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DISSERTATION

**FACTORS INFLUENCING SUCCESS OF CUTTHROAT TROUT
TRANSLOCATIONS**

Submitted by
Amy Leigh Harig
Department of Fishery and Wildlife Biology

**In partial fulfillment of the requirements
for the Degree of Doctor of Philosophy
Colorado State University
Fort Collins, Colorado
Spring 2000**

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COLORADO STATE UNIVERSITY

February 15, 2000

WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED
UNDER OUR SUPERVISION BY AMY LEIGH HARIG ENTITLED FACTORS
INFLUENCING SUCCESS OF CUTTHROAT TROUT TRANSLOCATIONS BE
ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY.

Committee on Graduate Work

Melinda J. Gantner

W. K. C.

Patricia L. Kennedy

Kurt D. Fausch

Advisor

Handell Kobert

Department Head

ABSTRACT OF DISSERTATION
FACTORS INFLUENCING SUCCESS OF CUTTHROAT TROUT
TRANSLOCATIONS

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) in the western United States have experienced drastic declines in their distributions, often to <5% of their native range, due to habitat degradation, the introduction of nonnative salmonids, and overharvest. Of the 14 subspecies recognized (three are undescribed), one is extinct, four are listed as threatened or endangered under the Endangered Species Act, and conservation plans have been developed for most others. Cutthroat trout readily hybridize with other spring-spawning salmonids and are apparently displaced by fall-spawning species such as brook char (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*), so many populations are restricted to remote headwater streams and lakes above fish movement barriers that prevent invasion by nonnative salmonids. Isolating barriers, while protecting cutthroat trout populations from nonnative salmonids, restrict them to areas that may be too small or have insufficient habitat to support a viable population.

Establishing new cutthroat trout populations through translocation of genetically pure fish into fishless waters or those treated with toxicants to remove nonnative salmonids is one of the few management strategies for increasing their range. Unfortunately, success rates for establishing self-sustaining fish populations through

translocation are generally less than 50%, and few recovery programs have reviewed their translocation attempts to determine the factors influencing translocation success in aquatic systems. Conservation of greenback cutthroat trout (*O. c. stomias*), one of the listed subspecies, and Rio Grande cutthroat trout (*O. c. virginalis*), a subspecies that has been petitioned for federal listing, has been ongoing for more than 25 years. More than 40 translocations have been conducted for each subspecies, so they represent two of the few cases where recovery efforts for any fish are sufficient to evaluate factors influencing success. My objective is to identify factors that promote establishment and persistence of cutthroat trout populations isolated in headwater streams by comparing attributes of translocations that successfully established a naturally reproducing cutthroat trout population to those that failed.

In Chapter I, I analyzed unpublished data from natural resource agencies on reinvasion of nonnative salmonids, potential limiting habitat factors, and source of translocated fish to identify the attributes of the 14 of 37 translocations of greenback cutthroat trout that were successful, and the probable cause of failure for the other 23. Of the 23 that failed, 11 (48%) were reinvaded by nonnative salmonids, 10 (43%) apparently had unsuitable habitat, and 2 were depressed by other factors. Reinvansion occurred most often because of incomplete removal of nonnative salmonids in complex habitats or failed artificial barriers. Of those not reinvaded, success was highest in receiving waters with at least 2 ha of habitat that previously supported reproducing trout populations.

In Chapter II, I report on a detailed analysis of the minimum habitat required for establishment and persistence of populations from field surveys of stream-scale habitat and map measurements of basin-scale habitat for 27 greenback and Rio Grande cutthroat

trout translocations in Colorado and New Mexico. The best models developed from these data using polytomous logistic regression predict risk of translocation success or failure from stream-scale habitat attributes. These models indicate that cold summer water temperature, narrow stream width, and lack of deep pools limit populations of cutthroat trout. Cold summer temperatures are known to delay spawning and prolong egg incubation, which reduces the growth of fry and likely limits their overwinter survival. Furthermore, small streams with few deep pools may lack the space necessary to promote overwinter survival of a sufficient number of individuals to sustain a viable population. Models of basin-scale habitat were not as effective as stream-scale habitat for distinguishing between successful and unsuccessful translocations of cutthroat trout but indicate that watershed area is useful as a coarse filter for selecting potential translocation streams. Presumably, large watersheds ($> 14.7 \text{ km}^2$) encompass low elevation habitat that provides warmer summer temperatures, and would have relatively wide stream channels of sufficient length to provide an adequate number of deep pools.

This research is one of the few attempts to determine the specific factors influencing translocation success for fishes, and it demonstrates that measuring attributes of local habitat over a whole watershed scale that matches the life history of the organism can be highly useful for identifying critical habitat factors. The models I developed from stream- and basin-scale habitat attributes will allow managers to choose future restoration sites with a high probability of translocation success and to identify whether populations in fragments of historical habitat are likely to persist. A multi-scale analysis of this type may also be applicable to other subspecies of cutthroat trout in central and southern

Rocky Mountain streams (e.g., Colorado River cutthroat trout, *O. c. pleuriticus*) because similar habitat attributes probably limit their populations.

Amy Leigh Harig
Department of Fishery and Wildlife Biology
Colorado State University
Fort Collins, Colorado 80523
Spring 2000

ACKNOWLEDGMENTS

Financial support for this research was provided by the Colorado Division of Wildlife West Region, the U.S. Forest Service Rocky Mountain Region Fish Habitat Relationships Unit at the University of Wyoming, Pike-San Isabel National Forest, and the Trout Unlimited Coldwater Conservation Fund. I am especially grateful to Dave Langlois, Nick Schmal, Mike Young, and my advisor, Dr. Kurt Fausch, for making certain that I was funded throughout my entire degree program.

I would like to thank Dr. Kurt Fausch for his guidance and support over the past 5 years, particularly during my adjustment to motherhood after the birth of my son. I recognize how lucky I was to have such an understanding mentor. I also thank my graduate committee, Drs. Will Clements, Pat Kennedy, Melinda Laituri, and Mike Young, for their contribution to my degree program. My research and dissertation was greatly improved by their advice. This research could not have been accomplished without many field technicians, especially Michael Aaron, Warren Colyer, Stephanie Comfort Johnson, Josh Latterell, and Ryan Smith who persevered through difficult days of monotonous stream surveys, heavy back-packs, long hikes, monsoon seasons, trips to the hospital, and lost thermographs. I am also grateful for the friendship of Elisabeth Harrahy, my office mate and fellow Ph.D. student, for sharing the triumphs and trials of our time at Colorado State University.

DEDICATION

To my husband, Michael, and my son, Cooper.

Your infinite support and inspiration make anything possible.

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CHAPTER I:
FACTORS INFLUENCING SUCCESS OF TRANSLOCATED
GREENBACK CUTTHROAT TROUT POPULATIONS

ABSTRACT

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) have declined drastically due to the introduction of nonnative salmonids, overharvest, and habitat degradation. Conservation of most declining subspecies will include establishing new populations through translocation of genetically pure fish. Recovery of greenback cutthroat trout (*O. c. stomias*) has been ongoing for 25 years, so their translocation attempts provide unique empirical information to guide recovery of other non-anadromous salmonids. I compared 14 translocation projects that established populations of greenback cutthroat trout to 23 that failed, to determine factors that influence translocation success. Of the latter, 11 (48%) were reinvaded by nonnative salmonids, 10 (43%) apparently had unsuitable habitat, and 2 were depressed by other factors (heavy metal pollution or bird predation). Reinvasion occurred most often because of incomplete removal of nonnative salmonids in complex habitats or failed artificial barriers. Of those not reinvaded, success was highest in receiving waters with at least 2 ha of habitat that previously supported reproducing trout populations.

INTRODUCTION

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) in the western United States have experienced drastic declines in their distributions, often to <5% of their native range, due to the introduction of nonnative salmonids, overharvest, and habitat degradation (Gresswell 1988; Behnke 1992; Young 1995). Of the 14 subspecies recognized (Behnke 1992; three are undescribed), one is extinct, four are listed as threatened or endangered under the Endangered Species Act (ESA), and conservation plans have been developed for most others. Several of the latter have been petitioned for federal listing.

Primary management strategies in the conservation of declining cutthroat trout subspecies include identifying and maintaining remnant populations and establishing new populations through translocation of genetically pure fish. However, managers' ability to implement effective translocation projects is currently limited by lack of experimental or observational data on what influences successful establishment of a population. Recovery of greenback cutthroat trout (*O. c. stomias*), one of the listed subspecies, has been ongoing for more than 25 years, and managers have attempted 44 translocations. This represents one of the few cases where recovery efforts for any fish are sufficient to evaluate factors influencing success (Williams et al. 1988; but see Simons et al. 1989). My objective is to analyze such factors for the greenback cutthroat trout translocations, and thereby seek lessons to improve conservation of other non-anadromous salmonids.

HISTORY OF GREENBACK CUTTHROAT TROUT RECOVERY

Greenback cutthroat trout were once abundant in the South Platte and Arkansas river basins of Colorado and Wyoming (Figure 1.1; Behnke 1992), but were decimated after 1860 by overfishing and habitat degradation from mining, railroad construction, and agriculture (Jordan 1891; Wiltzius 1985). Unfortunately, the few early attempts to culture greenback cutthroat trout failed (Dwyer and Rosenlund 1988), so nonnative brook trout (*Salvelinus fontinalis*) and rainbow trout (*O. mykiss*) were introduced widely after railroad lines reached Denver in 1870 (Wiltzius 1985). Greenback cutthroat trout readily hybridize with spring-spawning rainbow trout, are apparently displaced by fall-spawning brook trout and brown trout (*Salmo trutta*; Wang and White 1994), and are more vulnerable to angling than these species (MacPhee 1966; Behnke 1992). Consequently, the remaining greenback cutthroat trout populations are restricted to small remote, high-elevation streams and lakes above fish movement barriers. Many of these habitats are cold and unproductive and undergo extreme flow fluctuations, leading to small, slow-growing trout populations (cf. Young 1995). Thus, the two main biological constraints hampering greenback cutthroat trout populations are invasions by nonnative salmonids and lack of high quality habitat.

