		October 1985	
				Length (km)	Gradient (m/m)	Flow in upstream site (m³/s)	Change over study reach (%)	Flow in upstream site (m³/s)	Change over study reach (%)
McGee Creek	2190	54	0.84	1.7	0.082	0.97	−08	0.46	−12
Rock Creek	2220	93	0.86	3.7	0.072	0.72	−17	0.41	−03
Pine Creek	1600	98	1.34	8.9	0.070	0.42	+180	0.25	+275
Horton Creek	1690	35	0.24	3.3	0.165	ND	ND	0.12	−02
Big Pine Creek	1390	82	1.18	4.1	0.099	1.44	+09	0.39	+16
Tinemaha Creek	1680	21	0.24	3.6	0.115	0.39	−20	ND	ND
Red Mountain Creek	1680	24	0.12	4.2	0.099	0.20[e]	−17	0.10[e]	−20

[a] Gage locations are given in Figure 1; all gages are near the lower end of the study reaches except on Horton Creek and Big Pine Creek, where they are 5.7 and 4.9 km downstream, respectively. Gage elevations and drainage areas were taken from the annual US Geological Survey publication *Water Resources Data for California* or from USGS topographic maps.

[b] Average flows were taken from the Los Angeles Department of Water and Power gaging records through 1985 except for McGee Creek, which was taken from the license application. Big Pine Creek values reflect combined flows at the gage and the diversion to Giroux ditches upstream.

[c] Study reaches are reaches proposed for diversion. Lengths and gradients of proposed diverted reaches were measured from USGS topographic maps, based on project specifications in licence applications.

[d] See FERC (1986: appendix B.7) for exact dates and locations of measurement, margins of error, and so forth.

[e] Values for Red Mountain Creek were taken not from synoptic flow measurements but from the regression line of historical flow data presented in Figure 7. Flows in columns 7 and 9 are approximately equal to long-term mean flows for August and October, respectively.

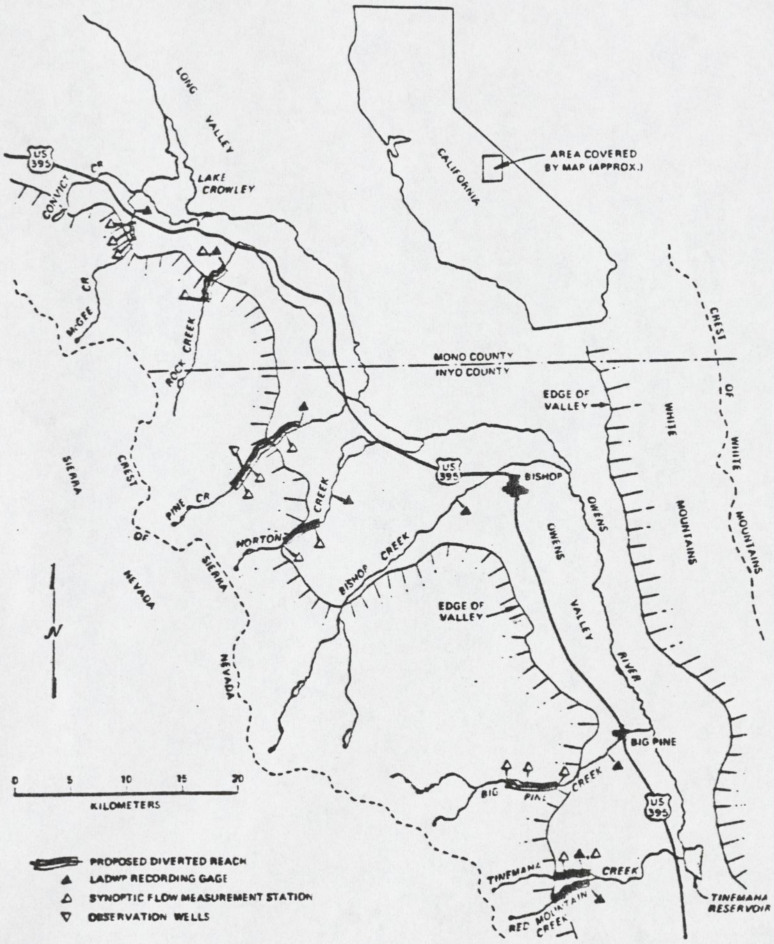

Figure 1. Location map showing streams proposed for hydroelectric development. Reaches to be bypassed by the proposed projects are *highlighted;* Los Angeles Department of Water and Power (LADWP) recording gages are designated by *solid triangles;* synoptic flow measuring sites are designated by *open triangles;* and Pine Creek observation wells 1A and 2 (which plot together at this scale) are designated by *downward-pointing, open triangles.* Based on Mariposa Sheet, US Geological Survey 1:250,000 series.

Resources 1980). Accordingly, the study streams receive nearly all their runoff from snowmelt high in the watershed and flow through a semiarid environment in their lower reaches, including the reaches proposed for hydroelectric development. Annual hydrographs are characterized by high summer snowmelt flows and low winter base flows.

All of the natural stream reaches that would be affected by the proposed hydroelectric projects are flanked by relatively narrow, well-defined strips of riparian vegetation that stand out as dark green ribbons against the surrounding semiarid scrublands (Figure 2). Riparian vegetation is defined, for the purposes of this analysis, as streamside vegetation that is structurally and floristically distinct from adjacent plant communities (Taylor 1982). In most cases, the riparian canopy is dominated by trees such as willows (*Salix* spp.), water birch (*Betula occidentalis*), and black cottonwood (*Populus trichocarpa*), but ponderosa (*Pinus ponderosa*) or Jeffrey pine (*P. jeffreyi*) and aspen (*Populus tremuloides*) may be present on some reaches. The understory frequently includes shrubs or small trees such as wild rose (*Rosa woodsii*), serviceberry (*Amelanchier utahensis*), elderberry (*Sambucus* spp.), and a vine component (for example, *Clematis ligustifolia*). The nature and extent of the herbaceous layer varies considerably among creeks and reaches, but frequently includes sedges (for example, *Carex lanuginosa*) and rushes (for example, *Juncus orthophyllus*). Many of these plants are recognized as phreatophytes in the area of the proposed projects.

Development of riparian vegetation is extremely limited in the study area, existing almost exclusively along perennial streams that originate in high, snowy elevations. In the Inyo National Forest, where most of the project reaches are located, riparian areas account for only 0.2% of nonwilderness land and 0.4% of all land. Riparian vegetation has already been heavily disturbed in the Owens Valley and the adjacent Mono Basin. The Owens River itself has been diverted over much of its length to supply water and power to the city of Los Angeles (Kahrl 1982), resulting in large losses of riverine riparian habitat. The California Department of Fish and Game estimates that 88% of the stream miles in the Owens–Mono region have been affected by diversion for power production and irrigation, including about 20% totally diverted and 37% having 50% or more of their flow diverted (Wong and Shumway 1985).

The ecological importance of riparian zones is widely recognized [see, for example, Warner and Hendrix (1984), Johnson and others (1985)]. However, because of the extreme contrast with their near-desert surroundings, the riparian areas remaining in the Owens River basin are of exceptional ecological importance. In general, riparian vegetation stabilizes stream channels and floodplains; regulates biogeochemical cycles, water temperature and quality, and the duration and magnitude of flooding; and provides diverse cover, food, water, reproductive habitat, and migration corridors for many aquatic and terrestrial fauna. In the eastern Sierra Nevada, riparian areas support a wide variety and relatively large density of terrestrial wildlife, including native ungulates such as mule deer, many small mammals, songbirds, raptors, and amphibians. These riparian zones support more species diversity, higher population densities, and greater plant and animal biomass than any other habitat in the Owens River watershed, and about 75% of local wildlife species require riparian habitat at some phase of their life cycle.

Data? Refs?

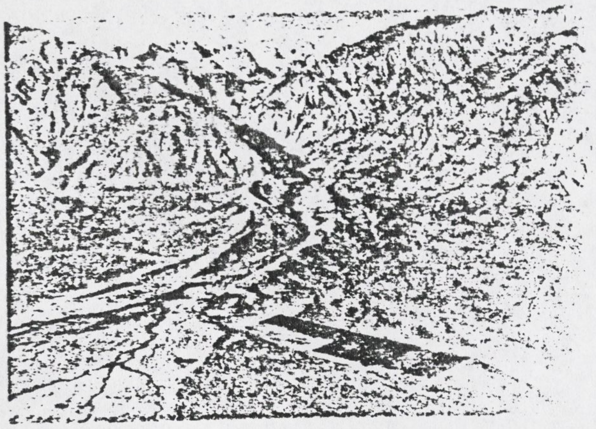

Figure 2. Oblique aerial view southeast to Pine Creek as it leaves its U-shaped, glaciated mountain valley to flow between glacial moraines and over its alluvial fan. The community of Rovana, which provides housing for workers at the UMETCO mine and mill, is visible on the alluvial fan. To the left (south) of Pine Creek is Mount Tom; at over 4100 m, it towers 2300 m over the Pine Creek valley. Flow is from right to left. Photo by G. M. Kondolf.

Background

Taylor (1982) developed a multiple linear regression model relating the width of the riparian corridor along undiverted reaches in a variety of geomorphic settings in the eastern Sierra Nevada to average flow, gradient, and degree of channel incision. He found that average flow alone explained 44% of the variance in width of riparian strip; all three variables explained 68%. [The width of the riparian corridor was measured from aerial photographs, average flow was obtained from stream gage records maintained by the LADWP, and channel gradient and incision index, defined as one-half the average distance between 80-foot (24-m) contours paralleling the stream, were measured from topographic maps.] Risser (1986) has since found that Taylor's model does not apply to available data from all types of diverted reaches. Risser developed a new relationship between flow and riparian strip width that explained 53% of the variance for diverted reaches on alluvial fans. However, mean annual flow did not correlate with riparian strip width on diverted reaches in glaciated valleys (Risser 1986).

The imperfect correlation between average flow and riparian strip width is not surprising because unregulated average flow is a variable that integrates many characteristics that increase with the size of the drainage basin. Larger streams tend to have lower gradients and are more likely to have broader valleys with more extensive and better sorted alluvial deposits that provide larger areas of relatively flat ground with potentially shallow water tables. When the average flow is reduced by diversion, these other size-related characteristics do not change. Other reasons why the model might not fit presently diverted reaches include potentially long lag times for biological response to reduced water availability, differences in timing between the diverted and natural flow regimes, and possible contributions to the diverted reaches from groundwater that was recharged from the streambed upstream of the diversion.

The importance of geomorphic control along streams of the eastern Sierra Nevada was demonstrated in a study by Jones and Stokes Associates (1985) in which riparian strip width was found to be most closely correlated with floodplain width, which in turn was found to be highly variable but generally greater in valleys underlain by glacial till and on alluvial fans than in valleys with shallow bedrock floors. It is generally agreed that the geomorphic setting of a site must be considered before the probable response of riparian vegetation to future streamflow diversions can be assessed (Risser and others 1984). However, a simple system for quantifying geomorphic influences for use in predicting impacts has proved elusive, and none is presently available for use in impact assessment.