Though believed extinct by the 1930s (Green 1937), greenback cutthroat trout were discovered in two streams in 1965 and 1970 and listed as endangered under the ESA in 1973 (USFWS 1998). Listing initiated a formal multi-agency recovery effort to restore the greenback cutthroat trout to a portion of its native range by 2000 (Stuber et al. 1988; USFWS 1998). A primary management objective in recovery is to establish new populations by translocating genetically pure greenback cutthroat trout into fishless

waters or those treated with toxicants to remove nonnative salmonids. Of 44 translocations since 1965, 37 were before 1991, which is long enough to assess whether they were successful in establishing a self-sustaining cutthroat trout population (USFWS 1998). No new translocation projects were conducted during 1991-1994 due to lack of funding, and seven translocations conducted after 1995 are too recent to assess population status.

METHODS

I identify factors influencing success of the 37 greenback cutthroat trout translocations conducted prior to 1995 by comparing translocations that produced self-sustaining trout populations to those that failed. Translocations were considered successful if they established a population that met the four quantitative criteria defined in the greenback cutthroat trout recovery plan (USFWS 1998):

1. Maintain a minimum biomass of 22 kg/ha through natural reproduction;
2. Support at least 500 adult greenback cutthroat trout (>120 mm total length);
3. Produce at least two year-classes by natural reproduction within 5 years; and
4. Be isolated from nonnative salmonids.

The recovery plan also states that more than one population can occupy a drainage fragmented by natural barriers to upstream fish movement if each habitat fragment contains at least 2 ha of habitat and meets the four recovery criteria (USFWS 1998). Although a minimum habitat requirement was applied only to populations in fragmented systems, it could also be used to assess habitat for other greenback cutthroat trout populations (Dwyer et al. 1994). Unfortunately, few data are included in the recovery

plan or other reports to independently evaluate whether the translocated populations meet any of the criteria. Although some data on standing stock and habitat size are included in the recovery plan, none are available for number of adults, success of year classes, or isolation from nonnative salmonids. Therefore, I assume that translocations considered successful in the recovery plan meet all criteria. Translocations were considered unsuccessful if they failed to meet one or more of the criteria.

For each translocation, I identified population status (successful or unsuccessful), type of translocation (into previously barren waters or those treated with chemical toxicants to remove nonnative salmonids), source of translocated fish (hatchery or wild), location and type of fish movement barrier, median elevation, habitat area, and probable cause of failure, if applicable (see Appendix I). Data were collected from the greenback cutthroat trout recovery plan (USFWS 1998), agency reports (Mullan 1973; Behnke and Zarn 1976; Hickman and Miller 1977; Langlois et al. 1978, 1979; Washington 1981; Culver and Lentsch 1982; Culver and Bestgen 1983, 1985; Culver 1983; Davis and Culver 1984; Chart et al. 1986, 1987; Stevens and Rosenlund 1986; Kehmeier et al. 1988; Rosenlund and Stevens 1988, 1990; Kehmeier and VanBuren 1990; Rosenlund et al. 1994; Winters et al. 1995), and personal communications with the Greenback Cutthroat Trout Recovery Team. Some explanatory variables were further evaluated with a two-sided Fisher's exact test (Agresti 1996; S-PLUS 4 1997). This is a conservative statistical test, so P-values were assessed as a measure of their strength of evidence rather than choosing an arbitrary significance level (Agresti 1996). However, translocation sites were not chosen randomly by managers, so statistical results imply only an association between success and significant factors, not a direct cause-and-effect relationship.

RESULTS AND DISCUSSION

Fourteen of the 37 translocations of greenback cutthroat trout were considered successful in establishing populations according to recovery plan criteria (USFWS 1998).

This translocation success rate (38%) is similar to that reported for threatened and endangered birds and mammals worldwide (44% [n=80]; Griffith et al. 1989), and much higher than for one fish (18% [n=191] for Gila topminnow, *Poeciliopsis occidentalis occidentalis*; Simons et al. 1989). However, translocation success was somewhat lower than reported for other non-anadromous salmonids. Of those for which sufficient time was available to assess population status, 80% of Gila trout (*O. gilae*) translocations (n=5; Propst et al. 1992), and 83% of Bonneville cutthroat trout (*O. c. utah*) translocations (n=6; Hepworth et al. 1997) were considered self-sustaining or expanding. Nonetheless, sample sizes were small for both translocation programs, and criteria for defining success may have been less stringent than the greenback cutthroat trout recovery criteria. Of the 23 unsuccessful translocations of greenback cutthroat trout, 11 (48%) were reinvaded by nonnative salmonids, 10 (43%) apparently had unsuitable habitat, and 2 were depressed by other factors (heavy metal pollution or bird predation; USFWS 1998). I address the first two factors in greater detail.

Nonnative Salmonids

Nonnative salmonids are probably the greatest threat to the establishment of cutthroat trout populations because nearly all are vulnerable to natural invasion or human transplant from downstream. Empirical data show that nearly half of translocations into waters with supposedly suitable habitat failed due to reinvasion by nonnative salmonids

(11 of 25; other 12 waters excluded due to unsuitable habitat or other factors). Five translocations were unsuccessful apparently because of incomplete removal of nonnative salmonids during a single treatment with fish toxicants (rotenone or antimycin). Complex habitats such as beaver ponds, springs, bogs, and multiple tributaries were thought to have provided refugia that allowed some fish to survive, because nonnative salmonids were later found concentrated in these areas. The probability of achieving a complete fish kill during chemical reclamation might be improved by treating with toxicants for multiple years and monitoring for nonnative salmonids before introducing the native trout (cf. Rinne et al. 1981; Propst et al. 1992). Nonetheless, one-time chemical treatments were successful in eliminating nonnative salmonids from 13 waters.

Four translocations were thought to have failed when brook trout, brown trout, or rainbow trout breached artificial barriers to fish movement, because the invading species were found just upstream from the barriers. Nonnative salmonids breached rock gabions and wooden dams either by surmounting structures (heights 1-3 m) or through channels eroded around or beneath the barrier. No artificial barrier except an 18-m high reservoir spillway successfully prevented upstream movement of nonnative salmonids, so choosing only natural barriers unlikely to fail could reduce the risk of translocation failure. All other successful greenback cutthroat trout populations were above natural waterfalls, steep cascades, or stream reaches with temperatures too warm for trout survival.

Managers suspect that at least two translocations failed because anglers deliberately reintroduced brook trout. Brook trout in these systems were concentrated around popular campsites and trail crossings, and anglers had threatened to restock brook trout after chemical treatments (B. Rosenlund, U.S. Fish and Wildlife Service, personal

communication). Deliberate introduction of nonnative salmonids by anglers does not have a simple solution. Selecting either remote locations or those that can be closed to motorized traffic reduces potential human intervention, but in the past has focused recovery efforts on small, high-elevation, unproductive habitats. Establishing populations on private property may provide low-elevation, productive habitats in a protected environment (T. Nesler, Colorado Division of Wildlife, personal communication).

Habitat Size and Quality

The 10 unsuccessful translocations not invaded by nonnative salmonids or depressed by other factors were assumed to have failed due to unsuitable habitat. The small size and low productivity of the headwater sites chosen for most translocations suggests that either habitat size or quality may be limiting cutthroat trout populations. Comparison of these 10 translocations to successful ones showed that success in establishing translocated populations was higher in receiving waters with sufficient minimum habitat area (ca. 2 ha) that previously supported reproducing trout populations (Table 1.1), similar to results for birds and mammals worldwide (Griffith et al. 1989).

My analysis combined with information from managers suggests that lakes ≥ 2 ha with inlets or outlets that warm early and have sufficient baseflow for fry rearing have the highest probability of supporting populations. These larger habitats may buffer individual greenback cutthroat trout populations from environmental fluctuations and reduce their risk of extirpation from floods, wildfires, and droughts (Propst et al. 1992). Streams may also be suitable if long enough to encompass sufficient habitat. For example, mean bankfull stream widths from recent surveys of greenback cutthroat trout translocations

indicate that streams at least 5.7-km long are needed to provide 2 ha of habitat at bankfull flows. Only one translocation that failed due to unsuitable habitat was a stream longer than 5.7 km, thus many of the shorter streams might have had insufficient space to sustain adults and juveniles. Native trout translocated to smaller habitats such as these will likely depend on continued management to maintain population persistence (Lubow 1996; Foin et al. 1997).

Translocation attempts in previously barren waters were less successful than projects where nonnative trout were successfully removed using toxicants (Table 1.1), even though many of the latter were subsequently reinvaded. Barren waters may provide high quality habitat if downstream barriers prevented colonization, but it is likely that many lacked trout due to unsuitable habitat. For example, some lakes lacked inlets or outlets with clean gravel that could support spawning, although translocation into one of these lakes was primarily for angling opportunities to engender public support for restoration (USFWS 1998). Translocations to barren waters may also endanger aquatic fauna not adapted to trout predation or competition, such as amphibians or large macroinvertebrates (Harig and Bain 1998, Tyler et al. 1998).

Elevation apparently had little direct effect on translocation success (Table 1.1), although experimental stocking of greenback cutthroat trout in one lake at 3511 m showed they failed to reproduce (B. Rosenlund, U.S. Fish and Wildlife Service, personal communication). Nearly all waters where greenback cutthroat were translocated before 1995 are at high elevation (70% >2800 m, 40% >3000 m; n=37), and within the elevation range of 1900-3500 m where translocations were attempted, there is no apparent relationship between elevation and population status. However, it is likely that low

summer water temperatures, which are strongly influenced by factors other than elevation (Smith and Lavis 1975), may have inhibited reproductive success (Harig and Fausch unpublished). In streams where water temperatures do not reach 4-8°C by early July, greenback cutthroat trout spawning is delayed (Behnke 1992; USFWS 1998) and egg incubation is prolonged, leading to low embryo survival and late hatching (Rinne 1980; Hubert et al. 1994; Stonecypher et al. 1994; Thurow and King 1994). The smaller fry that result risk overwinter starvation if they cannot grow enough to withstand metabolic deficits at low winter temperatures (Hunt 1969; Cunjak and Power 1987; Shuter and Post 1990). Thus, the greater success of translocations of other non-anadromous salmonids (Propst et al. 1992; Hepworth et al. 1997) may be partially explained by their choice of warmer, low-altitude streams.

Source of Translocated Fish

It does not appear that the source of translocated fish contributed to the failure of the 10 translocations thought to be affected by unsuitable habitat because early translocations with wild greenback cutthroat trout were about as successful as later ones with hatchery fry (Table 1.1). Wild, adult fish were used only in eight of the earliest translocations of greenback cutthroat trout (during 1965-1981; three were later invaded by brook trout.) Hatchery fry and juveniles were used in the remaining translocations and to supplement most of the previous populations. Greenback cutthroat trout recovery has relied on hatchery broodstocks to found most new populations because it was believed that large numbers of fish were needed to rapidly establish new populations and that removal of wild fish would threaten wild populations (USFWS 1998).

Although my analysis suggests that translocating wild fish was not more successful at establishing populations than hatchery fry, sample sizes were small (Table 1.1). Translocations of wild individuals have been successful in native trout restoration (e.g., Hepworth et al. 1997) and have been more successful than releasing captive-reared birds and mammals worldwide (Griffith et al. 1989). Other reasons to use wild fish are to avoid the myriad of genetic and disease problems (e.g., whirling disease) associated with captive breeding (Dwyer and Rosenlund 1988; Philippart 1995; Snyder et al. 1996; Cunningham and Daszak 1998), and to replicate pure remnant populations to conserve genetic traits. Populations may be evolutionarily important if they have unique adaptations to marginal habitats (Scudder 1989), such as rapid egg development despite cold water temperatures reported for one remnant population (Dwyer and Rosenlund 1988). In contrast, a criticism of translocations of wild individuals is that too few fish are used to represent genetic diversity in the original population (Stockwell et al. 1996), but additions of relatively few fish in later years could remedy this (Mills and Allendorf 1996).

Conclusions

During the last 25 years, managers have successfully established 14 greenback cutthroat trout populations from translocations of wild and hatchery fish. Comparing these translocations to the 23 that failed suggests that the greenback cutthroat trout recovery program has been most successful in establishing populations in waters that were isolated from nonnative salmonids by natural barriers, had effective chemical treatments not impeded by complex habitats, previously supported reproducing trout

populations, and had at least 2 ha of habitat. These results can help guide future translocation efforts for non-anadromous salmonids, but as previously noted, translocation waters were not chosen randomly so other unmeasured factors may contribute to success. Furthermore, judging the short-term (5-25 years) success of a translocation based on the recovery plan criteria is not the same as assessing long-term population persistence. It was assumed that these criteria would result in persistent populations, but confirming this will require data on population abundance, and size and age structure through time (Rieman and McIntyre 1993) that are not now available.

ACKNOWLEDGMENTS

I thank the many biologists who contributed data, personal experiences, or comments on earlier drafts of this manuscript, especially R. Behnke, W. Clements, P. Kennedy, M. Laituri, and B. Noon (Colorado State University); C. Frissell (Montana State University); B. Rosenlund (U.S. Fish and Wildlife Service); D. Krieger, K. Kehmeier, D. Langlois, M. Gassoway, J. Melby, B. Nehring, T. Nesler, and G. Policky (Colorado Division of Wildlife); P. Guenther-Gloss, K. Matthews, N. Schmal, G. Sloane, and D. Winters (U.S. Forest Service); and B. Shepard (Montana Department of Fish, Wildlife, and Parks). This research was funded by the Colorado Division of Wildlife, the U.S. Forest Service Rocky Mountain Region Fish Habitat Relationships Unit at the University of Wyoming, Pike-San Isabel National Forest, and the Trout Unlimited Coldwater Conservation Fund. I thank D. Langlois, T. Nesler, N. Schmal, D. Winters, and W. Fosburgh and R. Hammel (Trout Unlimited), for administering funding.

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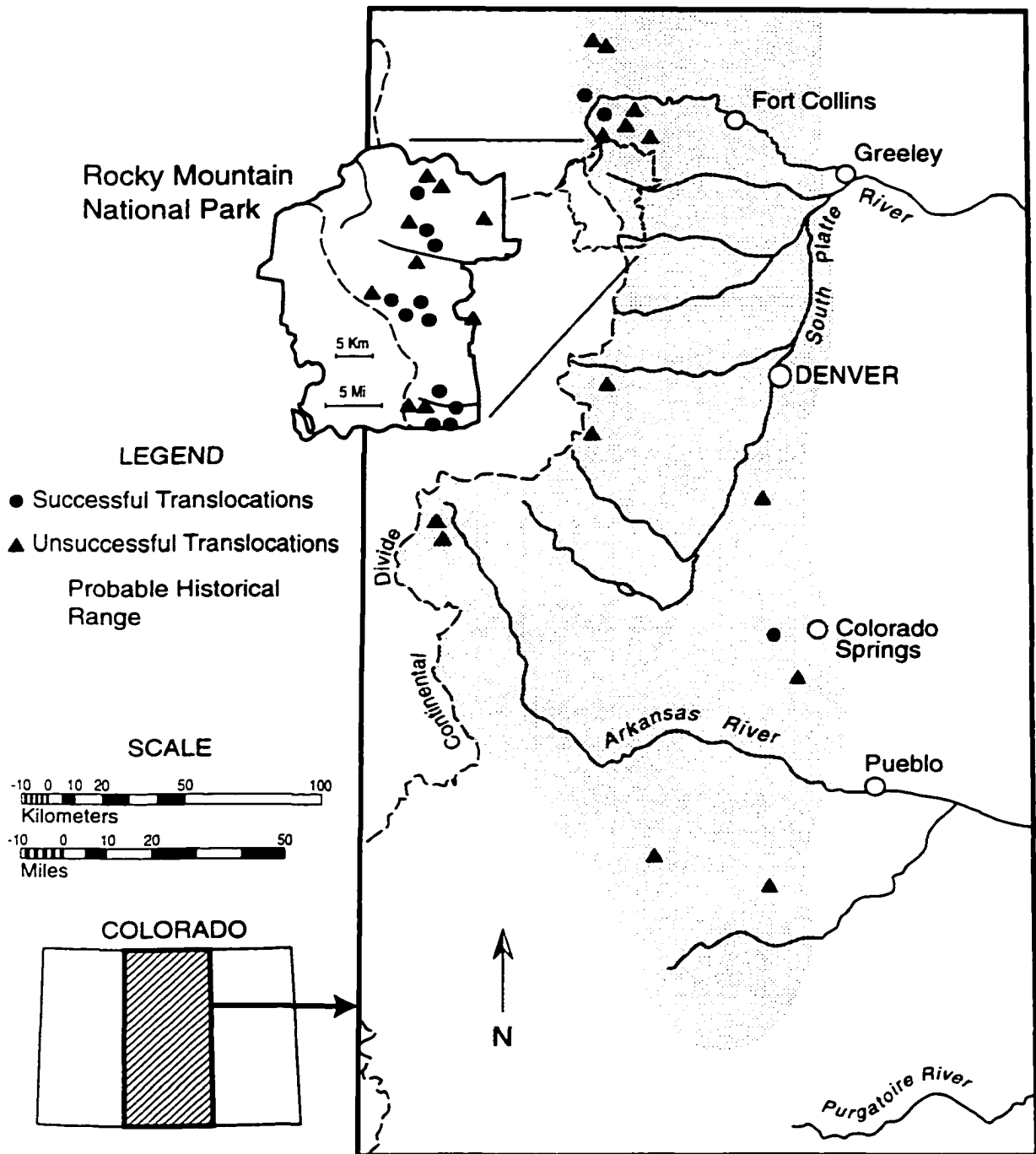
Table 1.1. Success of 24 greenback cutthroat trout translocations as functions of habitat area, previous ability of habitat to support trout, median elevation, and source of translocated fish. Eleven populations subsequently invaded by brook trout and two depressed by other factors were excluded (see footnotes).

Variable	Number of translocations	Percent successful	Fisher's exact test (<i>P</i> -value)
Habitat area of lakes and streams ^a			
<2 ha	10	30%	0.035
≥2 ha	14	79%	
Ability to support trout ^b			
Previously barren	10	40%	0.211 ^b
Supported trout	14	71%	
Median elevation			
<3000 m	9	56%	1.000
≥3000 m	15	60%	
Source of translocated fish			
Wild adults	5	40%	0.615
Hatchery fry and juveniles	19	63%	

^a Total habitat area of stream segments was calculated using mean width (3.5 m) from field surveys of 10 streams where greenback cutthroat trout were translocated (Harig and Fausch, unpublished).

^b Eleven other habitats supported reproducing populations of brook trout and may have supported greenback cutthroat trout prior to reinvasion, yielding a total success rate of 84% ($P = 0.016$ by Fisher's exact test, $n=35$).

Figure 1.1. The distribution in 1999 of translocated populations for which sufficient time was available to assess population status (n=37; see text). Translocations were considered successful (n=14) if they met recovery plan criteria and unsuccessful (n=23) if they were depressed by nonnative salmonids, unsuitable habitat, or other factors (bird predation, mining pollution).



CHAPTER II:

MINIMUM HABITAT REQUIREMENTS FOR ESTABLISHING TRANSLOCATED CUTTHROAT TROUT POPULATIONS

ABSTRACT

Translocation is an important management strategy in a majority of conservation programs for endangered or threatened species, including native cutthroat trout (*Oncorhynchus clarki*) in the western United States. Most subspecies of cutthroat trout have declined to less than 5% of their historical range, and both historical and translocated populations now persist in small isolated fragments of habitat.

Translocations remain one of the few methods available for increasing their range, but success rates are generally less than 50%, and habitat quality or quantity are frequently cited as the cause of failure. Therefore, managers would benefit from a quantitative assessment of specific habitat attributes that contribute to the success or failure of translocations.