Methods

To characterize the geomorphic and hydrologic regimes of the reaches proposed for diversion, we mapped geomorphic features, gathered existing groundwater data, and measured gains and losses in flow along the length of reaches proposed for diversion. Geomorphic features such as glacial moraines, faults, and debris fans were mapped directly from aerial photographs (ca. 1973, scale 1:24,000) provided by the Inyo National Forest. No attempt was made to compensate for radial distortion in the photographs, so the resulting maps are nonquantitative depictions of spatial relationships between the streams, geomorphic features, and riparian belts.

To compile groundwater data, we identified existing wells along or near the study reaches and obtained drillers' reports on the wells, measured water levels in the wells with an electric well probe, and surveyed each well reference point and stream water surface to a common datum. Unfortunately, most of these wells were perforated at such great depths and were located so far away from the streams that they did not reflect the shallow unconfined water table. (Bouldery substrates along the study reaches made it impractical to install new piezometers for this study.) However, on Pine Creek, an observation well maintained by UMETCO Minerals Corporation had the favorable characteristics of shallow screening and a 16-month record of observations in 1983 and 1984, so its fluctuations in water level could be compared with changes in flow in the adjacent stream.

To identify zones of groundwater recharge and discharge, and to quantify the gains and losses in flow, we conducted synoptic (simultaneous) flow measurements on six of the seven study streams at stations shown in Figure 1. For the seventh stream, Red Mountain Creek, historical flow records for the existing gage and the discontinued gage upstream were analyzed. Flow was measured with a Price AA or pygmy current meter (depending on flow) in accordance with standard practice (Buchanan and Somers 1969). The procedure for synoptic flow measurements, or "seepage investigations" (Riggs 1972), is to select days of steady or very slowly changing stage, measure the flow at two or more sites along the length of the stream, and, allowing for tributary contributions and surface diversions, compute the gains or losses in flow between the measuring stations. The measurements are treated as simultaneous and, with the assumption of steady flow, any changes in flow can be attributed to groundwater interactions or evapotranspirative losses along the intervening reach.

The greatest challenge we faced in conducting these synoptic flow measurements was to locate cross sections with flow characteristics favorable for measurement. The study reaches consist predominantly of steep boulder cascades with extremely turbulent, nonuniform flow. Flow measurements made in sites with such turbulent hydraulics may be subject to large errors because of the nonlogarithmic form of the vertical velocity profile (Jarrett 1985). Because we were measuring potentially subtle changes in flow, it was essential that we locate sites with better flow characteristics. The sites we selected for measurement were atypical of the steep study reaches and were found only after extensive searching. In general, our sites were located in the occasional pools, with lower velocities and more uniform flow than were characteristic throughout most of the study reaches. In some cases, no suitable sites existed within the study reach near its upstream end, and we had to make measurements some distance above the proposed points of diversion in low-gradient meadows, where the first suitable selections could be found. In several cases, we rearranged rocks in the channel to make flow more uniform in the measuring section and thereby increase measurement accuracy.

For most measurements, average velocity was measured at 0.6 depth, although measurements at 0.2 and 0.8 were also made at some verticals to check the 0.6 depth value. In one stream, Big Pine Creek, the best available section near the downstream end of the study reach was characterized by pronounced velocity fluctuations and an irregular vertical velocity profile. Here, velocity was measured at 0.2 and 0.8 depth and, at some verticals, other depths as well; the measurements were rated poor and assigned 12% margins of error. Elsewhere, measurements were rated good, fair, or fair to poor, with assigned margins of error of 5%, 8%, and 10%, respectively (Buchanan and Somers 1969). One measurement (the upstream site on Horton Creek) was rated excellent, with a 3% margin of error. Flow values for Pine Creek were confirmed by results of multiple flow measurements made at nearby sections in connection with a fish habitat study (FERC 1986: appendix B.7).

To include long-term records in our analysis of downstream changes in flow, we examined the historical gaging data for Red Mountain Creek and two other streams (Georges Creek and Independence Creek) in the basin, which had been gaged both at the base of the mountain front and farther downstream on the alluvial fan. These gages consisted of stage recorders at parshall flumes. The LADWP, operator of the gages, provided daily flow values computed from the flume ratings. In this study, downstream changes in flow were computed by subtracting flow at the upstream gage from flow at the downstream gage. Annual hydrographs were plotted to identify periods of diversions so that the analysis would encompass only data from diversion-free periods.

Results and Discussion

Although the methods described above were applied to all study streams, space limitations preclude presentation of all results here. Instead, we focus on results from the largest stream, Pine Creek, and the two smallest, Tinemaha Creek and Red Mountain Creek, because these streams display sharply contrasting geomorphic settings and hydrologic behavior and therefore show the range of variation that may be expected in the study area.

Geomorphic Mapping

The geomorphic maps illustrated the influence of geomorphic features on the riparian corridor width. For example, along deeply incised streams, the riparian zone is narrower because the area of shallow groundwater is restricted; wider riparian zones occur on broader valley floors. Along a given stream, the riparian strip is often wider along fault traces or above constrictions caused by debris fans or glacial moraines. In Figures 3 and 4, we present geomorphic maps for the Pine Creek and Tinemaha Creek–Red Mountain Creek project reaches.

The geomorphic setting of the Pine Creek study reach is complex. Upstream and just off Figure 3 to the left is UMETCO's Pine Creek tungsten mine and mill. (The tailings ponds at the upstream end of the project reach are part of this operation.) Pine Creek flows through a U-shaped, glacial valley underlain by a sedimentary fill. A deep test hole (well 1A) penetrated 115 m of sediments before encountering bedrock. The sediments consisted of alternating strata of low permeability (probably glacial till and debris flow deposits) and higher permeability (probably alluvium and outwash deposits) (Chen and Associates 1982). The more permeable units may serve as conduits for the flow of groundwater (discussed below) that emerges upstream of the valley narrowing produced by the recessional moraine labeled R-4 on Figure 3. The riparian strip width is extremely variable along Pine Creek, ranging over 2 orders of magnitude from 2 to 200 m. Wider zones of riparian vegetation are associated with moraines, debris fans, and springs. Moraines and debris fans, in many cases, act as groundwater dams, creating pools of shallow groundwater (and consequently dense riparian growth) upstream. Of the two named springs, the larger (Carpenter Springs) occurs near the intersection of two previously mapped faults, indicated on Figure 3. The springs also lie in line with the northwest–southwest trending escarpment of Mount Tom, the mountain immediately south of the study reach. If this escarpment is a splay from one of the mapped faults, as we have inferred (Figure 3), the fault may be responsible for the emergence of groundwater at the site of Carpenter Springs. Thus, the setting of Pine Creek is influenced in several ways by both recent glaciation and geologic structure.

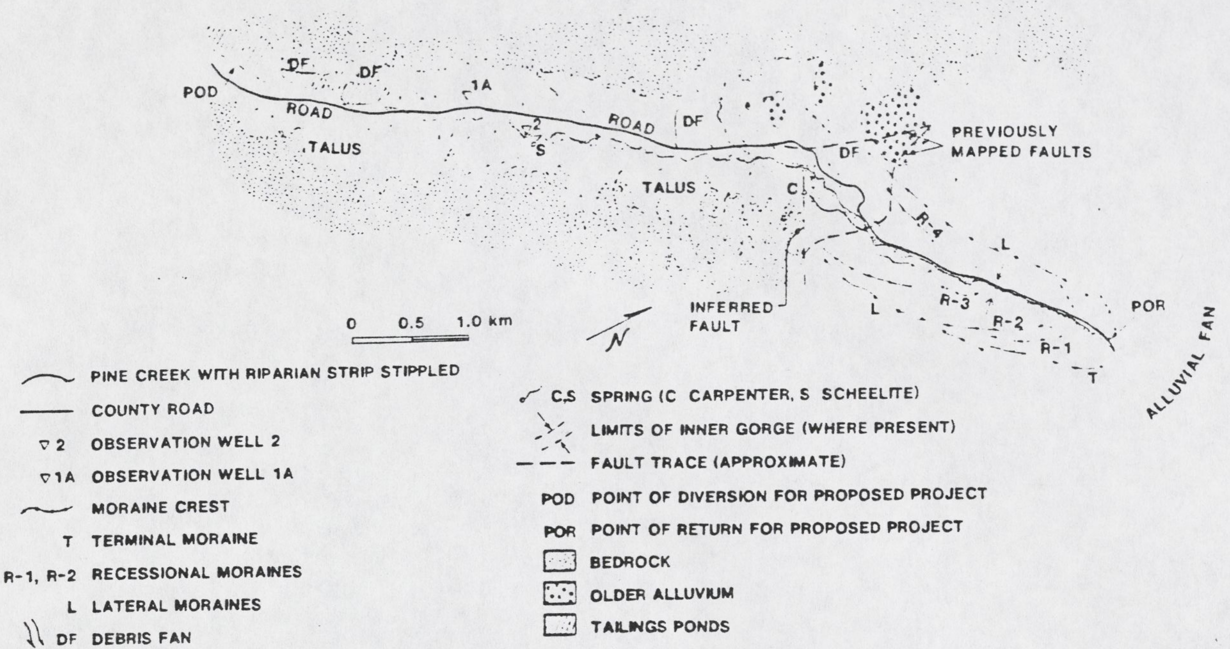

Figure 3. Geomorphic map of Pine Creek study reach, traced directly from aerial photographs (ca. 1973) in the collection of Inyo National Forest. No compensation was made for radial distortion in the photograph, so scale may not be consistent over the entire figure. See the legend for explanation of symbols. Streamflow is from left to right.

In contrast, the reaches proposed for diversion on Tinemaha Creek and Red Mountain Creek are located on alluvial fan deposits whose characteristics are relatively uniform compared with the complex setting of Pine Creek. The riparian belt is narrow and of fairly constant width on both Tinemaha and Red Mountain creeks, although the former has an average flow of about twice that of the latter and also has a wider riparian belt (Figure 4). The riparian strip width varies only from 6 to 41 m along the Red Mountain Creek study reach (except for a 70-m-wide band at the point of diversion). Red Mountain Creek was diverted from its original channel in the 1800s to its present northeastward course across the fan; at this old diversion point, the stream leaves its incised channel and begins flowing across the surface of the fan in an unincised, artificial channel (Figure 4).

Groundwater Levels

The only available well reflecting water table conditions in the riparian zone (UMETCO observation well 2) was located 23 m from the bank of Pine Creek (Figure 1) and was screened at a depth of 3 m (Chen and Associates 1982). In Figure 5, flows in Pine Creek (as recorded at the LADWP gage 7 km downstream) are presented along with water table levels observed by UMETCO personnel in well 2. From these records, it is clear that water table fluctuations followed changes in streamflow, lagging somewhat behind. For example, in 1984 the flow in Pine Creek hit the first of its twin peaks (5.71 m^3/s) on 24 May; water levels in well 2 peaked about 7 June. Groundwater levels in well 2 were 3 m lower than the water surface of Pine Creek when surveyed in August 1985, indicating that the local hydraulic gradient was from the stream into the bank. The water table fluctuations recorded in Figure 5 are within 2 m, so the hydraulic gradient would probably have been toward the bank in the monitored period of 1983–1984. Upslope recharge to the aquifer can be ruled out by the local climatic and geomorphic setting. These observations suggest that, in this reach of Pine Creek, shallow groundwater is recharged by streamflow, and water table elevations respond directly to changes in streamflow.