I conducted field surveys of stream-scale habitat and measured basin-scale habitat attributes using a Geographic Information System for 27 streams where two subspecies of cutthroat trout were translocated in their native range in Colorado and New Mexico. I developed habitat models using polytomous logistic regression that compare attributes of

sites where translocations established relatively high numbers of cutthroat trout, low numbers of cutthroat trout, and few or no cutthroat trout (absent), to identify macrohabitat features that promote establishment and persistence of native cutthroat trout populations. Models of stream-scale habitat attributes indicated that cold summer water temperature, narrow stream width, and lack of deep pools limit populations of cutthroat trout. Cold summer temperatures are known to delay spawning and prolong egg incubation, which reduces the growth of fry and likely limits their overwinter survival. Furthermore, small streams with few deep pools may lack the space necessary to permit overwinter survival of a sufficient number of individuals to sustain a viable population. Models of basin-scale habitat were not as effective as stream-scale habitat models for distinguishing among translocation sites with high, low, or absent population status, but indicate that watershed area is useful as a coarse filter for separating sites with high numbers of cutthroat trout from those with low or absent status. Large watersheds ($> 14.7 \text{ km}^2$) are expected to encompass low elevation habitat that provides warmer summer temperatures, and have relatively wide stream channels of sufficient length to provide an adequate number of deep pools. These macrohabitat models will be valuable to managers for selecting future translocation sites with a high probability of success and for identifying whether populations in fragments of historical habitat are likely to persist.

INTRODUCTION

Translocation of individuals to establish, reestablish, or supplement a population is often an important management strategy in the conservation of endangered or threatened animals (Griffith et al. 1989). In a review of recovery plans for threatened or

endangered species, 70% of all recovery programs (Tear et al. 1993), and over 80% of programs for fish (Williams et al. 1988), called for translocations. Some highly publicized translocation programs, many of which successfully founded self-sustaining populations (e.g., American bison [*Bison bison*], Kleiman 1989; Peregrine falcon [*Falco peregrinus*], Enderson et al. 1995, Millsap et al. 1998; see also Wolf et al. 1996), have established translocation as an effective management tool to natural resource managers and the general public. However, success rates for translocations of birds, mammals, and fish are generally less than 50% (Williams et al. 1988; Griffith et al. 1989; Simons et al. 1989; Hendrickson and Brooks 1991; Harig et al. in review). A survey of bird and mammal translocations indicates that habitat quality of the translocation site, number of individuals released, and the proximity of the site to the core of the species' historical distribution are three main factors influencing success (Griffith et al. 1989; Wolf et al. 1996, 1998). While these general patterns are useful for identifying areas where research is needed, they do not provide specific information for selecting a translocation site with a high probability of success. For example, the number of individuals needed to establish a self-sustaining population and the factors defining quality habitat may be specific to the taxon being considered for translocation. Unfortunately, most translocations have been inadequately documented, with much of the information unpublished or in reports with limited access, and few have had sufficient population monitoring to identify these factors (Minckley 1995; Hodder and Bullock 1997). Therefore, there is a need for quantitative assessment of specific ecological factors that contribute to the success or failure of translocations.

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) in the western United States have been reduced to a small portion of their historical range, nearly all less than 5%, primarily due to habitat degradation and interactions with nonnative salmonids (Gresswell 1988; Behnke 1992; Young 1995a). Of the 14 subspecies recognized (Behnke 1992; three are undescribed), one is extinct, four are listed as threatened or endangered under the Endangered Species Act, and conservation plans have been developed for most others. Establishing new cutthroat trout populations through translocation of genetically pure trout into fishless waters or those treated with toxicants to remove nonnative salmonids remains one of the few management strategies for increasing their range (Stuber et al. 1988; Young 1995b; USFWS 1998). Analysis of cutthroat trout translocations to identify factors that promote establishment and persistence of populations may improve the success of future translocations and provide an example for assessment of other recovery programs. Of 37 attempts to establish allopatric populations of greenback cutthroat trout (*O. c. stomias*), only 38% were successful, whereas 30% were reinvaded by nonnative salmonids, 27% apparently had unsuitable habitat, and 5% were depressed by other factors (Harig et al. in review). Similarly, 46% of 28 Rio Grande cutthroat trout (*O. c. virginalis*) translocations established naturally reproducing populations, whereas 32% were reinvaded, and 21% had unsuitable habitat (Harig and Fausch 1996 and unpublished data; Alves 1998; Bill Stumpff, New Mexico Department of Game and Fish [NMDGF], personal communication). Cutthroat trout readily hybridize with other spring-spawning salmonids like rainbow trout (*O. mykiss*), and are apparently displaced by fall-spawning species such as brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*; Wang and

White 1994), so reinvasion by nonnative salmonids results in a translocation failure. Isolating barriers, while protecting some cutthroat trout populations from upstream migration of nonnative salmonids, restrict them to areas that may be too small or have insufficient habitat to support a viable population (Moyle and Sato 1991; Moyle and Yoshiyama 1994).

Habitat quality has been emphasized as an essential component of translocation success (IUCN 1987, 1995; Williams et al. 1988; Kleiman 1989; Moyle and Sato 1991; Gordon 1994; Minckley 1995) and was the factor most often cited as leading to translocation failure (Griffith et al. 1989; Wolf et al. 1996, 1998). Success requires that habitat is of sufficient quality to meet the life history requirements of the species (Williams et al. 1988) and large enough to support a population that is self-sustaining in the face of demographic and environmental stochasticity (Moyle and Sato 1991). Some conservation programs for mammals have recommended against using translocations because sufficient habitat is not available (Ruth et al. 1998; Struhsaker and Siex 1998). Research on the minimum habitat requirements of a species may be necessary to identify a suitable translocation site, particularly if those contributing to translocation failure are not apparent (Hodder and Bullock 1997). Some species show metapopulation dynamics or use several spatially segregated sites with different resources in order to persist, so measuring regional spatial dynamics may be needed to identify their habitat requirements (Hodder and Bullock 1997).

There are few empirical data on minimum habitat requirements for entire salmonid populations because ecologists and fish biologists have historically focused on fine spatial scales to understand how environmental factors influence local abundance

and dynamics (cf. Gowan et al. 1994; Fausch and Young 1995; Schlosser and Angermeier 1995). High levels of natural spatial and temporal variation among sites at fine spatial scales can mask factors that influence stream fish populations (Hicks et al. 1991; Dunham and Vinyard 1997; Lohr and Fausch 1997), resulting in habitat models that lack generality for application beyond the stream or watershed for which they were developed (Fausch et al. 1988; Poff and Ward 1989; Rieman and McIntyre 1995). Moreover, fine spatial scales may not adequately reflect the life history of the species being studied (Torgersen et al. 1999; Labbe and Fausch in press). Salmonids and other stream fishes vary markedly in habitat use patterns during their life cycle, requiring appropriate temperatures, flow, substrate, and physical structure at each life history stage (Bisson et al. 1981; Schlosser 1995). These different habitats may be separated (Schlosser 1991, 1995), especially in disturbed watersheds (Fausch et al. 1995), so high spatial heterogeneity and connectivity of habitat patches is needed to maintain persistent salmonid populations. Therefore, analyses that include coarse-scale processes are likely to be most appropriate for stream populations of cutthroat trout.

The goal of my research was to identify macrohabitat features that promote establishment and persistence of translocated native cutthroat trout populations isolated in headwater streams by fish movement barriers. For study, I chose two subspecies of cutthroat trout in Colorado and New Mexico with nearly identical ecological requirements. I compared habitat attributes at stream and basin scales in streams where translocations successfully established a naturally reproducing cutthroat trout population to those where translocated populations were extirpated for reasons apparently related to habitat size or quality rather than invasion by nonnative salmonids. I fit models that

predict risk of translocation success or failure from macrohabitat measured at these two scales, which also provide information about habitat factors that are likely critical for native cutthroat trout isolated in headwater streams. They will be valuable to managers for choosing future restoration sites with a high probability of establishing a successful cutthroat trout population through translocation and for identifying whether populations in fragments of historical habitat are likely to persist.

SALMONID HABITAT AND POTENTIAL LIMITING FACTORS

Stream-Scale Habitat

Based on a literature search of stream salmonid ecology, I made a series of *a priori* predictions about the factors that potentially limit translocated cutthroat trout populations. Translocation sites are generally headwater reaches in high elevation streams isolated by barriers to fish movement, which protects the cutthroat trout population from invasion by nonnative salmonids but restricts them to a small stream segment that may not provide the minimum habitat necessary for population persistence (Moyle and Sato 1991; Moyle and Yoshiyama 1994). Persistent salmonid populations require enough habitat to support a sufficient number of adults to sustain a population, refuge from high flow during spring snowmelt run-off, refuge from low temperature and low flow during winter, optimum summer temperatures to promote spawning, incubation, and emergence prior to the onset of winter, and sufficient clean (i.e., silt-free) gravel to construct spawning redds (Bisson et al. 1981; Behnke 1992). Therefore, I predicted that cutthroat trout persistence is less likely in streams having short length, few pools, small

or shallow pools, pools with little physical structure providing refuges from flow, low winter and summer water temperatures, and little clean spawning gravel.

Habitat for cutthroat trout must provide sufficient space of appropriate quality to support a naturally reproducing population that is large enough to withstand the extirpation risks from environmental catastrophes and demographic stochasticity (Moyle and Sato 1991). In summer, adult salmonids favor pools created by large woody debris, boulders, or lateral scour (Bisson et al. 1981; Flebbe and Dolloff 1995; Young 1996), probably because they provide cover through high structural habitat complexity (Fausch and Northcote 1992; Richmond and Fausch 1995) and abundant food through invertebrate habitat (Angermeier and Karr 1984; Benke et al. 1985). Juvenile salmonids, which are susceptible to displacement by high water velocities (Heggenes and Traaen 1988; Nehring and Anderson 1993; Latterell et al. 1998), tend to occupy run and pool margins (Bisson et al. 1981) and backwater pool habitats characterized by low water velocities, abundant detritus, and abundant invertebrates (Moore and Gregory 1988). Therefore, short stream segments with only small, shallow, structurally simple pools may not provide the minimum habitat necessary to support high numbers of cutthroat trout.