Synoptic Flow Measurements and Historical Flow Data

Results of synoptic flow measurements are summarized in Table 1. Downstream changes in flow over the study reaches ranged from decreases of 20% to increases of 275%. Margins of error estimated from flow characteristics at measuring sections are generally 10% or less (FERC 1986: appendix B.7). These results indicate that four study reaches experienced a net loss of water to the groundwater (McGee, Rock, Tinemaha, and Red Mountain creeks), one remained virtually unchanged (Horton Creek), and two were gaining water from groundwater (Pine and Big Pine

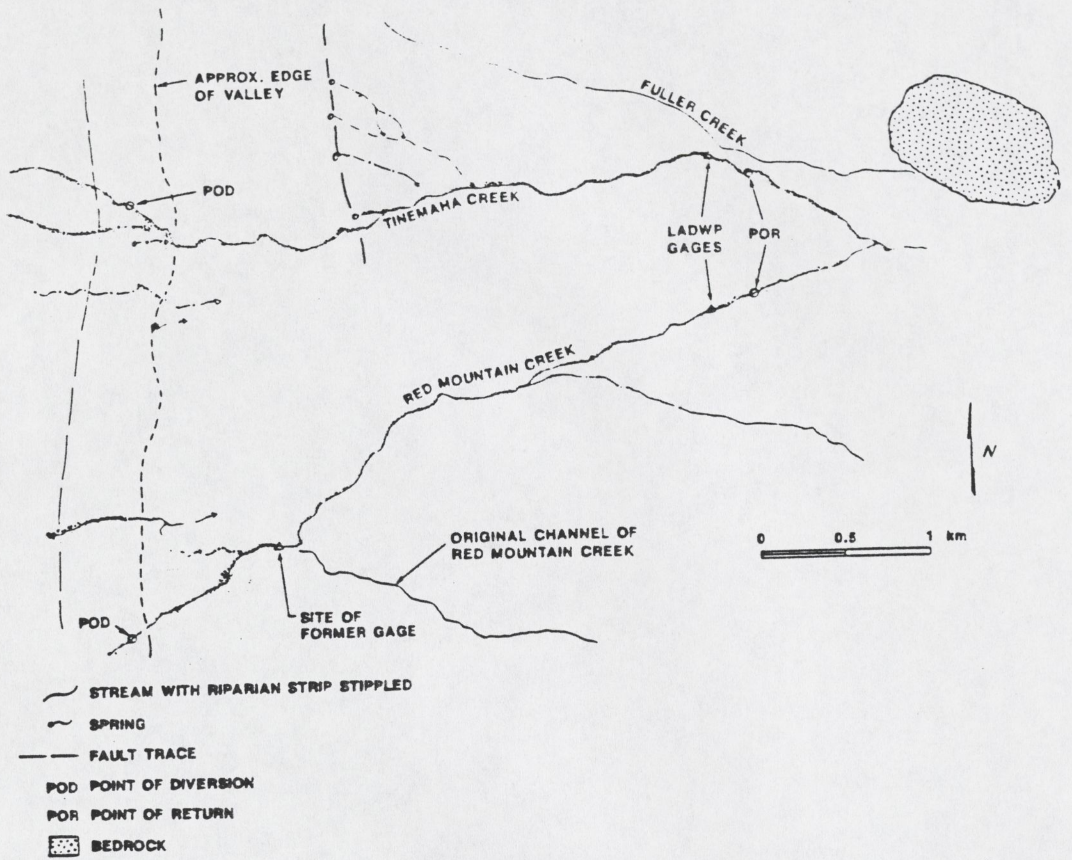

Figure 4. Geomorphic map of Tinemaha Creek and Red Mountain Creek study reaches traced directly from aerial photographs (ca. 1973) in the collection of Inyo National Forest. No compensation was made for radial distortion, so the scale may not be consistent across entire Figure. The symbols used are the same as those in Figure 2, except that geomorphic units are as follows: east of the *dashed line* indicating "approximate edge of valley" is an alluvial fan consisting of alluvium, debris-flow deposits, and glacial outwash deposits (except for indicated outcrop of bedrock); west of the *dashed line* is bedrock of the Sierra Nevada (except for deposits of glacial till and alluvium adjacent to the streams). Flow is from left to right.

creeks). It must be recognized that these results reflect conditions at the time of measurement, whereas historical gaging records show that patterns of gain and loss can vary enormously over months and years. In general, losing streams tend to lose more water during the high flows of the summer snowmelt, and gaining streams gain proportionately more during the low flows of fall through spring (FERC 1986: appendix B.7).

Pine Creek showed the most dramatic downstream changes in flow. The flow increases included contributions from two short, spring-fed tributaries as well as from direct groundwater inflow. The amount of the downstream increase remained quite constant (0.68–0.76 m^3/s) in the measurements presented in Table 1 as well as an earlier measurement in July when flow at the upstream site was 0.88 m^3/s. Because this seasonally consistent increase was added to a progressively declining contribution from upstream, the percentage increase over the study reach rose from 77% in July to 275% in October.

The study reaches on Tinemaha Creek and Red Mountain Creek are typical of stream reaches crossing alluvial fans along the eastern Sierra Nevada front. The fans are composed of relatively permeable sediments with water tables typically at depths of tens of meters (unpublished LADWP well data), although perched water tables may exist immediately under the riparian corridors of the streams. Because of their geomorphic setting, these stream reaches typically lose water to groundwater.

Our measurements on Tinemaha Creek in August indicated a 20% flow loss over the project reach. We did not measure flows on Red Mountain Creek, but we compared flow records for the still active gage near the downstream end of the study reach with flow records for a gage that was operated intermittently through 1983 about 2.9 km upstream.

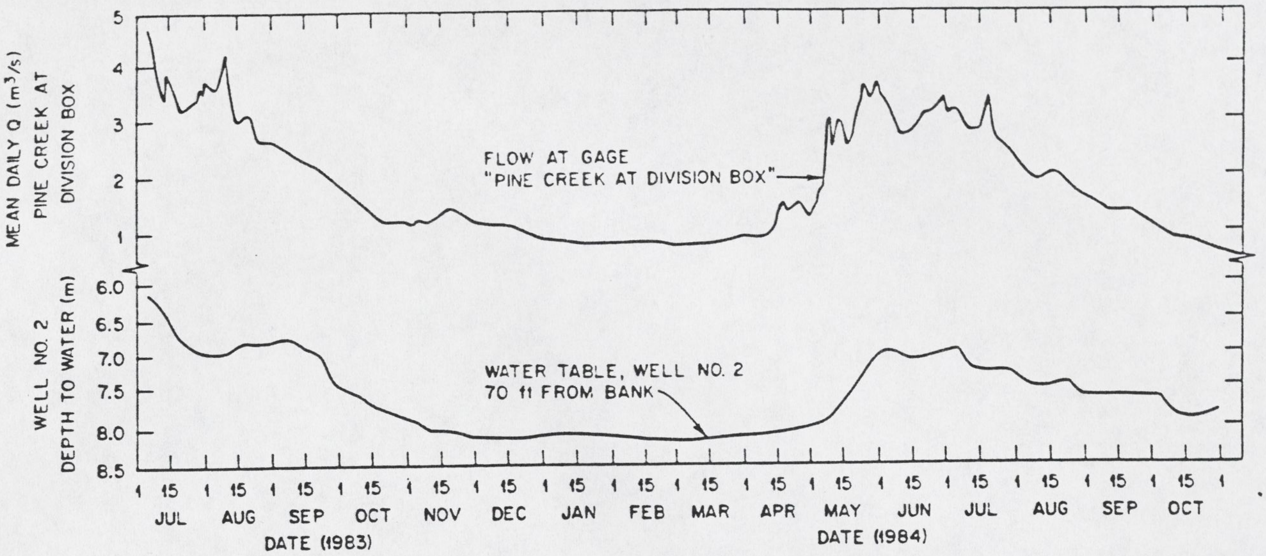

Figure 5. Streamflow (Q) and water table fluctuations, Pine Creek, July 1983–October 1984. Flow data are from Los Angeles Department of Water and Power gage *Pine Creek at division box.* Water table levels from UMETCO observation well 2, screened from 3- to 10-m depth and located about 23 m from the bank abot 6.4 km upstream from the gage. Actual flow values in Pine Creek near the well were probably somewhat smaller than those recorded at the gage, but the pattern should have been essentially the same. Sources of data: Los Angeles Department of Water and Power gaging records and unpublished UMETCO well observation records.

Hydrographs for summer snowmelt and recession limbs in 1977 for the upper gage, "Red Mountain Creek above diversions," and, for the downstream gage, "Red Mountain Creek above Forest Service boundary," are presented in Figure 6. Downstream changes in flow can be read from this plot. Prior to the steep rise in flow from snowmelt in May, flow losses of 0.014 m^3/s (20%) were typical. On the steep rising limb and peak, about 0.04–0.06 m^3/s (20%–30%) was lost between the gages; on the recession limb, flow losses were typically about 0.04 m^3/s (20%). We present data for 1977 because it was an exceptionally dry year and no water was diverted from the stream; all downstream losses can thus be attributed to evapotranspiration and infiltration into groundwater. During years of higher flow, however, water spilled or was diverted between the gages during periods of high runoff. For these other years, we assembled data for the period 1976 through 1983 and plotted hydrographs of mean daily flow at the two gages (as in Figure 6) to identify the onset and cessation of diversions between the gages. We used data points from diversion-free periods only, so the changes in flow between the gages would reflect losses due to evapotranspiration and groundwater infiltration. The downstream gage operated continuously, but in most years the upstream gage operated only during months of higher flow. The requirements for (a) simultaneous operation of both gages and (b) no diversions resulted in a data set of only 32 points for the years 1976 through 1983, with several years represented by only 1 or 2 points.

When flow at the upstream gage is plotted against flow at the downstream gage for these diversion-free periods, Figure 7 is obtained. Flow at the downstream gage, Q_D (m^3/s), is related to flow at the upstream gage, Q_u (m^3/s), by the following regression:

$$Q_D = 0.86\, Q_u - 0.006$$

with $r^2 = 0.97$ and $\alpha < 0.0001$. The regression (solid line) deviates substantially with a 1:1 relationship (dashed line), indicating (a) that flows at the downstream gage are lower than flows at the upstream gage, and (b) that the differences (that is, flow losses) are greater at higher flows. At flows of 0.20 m^3/s at the upstream gage (average for August), flow losses are about 0.034 m^3/s (17%); and at flows of 0.10 m^3/s (average for October), flow losses are about 0.020 m^3/s (20%). Expressed as flow loss per unit stream length, these are 0.008 and 0.004 $\mathrm{m^3 \cdot s^{-1} \cdot km^{-1}}$, respectively. These results are consonant with trends evident on two other alluvial fan streams with similarly paired gages (Independence Creek and Georges Creek). Flow losses between these gages are comparable in magnitude and also exhibit higher losses at higher flows.

These flow losses must be due to some combination of evapotranspiration and losses to groundwater. The months of high flows coincide with the growing season, so some part of this increased rate of loss is

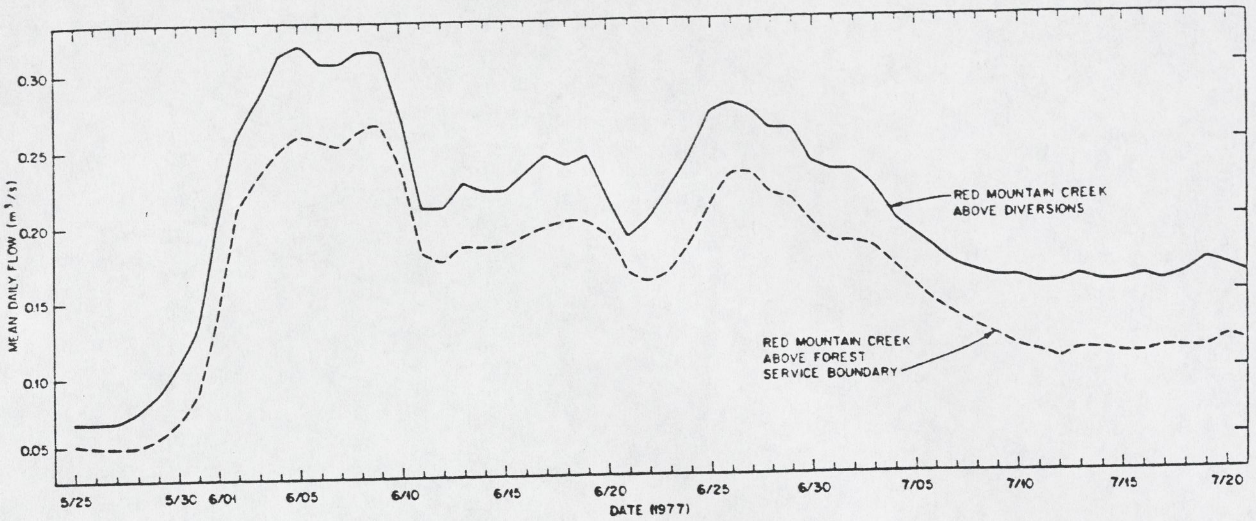

Figure 6. Hydrographs for Los Angeles Department of Water and Power gages *Red Mountain Creek above diversions* and *Red Mountain Creek above Forest Service boundary*, 25 May–20 July 1977. The former was located about 2.9 km upstream of the latter. Source of data: Los Angeles Department of Water and Power gaging records.

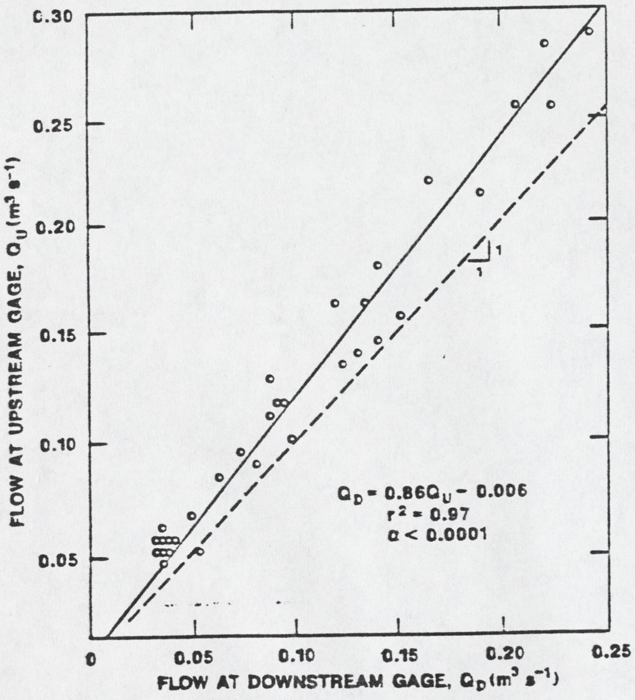

Figure 7. Relation between flow in Red Mountain Creek at the upstream gage ("Red Mountain Creek above diversions") and the downstream gage ("Red Mountain Creek above Forest Service boundary"). The *solid line* is the regression line, and the *dashed line* shows one-to-one relation for comparison. Points on this plot were selected to exclude periods of diversion, thereby reflecting losses to groundwater infiltration and evapotranspiration only. Source of data: Los Angeles Department of Water and Power gaging records.

probably due to higher water use by plants. However, the setting of these streams on highly permeable alluvial fan deposits suggests that the infiltration rate itself is probably higher at high flow. Higher infiltration rates at higher flows could be due to (a) increased wetted perimeter, possibly encompassing more permeable bank materials, (b) a steepened hydraulic gradient from stream to bank, (c) overbank flooding that may deliver water to portions of the floodplain where it is readily absorbed by the ground surface and rapidly taken up by plants, or (d) some combination of these. The available data are inadequate to make more than a few general observations on the probable relative importance of these phenomena.

Eastern Sierra Nevada streams, where they flow across alluvial fans, may be perched tens of meters above the general water table (LADWP unpublished well data). This implies the existence of some deposits of relatively low permeability beneath the streambed that can serve to prevent the entire streamflow from leaking rapidly into the unsaturated alluvial fan deposits, which are probably characterized by high hydraulic conductivities. Sands, silts, clays, and organic materials interstitial to framework boulders and cobbles would probably be adequate to "plug" the streambed.

We would expect the highly permeable deposits underlying a stream to be plugged eventually by fine sediment as sediment-laden water infiltrates into the bed, pulled downward by the hydraulic gradient. Harrison and Clayton (1970) observed the plugging of sands by silt and clay from the suspended load of an infiltrating stream under downwelling conditions and duplicated the phenomenon in the laboratory. Thus, while an initial condition of rapid infiltration can be imagined, eventually the infiltrating waters should deposit enough fine sediment to slow down the infiltration rate, a process of negative feedback analogous to the filter-clogging problem addressed in the environmental engineering literature [see, for example, Yao and others (1971)].

At higher flows, the wetted perimeter of the channel (and probably the hydraulic gradient from stream to banks) would increase, inducing lateral flow from stream to banks through deposits that may be less plugged and thus more permeable than the bed materials. Similarly, overbank flooding during the annual snowmelt may deliver water to floodplain surfaces distant from and above the stream, increasing the moisture content of these soils by a transient pulse of recharging water.

Application of Results to Impact Assessment

Our studies have helped to put the observed distribution of riparian vegetation in geomorphic context, to assess the relative dependence of riparian vegetation upon streamflow along individual study reaches, and to evaluate assumptions about flows and groundwater movement appearing in license applications.

By recognizing losing reaches as generally being more sensitive to diversions than gaining reaches, we could take into account the downstream changes in flow indicated by synoptic flow measurements and thereby lower the expected impact of the proposed diversion for reaches that were shown to be gaining. Our synoptic flow measurements only indicated net flow changes over the entire reach; portions of the reach could behave differently. For example, Pine Creek appears to be locally influent to groundwater at well 2 (Figure 5) despite the profound gaining trend evident over the entire project reach. Such considerations influenced the degree of "credit" granted for gaining in the assessment. Data in Figure 5 were also used in evaluating the statement in the environmental impact documentation for the proposed project that recharge from streamflow affects the bank for a distance of only one-half the channel width away from the bank (Groves Energy Company 1984). Figure 5 shows that the groundwater in well 2 was affected by streamflow at a distance of 23 m from the bank—nearly five times the channel width (about 5 m) at this site.

Flow measurements on Pine Creek also contradicted earlier estimates of the amount of water available for diversion at the proposed project's upstream end. In the environmental impact documentation for the proposed project, data from the LADWP gage (near the proposed project's downstream end) had been adjusted for the smaller drainage area at the upstream site and for differences in precipitation. Based on these calculations, flows at the proposed point of diversion were assumed to be 76% of the flows at the LADWP gage (Keating 1982). During the weeks of high snowmelt flows, this is probably a reasonable approximation. However, during other times of the year, actual flows are quite different. Flow measurements in July, August, and October showed that flows at the upstream site were probably closer to 56%, 36%, and 27%, respectively, of flows at the gage (FERC 1986). Thus, there appears to be considerably less water available at the proposed diversion site than assumed by the applicant, a fact with potentially profound implications not only for riparian vegetation in the reach, but also for the economic viability of the project itself (FERC 1986).

The rate of downstream flow increase observed in Pine Creek is unexpectedly high. It is probable that much of the drainage from the Pine Creek basin moves down the valley as groundwater flow, having emerged as surface flow at a point about 2 km above the gage. The subsurface stratigraphy (alternating units of low permeability with units of higher permeability) could provide conduits for subsurface flow. Accurate prediction of the flow available at the proposed diversion site would require multiple flow measurements in all seasons at nearby cross sections selected for favorable flow characteristics for measurement.

Downstream changes in flow were most pronounced on Pine Creek, but synoptic flow measurements show substantial downstream changes in flow over other study reaches as well (Table 1). This has implications for the design of instream flow monitoring programs. For example, on a losing stream, flow should be gaged near the downstream end of the diverted reach so that, when instream flow requirements are met at the gaging site, they are met over the entire diverted reach. Similarly, on gaining streams, instream flows should be monitored at the upstream end of the diverted reach, where flows can be expected to be lowest.

These methods may prove to be generally applicable to impact assessment of small hydroelectric projects. Simple geomorphic maps can be drawn from aerial photographs, which are readily available for the entire conterminous United States and much of the world. However, the riparian strip may be more difficult to map from aerial photographs in less arid environments because the contrast with nonriparian vegetation would not be as pronounced as in the eastern Sierra Nevada. We were fortunate in this study to have access to unusually extensive streamflow records: this was due to the fact that most of the water in these streams eventually flows, via aqueduct, into the Los Angeles municipal water system. In most parts of the United States, the US Geological Survey commonly maintains gages on larger streams and, more rarely, on smaller ones. In many localities, more groundwater-level data are available than was the case for our study reaches. Land-ownership patterns in the region and the difficulty encountered in drilling wells in the bouldery glacial and debris-flow deposits underlying the project reaches disouraged well drilling here, but suitable observation wells may be more common in other settings. Synoptic flow measurements can be conducted on nearly any stream, provided that measuring sections are carefully chosen (or substantially improved by channel modification when necessary).