In winter, survival depends on having adequate refugia from low temperature and low flow. Salmonids tend to aggregate in deep pools with low flow velocities and areas of cover (Bustard and Narver 1975; Chisholm et al. 1987; Griffith and Smith 1993) or near sources of groundwater discharge (Cunjak and Power 1986). Thick surface ice resulting from severe winter temperatures threatens salmonid survival if few deep pools are present (Chisholm et al. 1987). Cutthroat trout at high elevations may also be subject to rapid winter temperature acclimation and overwinter starvation, so survival depends on

their ability to attain a body size large enough to withstand metabolic deficits (Hunt 1969; Cunjak and Power 1987). This is particularly important for young cutthroat trout because most metabolic functions are limited by body size (Shuter and Post 1990). Therefore, streams with few deep pools and low winter water temperatures may not be able to support overwintering cutthroat trout.

Cutthroat trout are a spring-spawning species with specific temperature and substrate requirements. Water temperatures below a daily maximum of 4 to 8 °C can delay spawning (Rinne 1980; Thurow and King 1994) and prolong egg incubation, lowering embryo survival and increasing time to hatching (Hubert et al. 1994; Stonecypher et al. 1994). Embryos that hatch late may not be able to attain a body size needed to survive the winter energy deficit (Hunt 1969; Cunjak and Power 1987). Clean, silt-free gravel, predominantly less than 4 cm in diameter, is used for egg deposition (Rinne 1980; Thurow and King 1994). A high proportion of fine sediment in a redd can lead to low fry emergence success, possibly from lack of dissolved oxygen (Weaver and Fraley 1993), which may ultimately reduce juvenile recruitment and influence adult population levels (Scrivener and Brownlee 1989; Beard and Carline 1991). Therefore, streams with low summer temperatures and insufficient clean gravel may not be able to support a naturally reproducing cutthroat trout population.

Basin-Scale Habitat

Stream ecosystems may be viewed as hierarchically organized physical environments consisting of microhabitat, habitat, stream, subbasin, and drainage basin units (Frissell et al. 1986). Each level in the physical hierarchy is controlled by processes

functioning at particular temporal and spatial scales that are nested within the next level of organization. Theoretically, the functional processes that structure the coarse-scale levels of organization i.e., subbasin and drainage basin, also influence habitat at finer spatial scales (Platts 1979; Frissell et al. 1986; Lanka et al. 1987; Poff 1997). For example, watershed area influences the amount of sediment input into the channel and total discharge, slope regulates flow velocity and pool morphology, and aspect affects vegetation and stream temperature (Mark 1975; Dunne and Leopold 1978; Hicks et al. 1991; Moore et al. 1991; Eash 1994). Thus, coarse-scale analyses of attributes that govern vegetative patterns, drainage networks, erosive mechanisms, and fluvial processes (Lanka et al. 1987; Overton et al. 1995) should be useful for predicting the potential habitat quality for stream salmonids (Poff and Ward 1989; Nelson et al. 1992; Rieman and McIntyre 1995; Schlosser 1995; Schlosser and Angermeier 1995).

Based on this theory, I predicted that cutthroat trout persistence is less likely in sites with small watershed area, short channel length, high elevation, high basin relief, high latitude, steep channel slope, and a North-South aspect. Small watersheds and short channels may not be able to support a population large enough to withstand possible extirpation from environmental and demographic stochasticity or catastrophic events (Lande 1993). Translocation sites at high elevation and high latitude or with a North-South aspect may have colder temperature regimes that cannot support overwintering trout or successful reproduction. Finally, streams with high channel relief and steep channel slope may not have sufficient refuge from high flow or enough low gradient pool habitat to support trout, especially juveniles.

METHODS

Study Sites

I selected 27 cutthroat trout translocation streams (12 greenback and 15 Rio Grande) in Colorado and New Mexico for study (Figure 2.1; Table 2.1). These sites represent all but two known stream translocations not invaded by nonnative salmonids of these cutthroat trout subspecies made throughout their historic range prior to 1996 (Harig and Fausch 1996; Harig et al. in review; B. Stumpff, personal communication). I also included one additional stream where a translocation was conducted in 1997 to demonstrate use of the final model (i.e., as a test site). Other translocations conducted during 1996 and after were excluded because they were too recent to assess population status. Translocations into lakes alone were also excluded because lake habitat is not directly comparable to stream habitat.

Before translocation of wild or hatchery cutthroat trout, streams were either barren of fish or had been chemically treated with a fish toxicant (antimycin or rotenone) to remove nonnative salmonids. Unpublished data from natural resource agencies on the frequency, number, size, and source (hatchery-reared, wild broodstock, or wild) of translocated cutthroat trout were incomplete (see Appendix II), but do not suggest that initial stocking practices influenced success of most cutthroat trout translocations. In some streams, cutthroat trout populations were established from introductions of relatively few wild fish (e.g., 182 for San Francisco Creek), whereas others failed to support high numbers of cutthroat trout despite repeated stocking of both hatchery and wild individuals (e.g., five stocking events totaling > 6200 in May Creek). Typically, Rio Grande cutthroat trout populations were founded from one to three translocations

averaging 30 to 1000 wild fish ranging in size from fry to small adults (15 cm). Most greenback cutthroat trout populations were founded from two to five introductions of 400 - 5000 hatchery fry and juveniles (< 8 cm), although small adults were included in some introductions.

All translocation streams were headwater sites above 2400 m in elevation (Table 2.1) and their headwaters extended to elevations as high as 3600 m. Stream habitat generally alternated between steep, forested reaches with conifers in riparian zones, and low-gradient, meandering reaches lined with forbes, willows (*Salix* spp.), or cottonwoods (*Populus* spp.). Mean channel gradients, measured from digital data using a Geographic Information System (GIS), ranged from 7.3 to 20.2% (Table 2.1), but these are likely to be higher than actual gradients due to greater stream sinuosity (and therefore greater channel length) on the ground than can be shown on maps. Length of summer habitat for cutthroat trout ranged from 1.0 to 20.5 km (Table 2.1), and was isolated from encroachment of nonnative salmonids by barriers to upstream fish movement including waterfalls, cascades, steep gradients, dry channels, or manmade structures like rock-filled gabions (Table 2.1). Nine streams (Cony, Fern, Greenhorn, Jacks, Medano, Ouzel, and West creeks, Roaring River, and Rough Canyon/Rhodes Gulch) also have waterfalls or steep cascades in their middle reaches that divide them into multiple habitats. I considered them single populations even though fisheries managers regard four of them as multiple populations because downstream reaches potentially depend on emigration of cutthroat trout from upstream areas.

Field Surveys of Stream-Scale Habitat and Cutthroat Trout Abundance

I conducted field surveys of stream-scale habitat and cutthroat trout abundance along the entire length of each translocation stream from the downstream fish movement barrier upstream to the end of pool habitat, where the bankfull channel width was < 2.0 m and the wetted width was usually ≤ 1.0 m. Surveys were conducted during June through October 1996 through 1998. During each of the first two years, I randomly selected streams from among 29 sites in Colorado where either cutthroat trout subspecies had been translocated and managers reported that nonnative salmonids had not invaded. However, I found that nonnative brook trout had invaded six of them, so in 1998 I randomly selected five additional translocation streams (of seven known; B. Stumpff, personal communication) from the rest of the range of Rio Grande cutthroat trout in New Mexico. One of these later proved to have been translocated after 1995, and so is used as a test of the final model.

I counted the number of fish observed in pool and fast-water channel units (i.e., riffle, run, or cascade; Hawkins et al. 1993) to determine: (1) which reaches were inhabited by trout in the summer, (2) if they had successfully reproduced, based on observing age-0 or age-1 trout, and (3) the minimum abundance of trout. These visual fish counts were not intended as population estimates, but as measures of the minimum number of trout supported by the stream for classifying relative translocation success. The number, species, and approximate size of each fish were determined using polarized glasses by carefully approaching a channel unit from downstream. Cutthroat trout generally hold positions in the open water near the surface (Griffith 1972; Nakano et al. 1998), so are highly visible. However, I used a depth staff to sweep beneath undercut

banks, large woody debris, and boulders to detect any additional fish. Usually, none were found. I also recorded water transparency during surveys to help assess whether the fish count was hindered by turbidity, turbulence, deep water, low light levels, or thick vegetation, and thereby underestimated relative to other streams. However, most streams were small and clear, so cutthroat trout were easily observed if approached quietly.

I surveyed stream-scale habitat at each pool, defined as a channel unit that was at least a $\frac{1}{2}$ channel width long, relatively deep (minimum residual depth of 18 cm) and slow flowing, with a gentle surface water slope (criteria adapted from Hawkins et al. 1993). Stream surveys focused on pool habitat and the structure and spawning gravel they provided because salmonids primarily rely on pools for both summer and winter habitat. I measured bankfull pool width, residual depth, presence of sediment-free substrate, and physical habitat structure for trout (Table 2.2). The number of variables I could include in a model of translocation success was limited by the modest sample size of streams, so I calculated the geometric mean of the number of pools with large woody debris, boulders, and ≥ 0.2 m of undercut bank to create one variable indicating the number of pools with physical structure.

Stream temperatures were measured (± 0.2 °C) for each translocation stream at least every 96 minutes for a minimum of one year using either an Optic StowAway[®] or TidbiT[®] thermograph (Onset Computer Corporation, Pocasset, MA) placed in the largest pool. Thermographs were placed in each stream when first surveyed and replaced each year. Thermal regimes were measured for three years (1996-1999) in 8 streams (i.e., thermographs were first installed in 1996), at least two years in 16 streams, and at least one year in all 28 streams. I analyzed two thermal characteristics; mean daily

temperature for the months of June through August to encompass periods of egg incubation and emergence of cutthroat trout fry, and for the 3-mo. period from December through February to measure mean overwinter temperatures. For four translocation streams that contained more than one thermograph (Medano Creek and tributaries, Rough Canyon/Rhodes Gulch, San Francisco Creek and tributaries, Cony Creek and Lower Hutcheson Lake), daily temperatures were averaged.

Measurement of Basin-Scale Habitat using Digital Data

A Geographic Information System and digital map data were used to measure basin-scale habitat attributes for the 28 greenback and Rio Grande cutthroat trout translocation streams surveyed in the field and all 70 known remnant, historical stream populations not invaded by nonnative salmonids (7 greenback and 63 Rio Grande). Digital elevation models (DEMs) corresponding to 1:24,000 scale topographic quadrangles were provided by the U.S. Forest Service (USFS) Geometronics Unit in Salt Lake City, UT (D. Wolf, USFS) or purchased from the U.S. Geological Survey (USGS). These 7.5-minute grids were in raster format with 30 m x 30 m resolution.

I used the hydrological modeling tools in the GRID[®] module of Arc/Info[®] (ESRI 1995; specific commands are listed in capitals letters) to quantify basin attributes (Table 2.3) from the DEMs using a series of functions that set up the proper DEM conditions and delineated basin boundaries and stream networks. To set up proper conditions, DEMs were converted to elevation grids (DEMLATTICE), sinks were filled using appropriate procedures in GRID[®] (FLOWDIRECTION, SINK, WATERSHED, ZONALFILL, ZONALMIN, FILL) and adjacent quadrangles were joined (MOSAIC).

Sinks, which are areas surrounded by higher elevations, are usually data errors resulting from the spatial interpolation procedure used to create the DEM and can create false stream networks. Lakes were also filled even though they are natural sinks because they were part of cutthroat trout stream habitat and were not modeled as separate basins.

Stream networks were identified from the depressionless grids using the commands FLOWDIRECTION to find the steepest descent from each cell, and FLOWACCUMULATION to calculate the number of upslope cells contributing flow to each cell. Cells with high flow accumulation were used to identify stream channels. I chose cells that accumulated flow from at least 1000 other cells (CON) to delineate streams because this value most closely matched the smallest flowing channels in field surveys. Cells with no flow accumulation were used to identify drainage basin boundaries. I set the downstream end of cutthroat trout basins as the fish movement barrier (SNAPPOUR) to delineate the watershed (WATERSHED). The other data layers, including elevation, stream network, slope (SLOPE), and aspect (ASPECT), were then cropped to calculate watershed area, channel length, latitude, and topographic relief that define characteristics of basin and channel morphology (Table 2.3).

Statistical Models of Translocation Success

I used the predictions described in *Salmonid Habitat and Potential Limiting Factors* to develop four sets of formal *a priori* statistical models predicting cutthroat trout translocation success; one from summer water temperatures, a second from water temperature and stream-scale habitat attributes measured during field surveys, a third from basin-scale habitat attributes measured using digital data, and a fourth from a

combination of the variables in the “best” models of the previous model sets. Each group of models is a nested set that represents formal hypotheses about attributes that potentially limit persistence of translocated cutthroat trout populations (Burnham and Anderson 1998). Summer water temperatures were analyzed as a separate model set to determine which month was the “best” measure to include in models of stream-scale habitat attributes. For this analysis, the 27 streams were assumed to be a random sample of all streams where managers might attempt translocations. Given this, models can be used to predict success of future translocations.

Response variables for the models were based on ranked categories of translocation success (absent, low, high) determined from my visual estimates of cutthroat trout minimum abundance. I assumed that streams supporting relatively *high* numbers of cutthroat trout had minimally sufficient habitat, streams supporting relatively *low* numbers had marginal habitat, and streams *absent* of cutthroat trout had insufficient habitat. Number of dependent variables per model was limited to three because too many variables can result in a statistically unstable model for small data sets (i.e., over-fitting; Burnham and Anderson 1998). Therefore, interaction and quadratic terms were excluded except for the interaction between latitude and elevation in models of basin-scale habitat. All pairwise correlations of independent variables were evaluated to assess multicollinearity.

All models were fit using ordinal polytomous logistic regression (PLR; Agresti 1996; SAS Institute Inc. 1996). Ordinal PLR directly incorporates ordering of response categories, which results in models with simpler interpretations and potentially greater power than ordinary multcategory logit models (Agresti 1996). Model diagnostics

included the Score Test for Proportional Odds Assumption to determine the validity of choosing ordinal over nominal PLR, and Deviance and Pearson goodness-of-fit statistics to examine model fit (SAS Institute Inc. 1996). Overall model significance was reported from the likelihood-ratio statistic for an ordinal test of independence ($H_0: \beta = 0$; Agresti 1996). For specific habitat models where the differences between predicted values for two or more status categories were similar, binary logistic regression was used for pairwise comparison of these subsets of the data.

One or more “best approximating” models were selected from each set of candidate models using Akaike’s Information Criterion corrected for small-sample bias (AICc; Burnham and Anderson 1998). Models were ranked using AICc weights, which is a measure of the weight of evidence in favor of a model given the data. The model with the highest weight is considered the “best” model. However, if there is not one model that is clearly the best, models within two AICc points of the highest weighted model are considered competing models. Results from model-averaging of the competing models based on the AICc weights provides a more precise, stable inference than from only one “best” model (Burnham and Anderson 1998). Model selection based on an information-theoretic approach such as this is superior to traditional hypothesis testing for this dataset because it allows comparison of more than two models at once, balances precision and bias in selecting an appropriate model, and does not require that the data were collected from a formal designed experiment (Burnham and Anderson 1998). It has been successfully used in similar ecological research to define optimal habitat for northern spotted owl juvenile survival (Franklin et al. 2000).

RESULTS

Relative Population Status

The 27 study streams to which cutthroat trout were translocated prior to 1996 were classified based on their population status to compare habitat variables and develop models of translocation success. Population status was assigned based on minimum trout abundance, the number of age-1 and older cutthroat trout observed during the visual survey (Table 2.4). If less than four age-1 and older trout were observed, then it was assumed that the stream was unable to support a population of cutthroat trout, and it was assigned a rating of *absent*. In none of these streams did other factors like turbidity reduce visibility of trout. Streams where less than 100 trout were observed supported a population with relatively low numbers of cutthroat trout and were rated *low*. Streams with more than 100 cutthroat trout were rated *high*. I chose the criterion of 100 trout to separate low versus high populations because streams with counts between 100 and 200 trout and one with a count of 61 (Fern Creek) were likely underestimates relative to other surveys. All but one of these streams had lakes or large beaver ponds, habitats where it was not possible to make accurate visual counts. All other streams with lakes or large beaver ponds had counts greater than 200 cutthroat trout and were ranked as high, so I felt it appropriate to rank the streams with counts between 100 and 200 as high also.

To assess the accuracy of my population status categories, I compared my basin-wide visual counts and subsequent status rating to estimates of standing stock (kg/ha) made by two-pass removal electrofishing (cf. Otis et al. 1978) for shorter reaches of 22 streams for which data were available from natural resource management agencies. My visual estimates, calculated as minimum density of trout (no./km), were positively

correlated with agency estimates of standing stock (kg/ha; $r = 0.70$, $P = 0.003$, $n = 22$), and in 21 of 22 cases yielded similar status ratings to those designated by fisheries managers (Alves 1998; USFWS 1998). The differences in trout abundance estimated by the two methods for individual streams may be attributed primarily to differences in the length and habitat of reaches sampled. My visual sample included the entire stream whereas electrofishing estimates were for shorter reaches (typically 50 – 200 m; e.g., Harig and Fausch 1996) usually near the downstream terminus, that are often not representative of all habitat.

Models of Translocation Success Based on Water Temperature

Winter water temperatures were not included in higher order models because there was no difference in mean daily temperatures from December through February among cutthroat trout streams with absent, low, or high population status ($P = 0.60$ for ordinal test of independence by PLR). The mean daily water temperature for June through August, which encompasses the period of spawning, egg incubation, and fry emergence for cutthroat trout, ranged from 2.0 to 12.9 °C among study streams in June, 4.2 to 14.6 °C in July (Table 2.4), and 4.9 to 13.9 °C in August. Streams where cutthroat trout populations were classified as absent had significantly colder temperatures than streams with low numbers of cutthroat trout, and both had significantly colder temperatures than streams with high numbers of cutthroat trout for all months and combinations of consecutive months ($P \leq 0.005$ for ordinal test of independence by ordinal PLR) except for June ($P = 0.15$). Examination of the AICc values and their weights ranked mean daily water temperature for July-August combined the highest

(0.289), followed by July (0.256), then August (0.175; Table 2.5). July-August temperatures were only slightly better than July temperatures for predicting translocation success, so July temperatures were chosen to represent summer temperatures in all models. This allowed the largest sample of streams to be used because a thermograph in one stream where temperatures were measured for only one year malfunctioned in August. There was no significant difference in stream temperatures among years ($P = 0.15$ for year effect for 16 streams measured two years or more by two-way ANOVA; Figure 2.2), and the average difference among years was only 0.6°C ($\text{SE} = 0.08$), so temperatures measured the year after the habitat survey were used in models.

Models of Translocation Success Based on Water Temperature and Stream-Scale Habitat

Examination of the AICc values and their weights for 20 nested candidate models based on July mean water temperature and stream-scale habitat data collected during field surveys indicated that the model including summer temperature, bankfull pool width, and number of deep pools was the “best” model for predicting success of cutthroat trout translocations ($P = 0.001$ for ordinal test of independence by PLR; Tables 2.5 and 2.6):

$$P(\text{absent}) = \frac{\exp(11.454 - 0.891t - 1.451w - 0.017d)}{1 + \exp(11.454 - 0.891t - 1.451w - 0.017d)}$$

$$P(\text{low} + \text{absent}) = \frac{\exp(14.077 - 0.891t - 1.451w - 0.017d)}{1 + \exp(14.077 - 0.891t - 1.451w - 0.017d)}$$

where P = probability of a stream having a population status of absent, or low and absent combined, t = mean daily water temperature ($^{\circ}\text{C}$) for July, w = mean bankfull width of pools (m), and d = total number of deep pools (residual depths ≥ 30 cm). Point estimates

for probability of a stream being in the low and high categories are calculated as $P(\text{low}) = P(\text{low} + \text{absent}) - P(\text{absent})$ and $P(\text{high}) = 1 - P(\text{low} + \text{absent})$ (Figure 2.3).