Summary and Conclusions

The impacts of streamflow reductions on riparian vegetation depend largely on local hydrologic conditions, which in turn depend on local geomorphic setting. Accordingly, an understanding of site-specific hydrology and geomorphology is prerequisite to assessment of potential impacts. Moreover, flow data collected for impact assessment can provide an empirical basis for estimating flows available for diversion at ungaged sites. Geomorphic and hydrologic studies of seven eastern Sierra Nevada streams proposed for diversion demonstrate the utility of these methods as follows:

1) Geomorphic maps, drawn readily from stereo pairs of aerial photographs, can depict the overall settings of the study reaches and identify geomorphic features influencing local conditions for riparian vegetation. For example, width of the riparian corridor along Pine Creek is highly variable, influenced by bedrock structure and glacial features. By contrast, the Tinemaha and Red Mountain creeks' study reaches cross alluvial fans with comparatively uniform geomorphic characteristics and, as a result, have riparian corridors of more uniform width.
2) Where available, records of fluctuations in water table and stream stage can provide direct indications of the dependency of near-stream water tables upon streamflow. On Pine Creek, well records show that fluctuations in the shallow alluvial water table closely followed streamflow, indicating local groundwater dependence on stream flow.
3) Synoptic flow measurements can identify gaining and losing reaches; the former are regarded as less sensitive to flow reductions because inflowing groundwater may provide moisture for riparian plants. On Pine Creek, flow increases along the 9-km study reach of 180%–275% were measured during the period of July–October 1985. These surprisingly large increases (due to groundwater contributions controlled by local geomorphology) indicated that less water was available for diversion upstream than had been previously assumed, and cast doubt on the economic viability of a proposed hydroelectric project. By contrast, the study reaches on Tinemaha and Red Mountain creeks exhibited flow losses (typically about 20%) into the alluvial fan deposits over which they flowed.
4) On steep mountain streams dominated by boulder cascades, it is often difficult to locate sites with flow characteristics suitable for making accurate flow measurements. However, by searching, we were able to locate, above boulder steps or in alpine meadows, lower gradient sites where flow was more uniform and measurements were possible. For accurate measurements in mountain streams, such sites must be used, even if they are somewhat above or below the study reach itself. Otherwise, flow measurements, if made in steep, highly turbulent sites, are subject to large errors. Patterns of downstream changes in flow may vary substantially from season to season. Because synoptic flow measurements provide only an instantaneous "snapshot" of conditions, they should ideally be repeated over a range of flows and seasonal conditions.
5) Synoptic flow measurements can be conducted on almost any stream, provided suitable measuring sites can be located. Records from permanent gaging stations are less commonly available. Rarer still are streams with upstream–downstream gage pairs, but, where such paired gages exist, they permit tracking over time of flow gains or losses in the reach between gages. Historical records for such a pair of gages on Red Mountain Creek showed that flow losses to evapotranspiration and infiltration typically

ranged from about 16% during low flows of winter to 26% during high flows of summer.

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Riparian Areas: Perceptions in Management

Wayne Elmore and Robert L. Beschta

A Narrow Strip of Land

Until a few years ago, the phrase "riparian zone" was used primarily by researchers and managers in the arid Southwest. Their primary concern was the role of streamside vegetation (phreatophytes) in water loss from streams. Such is no longer the case. Today, throughout eastern Oregon and other parts of the West, people with diverse backgrounds and interests are taking notice of riparian zones for a variety of reasons.

Riparian zones or areas have been defined in several ways, but we are essentially concerned with the often narrow strips of land that border creeks, rivers or other bodies of water. Because of their proximity to water, plant species and topography of riparian zones differ considerably from those of adjacent uplands. Although riparian areas may occupy only a small percentage of the area of a watershed, they represent an extremely important component of the overall landscape (Fig. 1). This is especially true for arid-land watersheds, such as those in eastern Oregon. Even though our comments focus on issues related to riparian zones in eastern Oregon, similar concerns exist for riparian areas throughout the West.

Riparian areas can be the most important part of a watershed for a wide range of values and resources. They provide forage for domestic animals and important habitat for approximately four-fifths of the wildlife species in eastern Oregon. Where streams are perennial, they provide essential habitat for fish and other aquatic organisms. When overbank flows occur, riparian areas can attenuate flood peaks and increase groundwater recharge. The character and condition of riparian vegetation and associated stream channels influence property values. Other values associated with riparian areas, such as aesthetics and water quality, are also important but difficult to quantify.

Complex Riparian Issues Need Open Discussion

Interest of the public, landowners, and natural resource agencies in management of riparian areas is increasing. However, we are concerned that much discussion is misdirected, and that installing permanent instream structures in rangeland riparian areas without changing vegetation management will be counterproductive over the long haul. In addition, we suggest that several important issues that are not being addressed need to be subjected to the rigor of public discussion. Thus, the objectives of this paper are:

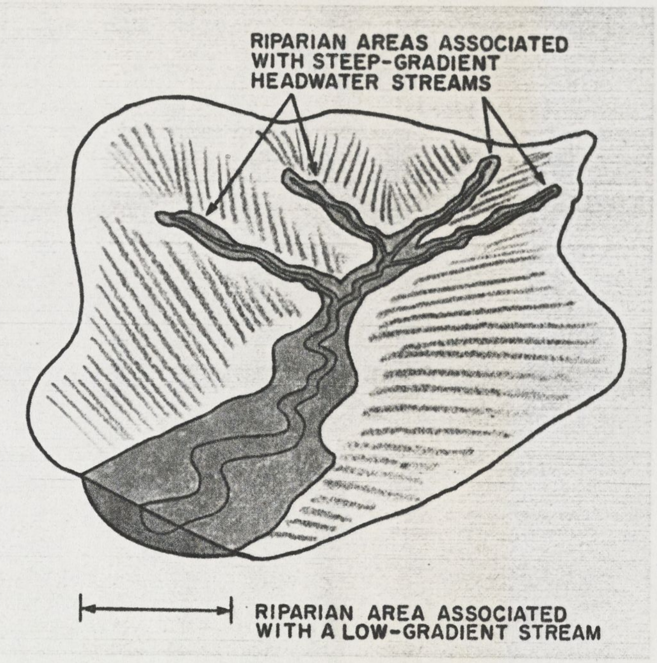

Fig. 1. *Riparian areas along a stream system.*

1. to promote awareness and discussion of riparian issues by and among livestock owners, land managers, environmentalists, biologists and the general public;
2. to identify the characteristics and benefits of productive riparian systems;
3. to encourage managers of public and private lands to reconsider the effects of traditional grazing practices and of recent efforts to control channels structurally.

What are the Problems?

The influence of European man in eastern Oregon's riparian areas began with the influx of fur trappers in the early 1800's. At that time, many streambanks apparently were lined with woody vegetation, such as willow, aspen, alder, and cottonwood. For example, the Indian term "Ochoco," which was used to name a mountain range in central Oregon, means "streams lined with willows." Widespread beaver trapping initiated changes in the hydrological functioning of riparian areas and streams. Beaver ponds, which had effectively expanded floodplains, dissipated erosive power of floods, and acted as deposition areas for sediment and nutrient-rich organic matter, were not maintained and eventually failed. As dams gave way, stream energy became confined to discrete channels, causing erosion and downcutting.

Homesteaders and ranchers followed the trappers. Grazing practices on the rangelands of eastern Oregon were similar to those throughout much of the West and relied primarily on year-long or season-long (April-October) use.

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Riparian Vegetation Instream Flow Requirements: A Case Study from a Diverted Stream in the Eastern Sierra Nevada, California, USA

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ABSTRACT / A methodology is described that allows determination of instream flow requirements for maintenance of riparian trees. Tree-ring data revealed strong relationships between tree growth and stream flow volume for riparian species at Rush Creek, an alluvial stream within an arid setting; these relationships allowed development of models that predict growth rates from hydrologic variables. The models can be used to assess instream flow requirements under the assumption that certain levels of growth are necessary to maintain the population. There is a critical need for development and use of instream flow methodologies for riparian vegetation, since present methodologies focus on needs of aquatic animals (e.g., fish) and may underestimate needs of the entire riparian ecosystem.

Riparian ecosystems are among the most valuable yet most threatened ecosystems in the arid Southwest (Hubbard 1977, Katibah 1984, Arizona State Parks 1989). To protect the few remaining riparian ecosystems, many private and governmental parties have applied for legally guaranteed instream flow rights to maintain fish and wildlife habitat (Gelt 1988). Before appropriate instream flow rights are granted, however, flows that will maintain all components of the riparian system must be known. Although several methods exist for determining flows for the fisheries component of riparian ecosystems (e.g., Morhardt 1986), methodologies are not well developed for other components. For example, instream flow needs of riparian vegetation are not well known, despite the key role of plants in creating fish and wildlife habitat and providing recreational and aesthetic values (Baltz and Moyle 1984, Knight and Bottorff 1984).

Various approaches that have been taken to determine flow needs of vegetation include that of Leighton and Risser (1990), who modeled water needs of trees based on ecophysiological parameters. This model, however, does not relate water use to processes such as growth or survival. Taylor (1982) developed models for alluvial streams in the Sierra Nevada that relate the width of the riparian strand and diversity of species to instream flow, but the models do not provide species-specific data nor do they consider time-lagged influences of antecedent flows (Petts 1985). Changes in species composition and structure on streams with reduced flows also provide evidence that low flows are insufficient for certain types of vegetation (Harris and others 1987, Nilsson 1982), but these relationships have not been quantified. Another possible methodology, and the one used in this study, is to relate instream flows to population growth rates (e.g., tree-ring widths), and then determine what level of growth will maintain the population. One advantage of tree-ring data is its historical component, in that the growth response to past flows is recorded in woody tissue, creating a record that spans the lifetime of the tree.

A gaged or diverted stream with known downstream releases offers opportunities to study the relationship between growth rates and stream flow. Rush Creek, the largest tributary to Mono Lake, flows from the eastern slope of the Sierra Nevada through narrow mountain valleys until it is impounded in Grant Lake Reservoir, from which water is diverted to the City of Los Angeles. Diversion was limited during the first few years after construction of the reservoir (1941), but from 1948 on releases into Rusk Creek were highly variable, ranging from none during drought years to >221,000,000 m^3 [180,000 acre-feet (af)] per year. Since 1984, flows have not dropped below 0.54 m^3/sec [19 ft^3/sec (cfs)], a result of a court order requiring sufficient flows to maintain the stream's fisheries.

The known flow releases and large variation in annual flows at Rush Creek created an experimental situation whereby flows (rate and volume) might be related to growth rates of riparian trees. The primary objectives of this study were to develop models relating

KEY WORDS: Instream flow; Riparian vegetation; Stream diversion; Computer model; Rush Creek.

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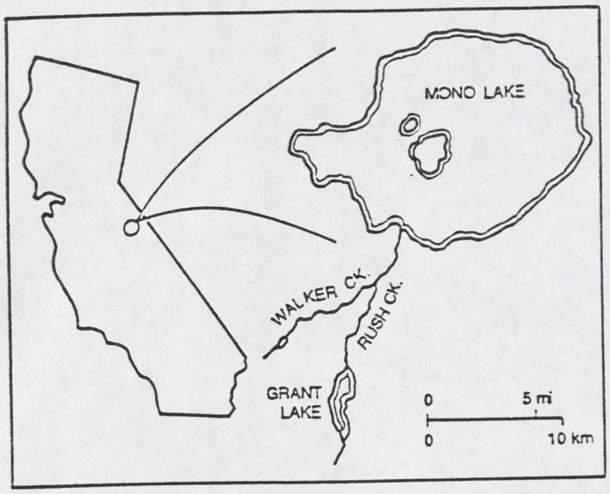

Figure 1. Rush Creek area: tree cores were collected along Rush Creek between Grant and Mono Lakes.

tree growth to hydrological variables and to use the models to determine instream flows for maintenance of riparian trees. Secondary objectives were to contrast instream flow requirements between two riparian tree species and determine whether diversion has altered relationships between flow variables and growth rates.

Methods

A three-step research approach was followed: (1) ring width chronologies were developed for riparian trees; (2) hydrologic chronologies were compiled; and (3) models relating ring widths to hydrologic variables were developed.

Tree-Ring Chronologies

In spring 1988, increment cores were collected from two riparian species in the lower Rush Creek riparian zone. Black cottonwood (*Populus trichocarpa*) was selected because it was the most abundant obligate riparian tree; Jeffrey pine (*Pinus jeffreyi*) because it was the most abundant facultative riparian tree. Cores were taken from 30 mature cottonwoods (>60 years), 13 young cottonwoods (<25 years), and 20 mature pines (>100 years) located throughout the riparian zone to achieve a composite picture of vegetation response. All pines and half the cottonwoods were between Grant Lake and the narrows, the point where Walker Creek enters Rush Creek (Figure 1). The rest of the cottonwoods were between the narrows and Mono Lake. Cored trees were at distances of 0–100 m from the stream edge and at elevations between 2000 and 2140 m.

Cores were mounted following standard procedures (Fritts 1976) and sanded with 300 and 900 grit (12 μm) paper. The untrafine sand paper was essential for distinguishing annual rings of cottonwood, a species that has been used in few dendroecological studies because of its diffuse porous cellular anatomy (Clark 1987). Cores were viewed at 30× to identify annual rings. Potential aberrations, such as missing or false rings, were identified by cross-dating between trees. Annual increment was then measured to the nearest 0.01 mm using an automated pulse counter that inputs data into a computer.

A first step in analyzing tree-ring chronologies is standarizing, that is, fitting a curve to each chronology to remove the growth trend. Growth curves could not be fit to most Rush Creek chronologies, however, because annual growth was so variable (a result of variable annual flows). Instead, the first five to ten annual rings were eliminated from analysis to remove the juvenile growth effect. Chronologies were then standardized to a mean value of one by determining mean growth over the life of the tree (minus the first five to ten years) and dividing actual growth by mean growth. Cores from mature trees were then combined into a mean chronology for each species. A mean chronology was also generated for the young cottonwoods.

Hydrologic Chronologies

Chronologies were developed for two hydrologic variables—stream flow and precipitation. Values of mean annual stream flow in lower Rush Creek were generated for 1910–1986, using three sources of data: (1) flow released into Rush Creek from Grant Lake in 1942–1987; (2) natural flow into Grant Lake in 1935–1941; (3) predicted flow into Grant Lake in 1910–1934, based on regression equations predicting flow into Grant Lake (y) from flow 5 km upstream at Rush Creek Power Plant (x) ($y = 7257 + 1.27x$; $r^2 = 0.89$; $df = 40$). Prior to 1941 flows into Grant Lake from upstream Rush Creek were essentially equivalent to flows below Grant Lake except for some irrigation losses, while after 1941 controlled releases from Grant Lake produced downstream flows. Mean annual precipitation values were generated from: (1) precipitation at a station near lower Rush Creek (Cain Ranch, 2090 m) during 1932–1987; and (2) precipitation predicted for Cain Ranch (y) during 1926–1931, based on values from upstream Ellery Lake (x) ($y = 2.94 + 0.34x$; $r^2 = 0.48$; $df = 54$).

Development of Models

Simple linear regression analysis (SPSS Inc. 1987) was used to determine the relationship between annual growth of cottonwood and pine and the following

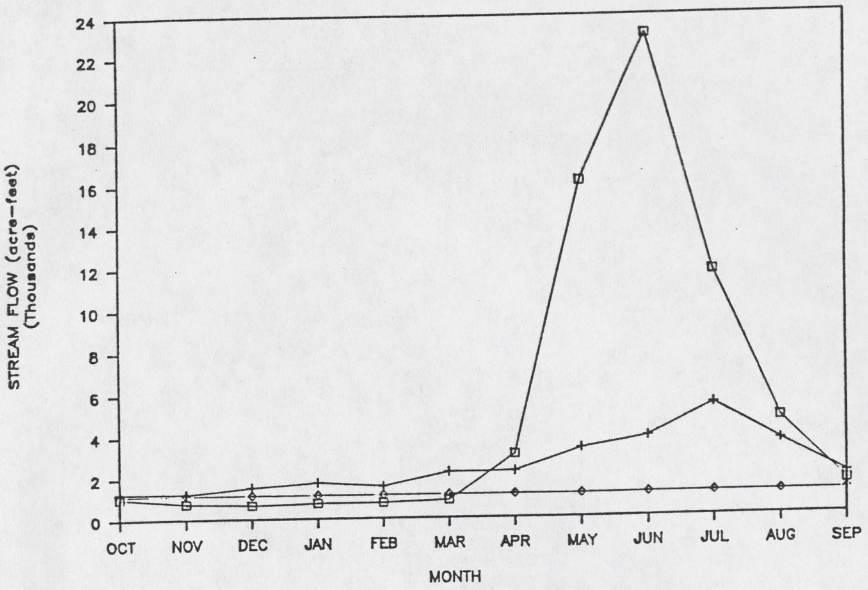

Figure 2. Seasonal hydrograph for lower Rush Creek showing mean monthly flow in the prediversion period (prior to 1947), diversion period (1948–1987), and under a court-ordered flow (minimum flow of 19 cfs). □, pre-diversion; +, diversion; ◇, 19 CFS release.

variables: (1) annual stream flow volume (af per water-year, October–September); (2) annual flow in the prior year (t_{-1}) and in year t_{-2}; (3) cumulative flow for years t through t_{-4}; (4) seasonal flow (October–March; April–June; July–September); (5) monthly flow; and (6) annual precipitation (inches per water-year). (English units were used because they are used by Los Angeles Department of Water and Power and others that manage Rush Creek flows.) Results of these analyses were used to determine whether growth has responded differently in prediversion and diversion times and to select hydrologic variables for development of growth models. Models relating annual tree growth to hydrologic variables were developed using stepwise multiple regression, with the mean pine and cottonwood ring-width chronologies as the dependent (predicted) variables. In addition to these abiotic models, composite models were developed using the added independent variable of prior year ring width. Models were developed for the prediversion period, diversion period, and combined period. Models were not tested on an independent sample of trees, since they were intended to reflect composite response of the riparian populations and were based on a large percentage of the population of surviving trees at Rush Creek. We intend to verify the models in the future by using post-1988 tree-ring data.

After selection of best-fit models, growth of each species was simulated under five flow release scenarios: 10 cfs (7410 af/yr); 19 cfs (14,080 af/yr), minimum flow required by the 1984 court order; 38 cfs (28,160 af/yr), average flows in the 1950s and 1960s; 68 cfs (50,340 af/yr), average flows in the 1930s and 1940s; and 100 cfs (74,100 af/yr). Flows necessary to produce normal growth (defined as average growth during the prediversion period) were determined for both species as an index of instream flow requirements.

Results

Hydrologic Conditions

Prediversion stream flow at Rush Creek averaged 62,000 af/yr, the annual equivalent of a constant flow of 84 cfs. Seasonal flow patterns were typical of eastern Sierra streams, with spring snowmelt resulting in high spring flows and low winter flows (Figure 2). Stream flow during the diversion period fluctuated considerably among years, with average flow about 50% lower than in prediversion times (Figure 3). Peak flows in the diversion period were in July rather than May/June, because of flow release after reservoir filling.

Relations between Hydrology and Growth of *Populus trichocarpa*

The volume of stream flow during the water year (*SF*) was the hydrologic variable with the strongest relationship to growth of *P. trichocarpa;* this relationship explained 66% of the annual variation in ring width during the diversion period (Figure 4). The relationship was linear, with a four- to fivefold increase in flow resulting in a doubling of the annual ring width. Regression equations relating flow to growth of *P. trichocarpa* were nearly identical for prediversion and diver-

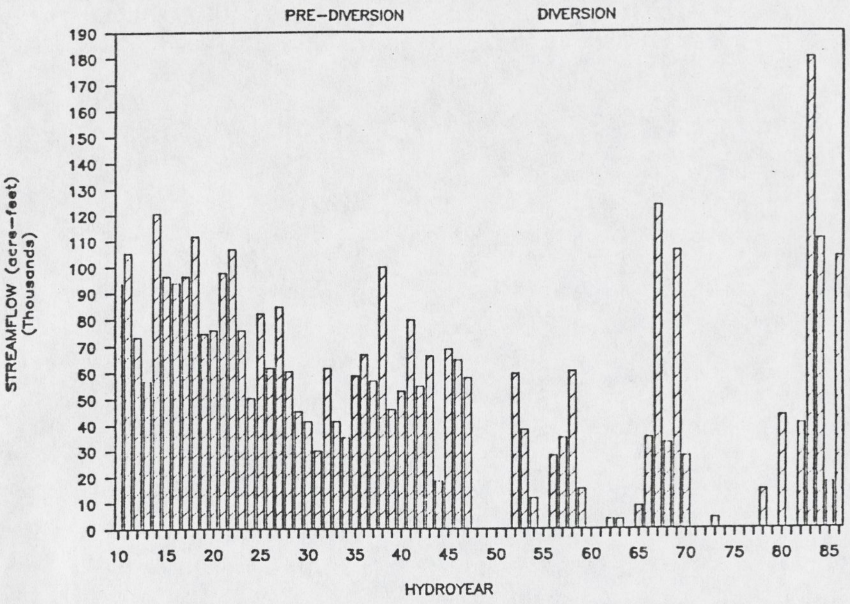

Figure 3. Annual flow (acre-feet) at lower Rush Creek from 1910 to 1986. Diversions from Rush Creek into the Los Angeles Aqueduct began in 1941, with the first major diversion occurring in 1948.

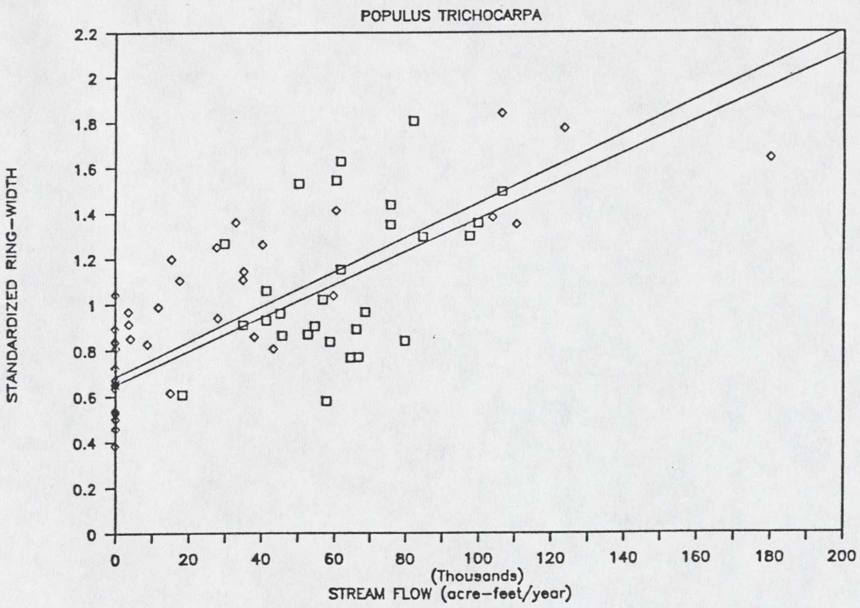

Figure 4. Relationship between annual stream flow (*SF*) in acre-feet and ring width (*RW*) for *Populus trichocarpa* at lower Rush Creek, during prediversion and diversion periods. Regression equations are: $RW = 0.7613[\pm 0.2149] + 0.000006861 [\pm 0.000000817]*SF$ ($r^2 = 0.66$; $df = 38$; diversion period) and $RW = 0.6579[\pm 0.2920] + 0.000007078*SF [\pm 0.000002692]$ ($r^2 = 0.21$; $df = 35$; prediversion). □, pre-diversion; ◇, diversion.

sion times. For example, high flows during the prediversion 1920s resulted in ring widths equivalent to those during the high flows of the 1980s.