Translocation streams with a high or low abundance of cutthroat trout had significantly warmer July water temperatures, greater bankfull widths, and more deep pools than streams with fewer cutthroat trout (i.e., $\text{high} > \text{low} > \text{absent}$; Table 2.4).

The AICc weight for this model was relatively low at 0.286 and was only about twice as likely as the next best model (0.162), so the next three models within two AICc points could be considered competing models (Burnham and Anderson 1998). In this case, a weighted average of the responses from all four models can be used to predict cutthroat trout translocation success, which would provide a more precise prediction than using the “best” model alone (Burnham and Anderson 1998; D. Anderson, Colorado State University, personal communication). However, the four models are similar because they all contain summer water temperature and bankfull pool width (Table 2.6). Moreover, the third variables in three of the models are all measures of the number of pools and are highly correlated with one another ($r \geq 0.95$, $P = 0.001$; see Appendix III). In fact, the predicted values for the “best” model and those from a weighted average of the four competing models differ by only 3%, on average (SE = 0.4; range 0-12%). Therefore, to simplify application by managers, I recommend using only the “best” model to predict cutthroat trout translocation success, recognizing that the variance will be greater than estimated because of the closely weighted competing models.

Models of Translocation Success Based on Basin-Scale Habitat

Examination of the AICc values and their weights for the 20 nested candidate models based on basin-scale habitat measured from digital data using a GIS indicated that the model including watershed area and basin aspect was the “best” model for predicting success of cutthroat trout translocations (Table 2.5). However, basin aspect has the opposite sign in the model from that expected (Table 2.7). Theoretically, basin aspect influences the amount of sun exposure and therefore stream temperature, so basins that drain along an East-West axis would receive more sun and have warmer water temperatures than North-South basins. Although I predicted that streams with high numbers of cutthroat trout would have an East-West aspect compared to streams with no cutthroat trout, my data show the opposite (Table 2.4). Streams with no cutthroat trout have an average basin aspect of 35° from North (SE = 9), whereas streams with low numbers have an aspect of 36° (SE = 7), and streams with high numbers have 22° (SE = 4). However, I believe that this result is spurious because of the influence of four large basins with high numbers of cutthroat trout that trend North-South (Figure 2.4), so I excluded this variable from further consideration.

Because the “best” model is not biologically meaningful and is weighted relatively low (0.201), it is not better at explaining the data than other closely weighted models. Therefore, the next three models within 2 AICc points (watershed area, watershed area and basin relief, and watershed area and latitude) should be considered competing models, and a weighted average of their responses used to predict cutthroat trout translocation success (Table 2.7; see Appendix IV). Although model averaging would provide a more precise prediction of translocation success than using only one

model, none of the three competing models based on basin-scale habitat data could detect differences between absent and low status categories ($P \geq 0.24$ for test of independence by binary logistic regression). However, they were able to detect differences between translocation streams in the high versus absent or low categories, so they are useful as a coarse filter.

The three competing models all include watershed area as a critical habitat factor. and basin relief was correlated with watershed area ($r = 0.42$, $P = 0.03$), so the model of watershed area alone may prove most useful as such a coarse filter for predicting translocation success ($P = 0.008$ for ordinal test of independence by PLR; Tables 2.5 and 2.7):

$$P(\text{absent}) = \frac{\exp(0.251 - 0.123a)}{1 + \exp(0.251 - 0.123a)}$$

$$P(\text{low} + \text{absent}) = \frac{\exp(1.804 - 0.123a)}{1 + \exp(1.804 - 0.123a)}$$

where P = probability of a stream having a cutthroat trout population status of absent, or low and absent combined, and a = planimetric watershed area (km^2). Although this model cannot detect differences between absent and low population status, a graph of the point estimates can be used to identify a minimum watershed area that has a high probability of supporting high numbers of cutthroat trout (Figure 2.5). For example, translocations into watersheds greater than 14.7 km^2 have greater than a 50% chance of establishing high numbers of cutthroat trout.

Models of Translocation Success Based on Stream- and Basin-Scale Habitat

Examination of the AICc values and their weights from 15 nested candidate models developed from combinations of habitat variables used in the “best” models with stream- and basin-scale habitat (i.e., July water temperature, pool bankfull width, number of deep pools, watershed area, and basin aspect) indicated that the stream-scale habitat models described previously explained the data better than models using basin-scale habitat attributes (Table 2.5). Models that included basin-scale habitat attributes were not within two AICc points of the “best” model, except for the one including basin aspect, so none were considered competing models.

DISCUSSION

Stream Habitat Attributes Limiting Cutthroat Trout

Greenback and Rio Grande cutthroat trout were originally distributed throughout the large river systems within their historical range (Figure 2.1; Behnke 1992). However, the streams available for translocations have primarily been small, isolated, headwater sites that provide only marginal habitat at best. Many of these streams have failed to sustain robust cutthroat trout populations despite repeated stocking of genetically pure fish. Even though I cannot determine a direct cause-and-effect relationship between habitat and translocation success without an experimental approach, my field surveys of stream-scale habitat indicate that low summer temperature and habitat size are critical factors limiting populations of translocated cutthroat trout.

My data indicate that many of the greenback and Rio Grande cutthroat trout translocation sites have temperatures that are too cold to support a naturally reproducing

population. Cold summer temperatures are known to delay spawning and prolong egg incubation, resulting in low embryo survival or increased time to fry emergence (Hubert et al. 1994; Stonecypher et al. 1994; Hubert and Gern 1995). Fry that survive and hatch late also may be unable to acclimate quickly enough to rapid drops in water temperature or may starve during winter, so survival may depend on their ability to attain a body size large enough to withstand metabolic deficits (Hunt 1969; Cunjak and Power 1987; Shuter and Post 1990). Therefore, in streams that support low numbers of cutthroat trout, cold temperatures (≤ 7.8 °C mean daily temperature for July) likely prevent successful reproduction and recruitment during most summers. Conversely, in streams that support high numbers of cutthroat trout, summer water temperatures are probably warm enough (mean = 10.0°C; SE= 0.6) to allow successful reproduction in most years.

My stream-scale habitat model also indicates that habitat size and abundance, as measured by mean pool bankfull width and number of deep pools, are factors limiting cutthroat trout populations. Other studies have found similar positive relationships between stream width and trout presence or abundance (Nelson et al. 1992; Clarkson and Wilson 1995; Kruse et al. 1997; Dunham and Rieman 1999). Furthermore, a manipulative experiment that increased the number of deep pools caused an increased abundance of adult trout in six northern Colorado streams (Riley and Fausch 1995; Gowan and Fausch 1996a), and further research showed that the mechanism involved was movement over long stream reaches (Riley et al. 1992; Gowan and Fausch 1996b). Larger streams can support larger populations, which are less vulnerable to environmental and demographic stochasticity (Lande 1993). Large streams are also more likely to provide enough habitat heterogeneity to meet the diverse habitat needs of

salmonids. For example, juveniles may be able to overwinter in relatively shallow pools or runs in low-elevation streams (Bustard and Narver 1975; Griffith and Smith 1993), but adult trout are believed to need large pool habitats to survive the winter (Cunjak and Power 1986; Chisholm et al. 1987). Therefore, translocation streams that were unable to support a persistent cutthroat trout population probably lacked sufficient or appropriate habitat to promote survival of enough individuals to sustain a population.

Three other models of translocation success with similar variables have similar AICc weights, and indicate that total number of pools and pools with physical structure (large woody debris, boulders, undercut banks) also limit cutthroat trout populations. Other investigations reported that measures of total pool habitat and pools with physical structure are associated with trout presence (Nelson et al. 1992; Young 1996), corroborating my models. A weighted average from all four models could be used to predict translocation success with greater precision than the one I chose, but the responses would differ by only 3%, on average. However, it is unlikely that increasing the precision of the probability of translocation success returned by model averaging is worth the extra time, effort, and money needed to measure physical structure in large numbers of pools ($n = 24 - 571$ for my streams).

Appropriate Spatial Scales for Isolated Trout Populations

Studying species distributions over multiple spatial scales to identify the appropriate scale for management has become prevalent in the literature on stream ecology (e.g., Allan et al. 1997; Lohr and Fausch 1997; Armstrong et al. 1998; Torgersen et al. 1999; Labbe and Fausch in press), partially due to the emergence of landscape

ecology (Wiens 1995). Investigations of resident stream salmonids have also expanded in spatial scale since the recognition that a substantial portion of fish in these populations are not sedentary but move, often over great distances (Gowan et al. 1994; Fausch and Young 1995; Young 1996). Traditional habitat studies at the microhabitat scale (see Fausch et al. 1988 for review) have not been able to develop general principles to guide management of fishes, so a coarser-scale approach to stream habitat may be warranted (Schlosser and Angermeier 1995).

I researched two scales of habitat for cutthroat trout, stream and basin scale, which roughly correspond to patch (i.e., local habitat characteristics) and landscape scales (e.g., habitat area) in the ecological literature. Stream-scale models of translocation success indicated that summer water temperature, pool width, and deep pools were critical factors limiting cutthroat trout populations, but models of basin-scale habitat were not as effective for distinguishing between successful and unsuccessful translocations. In a review of studies on a wide array of taxa that considered both patch- and landscape-scales for detecting species presence and abundance, Mazerolle and Villard (1999) found that landscape variables were significant predictors in more than half the studies (59%), but patch characteristics were significant in nearly all the studies (93%). Results are similar for salmonids. Coarse-scale geomorphic variables are good predictors of presence or abundance when they are measured at the reach level (approximates patch scale; e.g., Lanka et al. 1987; Fausch 1989; Clarkson and Wilson 1995). For example, channel slope was identified as a predictor of trout occurrence for stream reaches in the central Rocky Mountains (Chisholm and Hubert 1986; Kozel and Hubert 1989; Kruse et

al. 1997), but mean slope of the entire stream was not important in my models of basin-scale habitat for predicting cutthroat trout translocation success.