A second hydrologic variable, stream flow during the prior year (SF_{t-1}), also was significantly associated with ring widths during both prediversion and diversion periods. A third variable, annual precipitation (*P*), was related to growth only during the diversion period. Together, three hydrologic factors (annual flow, prior year flow, and annual precipitation) explained 71% of the variation in ring width (*RW*) growth of mature *P. trichocarpa* during diversion, and produced the best-fit abiotic model predicting growth: $RW = 0.4947[\pm 0.1280] + 0.0000048721[\pm 0.000001109]$ CFS? $*SF + 0.000002576[\pm 0.000001054]*SF_{t-1} + 0.02157 [\pm .01089]*P$ ($r^2 = 0.71$; $df = 38$; values in brackets indicate ±1 standard error). The effect of the latter two variables was subtle but important. For example, the three-factor model better approximated the dips in growth in 1976 and 1977 than did the one-factor stream flow model, since it took into account the low prior-year flows and low rainfall (83% of normal). *

Prediction of ring width was significantly improved by including a biotic variable, prior-year growth.

Table 1. Relationship between annual ring width and monthly and seasonal stream flow for two riparian species at lower Rush Creek

Time period	Month or season	*Populus trichocarpa* r^2	*Pinus jeffreyi* r^2
Prediversion	April	0.09	0.02
	May	0.27 **[a]	0.20 **
	June	−0.04	−0.03
	July	0.03	−0.05
	August	0.08	0.00
	September	0.17	0.01
	October	0.04	−0.05
	Spring (April–June)	0.10 *	0.08
	Summer (July–Sept.)	0.07	−0.04
Diversion	April	0.36 **	0.11 *
	May	0.28 **	0.24 **
	June	0.33 **	0.18 **
	July	0.37 **	0.04
	August	0.39 **	0.13 *
	September	0.35 **	0.10 *
	October	0.21 **	0.01
	Spring (April–June)	0.37 **	0.20 **
	Summer (July–Sept.)	0.42 **	0.11 *

[a] **Significant at $P < 0.001$; *$P < 0.05$.

Prior-year ring width filled the same function as prior-year flow but provided more predictive power. The best overall model predicting growth of mature *P. trichocarpa* during the diversion period was based on stream flow, prior-year ring growth (RW_{t-1}), and precipitation, and explained 79% of the annual variance in growth ($RW = 0.1072[\pm 0.1506] + 0.000003773[\pm 0.000000956]*SF + 0.4807[\pm 0.1000]*RW_{t-1} + 0.02540[\pm 0.00859]*PR$; $r^2 = 0.79$; $df = 38$). Young *P. trichocarpa* had the same relationships with hydrologic variables as did mature trees, confirming the importance of stream flow, precipitation, and prior-year growth to ring width: $RW = 0.2314[\pm 0.1146] + 0.000003101[\pm 0.000000721]*SF + 0.0303[\pm 0.0091]*PR + 0.02979[\pm 0.00535]*RW_{t-1}$; $r^2 = 0.87$; $df = 13$). The best model for predicting *P. trichocarpa* growth during the prediversion period was based on stream flow and prior-year ring width and explained 61% of the variance in growth.

Seasonal timing of growth, as determined by the amount of variance in growth explained by flow in each season, changed from the prediversion to the diversion period (Table 1). The main growth period for *P. trichocarpa* shifted from spring (May) to summer (July/August), following the shift in seasonal high flows during the diversion period (see Figure 2). Of equal note, *P. trichocarpa* during the diversion period responded to flows in all months during the growing season, in contrast to the vernal growth pattern evident in prediversion times.

Relations between Hydrology and Growth: *Pinus jeffreyi*

Prior-year flows were the hydrologic variables having the strongest relationship with *P. jeffreyi* growth. Although present-year flow was related to ring width (Figure 5), ring widths had stronger relationships with cumulative flows for the past five years ($r^2 = 0.69$) and with prior-year flow ($r^2 = 0.41$). The best-fit model overall for *P. jeffreyi* growth was based on two variables, prior-year growth and present-year stream flow (Figure 6). As was true for *P. trichocarpa*, prior-year growth substituted for prior-year flow as a predictive variable.

The relationship between stream flow and growth of *P. jeffreyi* changed as a result of stream diversion (Figure 5). A given flow in the diversion period resulted in considerably less growth than a similar flow in prediversion times. During both prediversion and diversion, *P. jeffreyi* increased less in growth with increasing flow compared to *P. trichocarpa*, particularly at very high flows (Figure 5).

Pinus jeffreyi retained its vernal growth pattern during the diversion period, despite shifts of peak flows to July. Spring (May) flows contributed most to annual growth for *P. jeffreyi* during both periods (prediversion and diversion) (Table 1).

Growth Simulations for *Populus trichocarpa*

Growth was simulated based on the best-fit model,

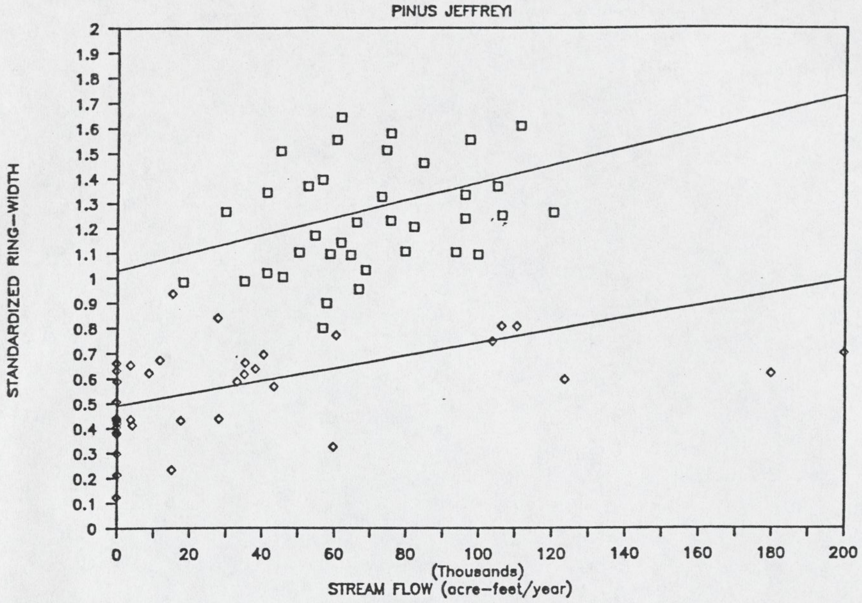

Figure 5. Relationship between annual stream flow (*SF*) in acre-feet and ring width (*RW*) for *Pinus jeffreyi* at lower Rush Creek, during prediversion and diversion periods. Regression equations are: $RW = 0.4800[\pm 0.0312] + 0.000001950[\pm 0.000000639]{*}SF$ ($r^2 = 0.20$; $df = 38$; diversion period) and $RW = 1.0170[\pm 0.0454] + 0.000003291[\pm 0.000000687]{*}SF$ ($r^2 = 0.13$; $df = 35$; prediversion). □, pre-diversion; ◇, diversion.

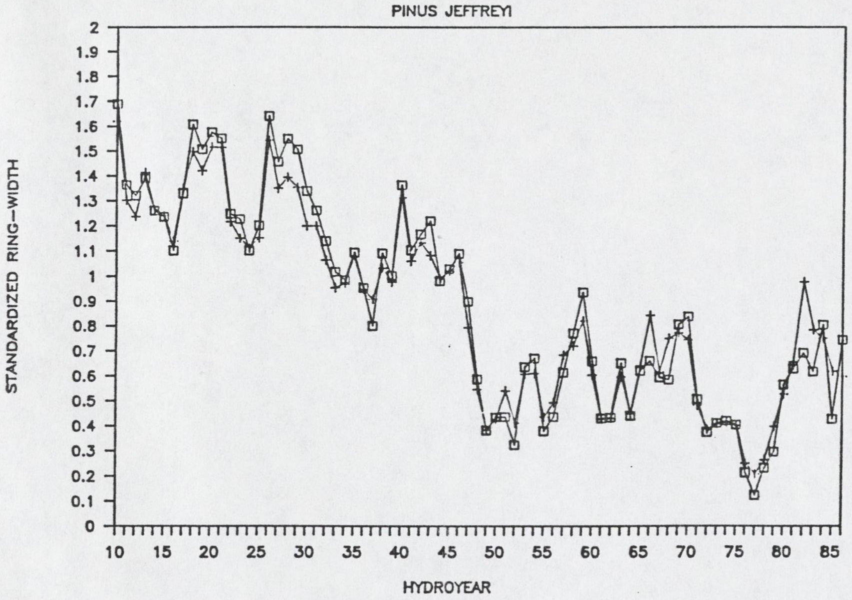

Figure 6. Annual ring-width (*RW*) for *Pinus jeffreyi* at lower Rush Creek showing actual values and values predicted from prior year ring-width (RW_{t-1}) and present year annual stream flow volume (SF). The predictive equation is $RW = 0.0812[\pm 0.0437] + 0.7913[\pm 0.0518]{*}RW_{t-1} + 0.000001926[\pm 0.000000539]{*}SF$ ($r^2 = 0.85$; $df = 76$). □, actual; +, predicted.

the three factor composite model (Figure 7). This model shows that the annual flow necessary to produce normal prediversion growth is 59,000 af/yr (80 cfs), a value nearly equal to average flow during the prediversion period. (Normal growth equates to a standardized ring width of 1.18.) Based on this, the present regime of a minimum flow of 19 cfs will result in below-normal growth of *P. trichocarpa* (Figure 7). Assuming the precipitation regime does not change [i.e., annual mean of 28.5 cm (11.2 in.), standard deviation of 9.8], such a flow will result in population growth ranging from 60% of normal in low rainfall years to 85% in wet years, and averaging 73% of normal. These values are averages for the composite population and may vary for individual trees depending on their location in the riparian zone (e.g., distance from the stream).

The strong relationship between present-year flow and growth of *P. trichocarpa* indicates the ability of this species to rapidly and opportunistically respond to increased moisture. High growth rates in response to high flow releases during wet years (e.g., 1969 and 1983) demonstrate this rapid growth response (Figure 7). Thus, restoration of flows to prediversion levels

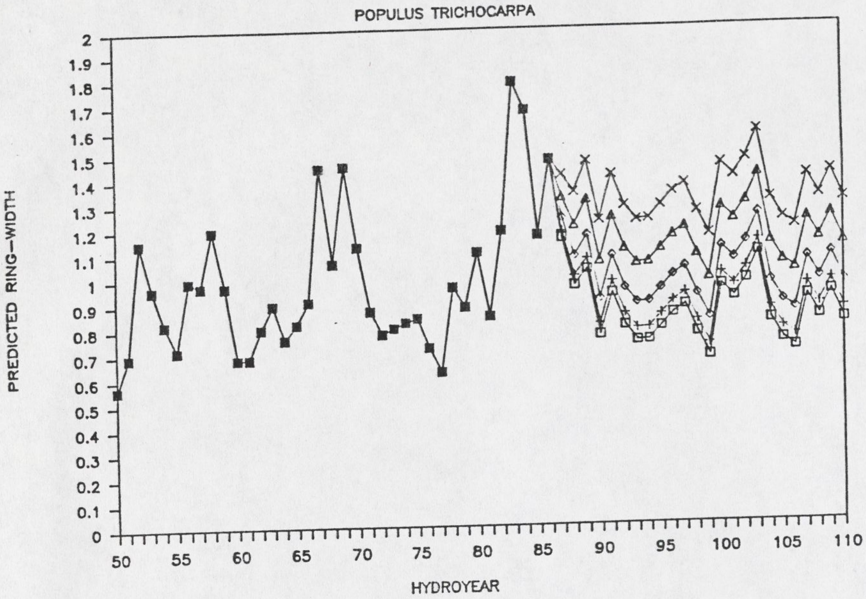

Figure 7. Simulated annual ring widths for *Populus trichocarpa* at lower Rush Creek under five flow regimes: 10 cfs (7410 af); 19 cfs (14,080 af); 38 cfs (28,160 af); 68 cfs (50,340 af); 100 cfs (74,100 af). Rainfall has average mean and variance. □, 10 CFS; +, 19 CFS; ◇, 38 CFS, △, 68 CFS; ×, 100 CFS.

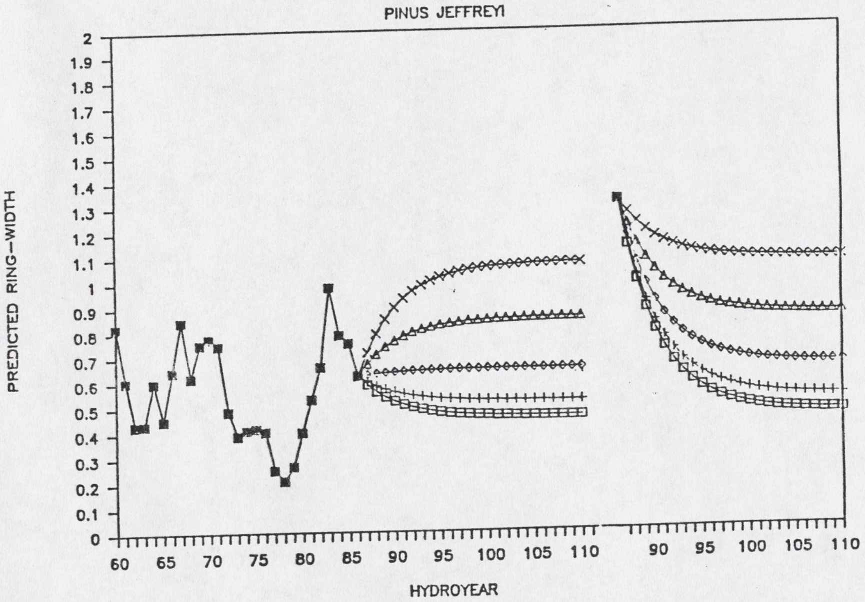

Figure 8. Simulated annual ring widths for *Pinus jeffreyi* at lower Rush Creek under five flow regimes (as in Figure 7). Two simulations are shown. One begins from a growth status equivalent to prediversion growth levels (right side of figure), and the other, to diversion growth levels (left side).□, 10 CFS; +, 19 CFS; ◇, 38 CFS; △, 68 CFS; ×, 100 CFS.

would rapidly restore growth rates of *P. trichocarpa* to normal levels. Growth decline, however, can also occur rapidly in response to low flows such as those in the late 1970s (Figure 3).

Growth Simulations for *Pinus jeffreyi*

The model used to simulate *P. jeffreyi* growth was the best-fit composite model for the combined periods 1910–1987 (Figure 8). Growth of mature *P. jeffreyi* at Rush Creek is predicted to remain well below normal with present flows of 19 cfs, as well as with all simulated flow regimes. (Normal prediversion growth equates to 1.23 on the standardized ring-width scale). The model predicts that restoring flows to prediversion levels will not restore growth to prediversion levels, indicating a loss of growth potential. Flows of 19 cfs will result in growth averaging <50% of normal, while flows of 68 cfs (equivalent to 1930s and 1940s flows) would restore growth to about 70% of normal.

The strong dependency of *P. jeffreyi* growth on prior-year events results in extreme growth declines under sustained low-water conditions, with one year of low growth compounding growth reduction in the next year. This effect is evident in the precipitous declines under the lowest flow simulation, 10 cfs (Figure 8), and in the major declines in *P. jeffreyi* growth during the late 1940s (in response to the first major water diversions) and the 1970s (a period of zero to very low flow releases (Figure 8). Growth recovery during high flow periods (e.g., early 1980s), on the other hand, shows a slow but steady increase, despite fluctuations in flow.

Rate of recovery of *P. jeffreyi* growth depends, in part, on the growth status at the time when water availability changes. Figure 8 shows growth simulated under five different flows, starting from two growth levels: one equivalent to prediversion growth status and the other to stress from 40 years to flow reduction. The endpoints are the same for both simulations, but the rates of change are different. Recovery in response to high flows proceeds at a slow rate for the stressed *P. jeffreyi*, with growth equilibrating only after 10–15 yr.

Discussion

Instream Flows for Vegetation: Refinements and Future Research

This study clearly shows the importance of stream flow volume to growth of riparian trees in one alluvial stream in a semiarid setting and the sensitivity of the tree species to reductions in stream flow. This relationship allows development of models that predict growth rates from flow parameters. These models can form the foundation of vegetation-based instream flow methodologies. Certain parameters, however, need to be refined in this relationship. For example, because the reduction in growth during low flows results from reduced moisture within riparian soils, the relationship of stream flow and growth should be quantified with respect to distance of trees from the stream and height above the water table. Results from this study also point out the need to consider the response of several species as well as several aspects of the flow regime when determining instream flow requirements for riparian communities. In addition to annual volume of flow used in models in this study, seasonal distribution of flow, magnitude of flood peaks, and annual variation in flow are also important (Ward and Stanford 1985). For example, the altered flow–growth relationship and reduced growth of *P. jeffreyi* during the diversion period compared to the prediversion period probably resulted, in part, from the altered seasonal hydrograph. For this species and others having a vernal growth pattern, high spring flows would optimize water-use efficiency.

It is important to understand relationships of stream flow with other plant processes, in addition to growth rate. The relationship between flow and tree mortality, in particular, is critically important to an understanding of effects of flow reduction on plant populations. This relationship is under investigation at Rush Creek, where mortality rates have been high (Stine and others 1984, Stromberg and Patten 1990). Effects of flow reduction on reproductive output also need to be considered, particularly since stress may change reproductive allocation patterns and reduce seed production more than stem growth (Bazzaz and others 1987). Flow requirements for maintenance of mature vegetation must also be integrated with those for seedling establishment, a neglected aspect of flow requirements (Strahan 1990). Establishment of some riparian trees (e.g., *Populus* sp.) may depend on receding flood stages during spring seed dispersal (Fenner and others 1985, Reichenbacher 1984), with timing needs varying among sympatric species depending on their dispersal phenology (Stromberg and Patten 1988). These relationships need to be quantified for *Populus* sp. and other riparian species.

Stream Hydrogeomorphology

This study needs to be replicated in riparian systems that are similar and different in their hydrogeomorphological setting to Rush Creek to determine how common it is for tree growth to be strongly related to stream flow volume. Although the Rush Creek case is somewhat unusual, given the long periods of low flow and the extreme fluctuations in annual flow during the diversion period, the existence of relationships between stream flow and growth in prediversion times (under more natural flow conditions) suggests that the findings are not unique. Other systems have also shown such relationships, such as the Missouri river, where stream flow was related to growth of *Populus deltoides* and other trees (Reily and Johnson 1982). Growth was not reduced, however, for *Alnus rhombifolia* in diverted streams in the western Sierra Nevadas of California (Doyle 1987).

Strong flow–growth relationships may be typical only of certain stream types, such as alluvial, losing reaches in semiarid settings where water limits growth. Methodologies exist for assessing stream types (Kondolf and others 1987; Rosgen 1988), and site characterization is recognized as a useful means of predicting consequences of flow diversion (Gustard 1982, Harris 1988). In reaches such as glacial valleys underlain by shallow bedrock, factors other than flow may be limiting, and high flows may themselves limit growth because of adverse effects of saturated soil. In such situations, flow reduction may have a minimal effect on riparian vegetation (Harris 1988). Integrated biotic and hydrogeomorphological studies are needed to answer these questions.

System-Based Instream Flow Considerations

The Rush Creek riparian system, like others, is an assemblage of biotic components with distinct environmental requirements. When determining instream flow needs of riparian systems, all resources terrestrial and aquatic alike, should be considered. Flow standards should be based on those resources with the highest needs, otherwise the system could degrade, given the interdependency of ecosystem components (Bleed 1987, Cummins 1988, Stromberg and Patten 1989). It is often assumed that if needs of the aquatic resources (i.e., fish) are met, needs of terrestrial vegetation will be satisfied. Results from our study showing the high flow needs of riparian vegetation question the validity of this assumption; a comparison with ongoing studies of fish requirements at Rush Creek should provide an answer. Intuitively, the dependence of riparian vegetation on sufficient lateral flow to wet soils at >100 m from the stream suggests that flow requirements of plants may be greater than those of aquatic in-channel organisms. If true, long-term success of the riparian system could be jeopardized under flows set for aquatic resources. On the other hand, terrestrial vegetation may be able to tolerate periods of no flow (if subsurface water is available) that are intolerable to aquatic organisms. Another resource in the Rush Creek area that could be imperiled under flows considered sufficient for one riparian component is Mono Lake (fed by Rush Creek flow). Interestingly, flows that would maintain Mono Lake at an ecologically safe level (National Research Council 1987) are nearly equal to those needed to maintain riparian tree growth.

*

why not changes in rip. comm.

Realistically, flow requirements cannot be determined for all components of riparian systems. However, it may be possible to adopt an approach taken in certain national parks, wherein changes in population dynamics of index taxa are used as cues of ecosystem degradation or change (Davis 1989). For riparian systems, this would entail identification of sensitive species within each resource group that could be used as index taxa for determining ecologically appropriate flow regimes.

Conclusions

1. A strong relationship exists between growth rates of riparian tree species and annual and prior-year flow volumes within the alluvial, desert riparian setting of Rush Creek. Seasonal distribution of flows also influences growth rates of those riparian species with vernal growth patterns.

2. The relationship between growth and flow can be used as the basis for determining instream flow needs of riparian vegetation, under the assumption that certain levels of growth are needed to maintain the individual and the population.

3. Tree growth–instream flow models for Rush Creek suggest that requirements of terrestrial vegetation may be greater than those of the fisheries. To ensure that instream flows are sufficient for long-term survival of the entire riparian system, flow requirements of the riparian vegetation must be integrated with those of fisheries and other riparian resources.

4. The presence of strong relationships between stream flow and tree growth rates can serve as an indicator of sensitivity, allowing detection of those riparian systems that are most sensitive to flow reduction.

Acknowledgments

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