One basin-scale habitat attribute that was useful as a coarse filter for translocation success was watershed area. Based on model results, watersheds larger than 14.7 km² are predicted to have greater than 50% probability of supporting high numbers of cutthroat trout. This result is not surprising considering the wealth of studies, most recently for investigations into metapopulation dynamics (Peltonen and Hanski 1991; Thomas et al. 1992; Wenny et al. 1993; Rieman and McIntyre 1995; Dunham and Rieman 1999), that support the species-area relationship, i.e. the probability that a species will be present in a habitat patch increases with increasing area (MacArthur and Wilson 1967; Diamond 1975). I presume that large watersheds encompass lower elevation habitat that also provides warmer summer water temperatures for cutthroat trout, and have relatively wide stream channels of sufficient length to provide an adequate number of deep pools. It is also probable that large watersheds have sufficient input of large woody debris and boulders to provide physical structure in pools. Only 1 of 6 streams where translocations failed have watershed areas > 14.7 km², but 8 of 13 streams with high populations have basins ≤ 14.7 km², so estimating the probability of translocation success in watersheds ≤ 14.7 km² requires measuring stream-scale habitat throughout the basin in the field. For example, five of the eight latter streams have lakes or large beaver ponds, which likely increase the probability of supporting high numbers of cutthroat trout. Lakes are detectable at the basin-scale, but beaver ponds are ephemeral and not usually marked on maps. Therefore, attributes measured strictly from maps would not be useful for predicting persistence of cutthroat trout populations in small watersheds.

Landscape-scale habitat variables other than habitat area have been useful for some species with large spatial habitat requirements such as mammals with large home ranges (Mladenoff et al. 1999) and birds with high dispersal capabilities (Ganey et al. 1990; Bellamy et al. 1998), but the landscape scale must be properly defined with respect to the taxon under investigation (Mazerolle and Villard 1999). The appropriate scale for predicting the attributes limiting cutthroat trout translocation success is determined by the ecological process being considered (Schlosser and Angermeier 1995), in particular the life history of the cutthroat trout. Greenback and Rio Grande cutthroat trout historically existed in large connected watersheds, which had high habitat heterogeneity that met the diverse habitat required by different life stages and were large enough to support their movements (Young 1995a). However, most populations of these cutthroat trout subspecies are now isolated in small watersheds (5 – 78 km²) and cannot move to a new area if their habitat needs are not met. Isolating cutthroat trout in small basins has removed this mobile component of their life history and reduced their spatial habitat use. Therefore, basin-scale analyses may be too coarse to identify whether an isolated habitat can support cutthroat trout, particularly if many critical habitat attributes are locally controlled. For example, water temperatures in small, mountain streams are difficult to predict without site-specific data (Smith and Lavis 1975). Models of stream temperature are usually complex combinations of stream aspect, riparian vegetation, air temperature, fluctuations in stream discharge, and meteorological data (e.g., Moore 1967; Morse 1970; Smith and Lavis 1975; Bartholow 1993). I found weak correlations between basin-scale measures of elevation or latitude and July water temperatures measured in the field

($r = -0.42$ and -0.41 , respectively, $P \leq 0.03$; Figure 2.6), but neither variable significantly improved upon the model using watershed area alone. Basin aspect was important in the basin-scale models, but the relationship with stream water temperatures was opposite that expected ($r = -0.25$, $P = 0.21$; Figure 2.6), supporting the view that its inclusion was a spurious statistical result. Therefore, measuring site-specific variables that would be accurate surrogates for stream temperature would require much more effort and money than simply measuring temperature directly using thermographs.

Probability of Long-term Persistence

The short-term (5-25 years) success of a translocation, based on the minimum abundance of cutthroat trout during my survey period, does not necessarily ensure that this population will achieve long-term population persistence (e.g., > 100 years; Reiman and McIntyre 1993). It is assumed that populations established through translocation will be long lasting, but this may not be true for streams supporting low numbers of cutthroat trout. Small populations are at greater risk from demographic and environmental stochasticity (Propst et al. 1992; Lande 1993; Reiman and McIntyre 1993), so it is likely that in many streams with low numbers of trout the population will eventually be extirpated. Furthermore, basin-scale models of cutthroat trout translocation success were unable to detect differences in streams with no trout versus those with low numbers. This suggests that these streams are geomorphically more similar to one another than to streams with high numbers of trout. As such, I suspect that numbers of cutthroat trout in streams with low status may be more likely to dwindle rather than increasing towards the high status category.

Management Implications

This research identifies critical habitat attributes that allow successful translocation and at least short-term persistence of cutthroat trout populations in isolated headwater reaches. Managers and biologists can use these habitat models to evaluate potential translocation sites and identify current populations at greatest risk from extirpation. For example, stream habitat was measured in Powderhouse Creek, a site where the translocation was too recent (1997) to assess success. The watershed is only 10 km², so my basin-scale model is not useful as a coarse filter. However, the stream-scale model for Powderhouse Creek, which has a mean daily July water temperature of 10.0 °C, mean pool bankfull width of 2.2 m, and 5 deep pools, predicts a 33% probability of supporting no trout, a 54% probability of supporting low numbers, and only a 13% chance of high numbers of cutthroat trout (Figure 2.3). Given this, managers will need to decide whether establishing a small population of cutthroat trout is worth the time, money, and effort of a translocation project, considering that it may not achieve long-term persistence.

In contrast, the basin-scale model of translocation success may be useful for identifying current populations at the greatest risk of extirpation. The habitat attributes necessary to establish a population through translocation may not be identical to those that have sustained these historical populations, but managers have limited time and budgets to devote to stream surveys. A coarse filter such as minimum watershed area could be used to prioritize streams. At last count, there were 50 streams with historical, remnant populations of Rio Grande cutthroat trout in New Mexico that remained free from nonnative salmonids (NMDGF unpublished data). Almost half (24) of these 50

populations are in watersheds $\leq 14.7 \text{ km}^2$ in area and may warrant first attention (Figure 2.7). Populations in many streams have not been surveyed in 10 or more years and may be at risk of extirpation, particularly if invading nonnative salmonids are reducing their available habitat. Similarly, 4 of 13 historical populations of Rio Grande cutthroat trout and 4 of 7 historical populations of greenback cutthroat trout in Colorado also persist in small watersheds $< 14.7 \text{ km}^2$ and may be at risk of extirpation from insufficient habitat. None of these smaller watersheds include lakes, which may reduce the probability of extirpation, so field surveys of stream-scale habitat could identify if they have appropriate summer water temperatures, and are wide enough and have a sufficient number of deep pools to support a high abundance of cutthroat trout.

Despite the low success rates for translocations of fishes (e.g., Simons et al. 1989, Harig et al. in review), my research is one of the few attempts to determine the specific factors influencing translocation success (cf. Williams et al. 1988). It demonstrates that measuring attributes of local habitat over a whole watershed scale that matches the life history of the organism can be highly useful for identifying critical habitat factors (Torgersen et al. 1999; Labbe and Fausch in press). The models I developed from stream- and basin-scale habitat attributes will be valuable tools for fisheries managers concerned with the conservation of Rio Grande and greenback cutthroat trout, particularly if included in an active management program that tests and refines these models with data from recent and future translocation sites. Moreover, results from a multi-scale analysis of this type may also be applicable to other subspecies of cutthroat trout in central and southern Rocky Mountain streams (e.g., Colorado River cutthroat trout, *O. c. pleuriticus*) because similar habitat attributes probably limit their populations.

ACKNOWLEDGMENTS

This research was cooperatively funded by the Colorado Division of Wildlife (CDOW) West Region, the U.S. Forest Service (USFS) Rocky Mountain Region Fish Habitat Relationships Unit at the University of Wyoming, Pike-San Isabel National Forest, and the Trout Unlimited (TU) Coldwater Conservation Fund. I thank D. Langlois and T. Nesler (CDOW); M. Young, N. Schmal, and D. Winters (USFS); and W. Fosburgh and R. Hammel (TU) for administering funding and providing helpful comments. This manuscript was improved from reviews by M. Young (USFS); and W. Clements, P. Kennedy, and M. Laituri (Colorado State University [CSU]).

I also thank J. Alves (CDOW); W. Colyer, S. Comfort, K. Harrington, P. Jensen, J. Latterell, and R. Smith (CSU); C. Brandon, K. Kurtpatrick, Y. Paroz, B. Stumpff, P. Wilkinson, and J. Zamora (NMDGF); M.K. Watry (Rocky Mountain National Park); G. Long, P. Geunther-Gloss, B. Sanchez, G. Sheppard, S. Swift-Miller, and R. Thibedeau (USFS); and C. Kennedy and B. Rosenlund (U.S. Fish and Wildlife Service) for assisting with field stream surveys. I owe special thanks to F. Bunch and the Great Sand Dunes National Monument, K. Czarnowski and Rocky Mountain National Park, T. Ryland and the Forbes-Trinchera Ranch, the landowners along San Francisco Creek, J. Binkley, and S. Barker for permission and assistance in surveying cutthroat trout streams on their land. D. Wolf (USFS Geometronics Unit) generously provided digital elevation data for many Colorado locations; S. Mahabir and D. Anderson (CSU) provided advice on statistical analysis; and M. Laituri (CSU) assisted in developing the GIS methodology.

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Table 2.1. Characteristics of the 28 study streams where cutthroat trout were translocated, including subspecies (GB = greenback cutthroat trout, RG = Rio Grande cutthroat trout), location (NF = National Forest, RMNP = Rocky Mountain National Park), latitude (degrees N), longitude, elevation of the lower stream terminus, stream length, mean stream gradient, and potential barrier to fish movement. Latitude, longitude, minimum elevation, and stream gradient were identified using a GIS. Stream length was the amount of stream habitat available in summer, calculated from a USGS 1:24,000-scale quadrangle as the distance from the lower fish movement barrier to the upper terminus of the field survey. Potential barriers are described as height above the water surface (h), reach gradient, or reach length (l). Heights of waterfalls and artificial structures were measured in the field (estimated for waterfalls > 7 m), whereas reach gradient and length were measured from a USGS 1:24,000-scale quadrangle.

No	Stream	Sub-species	State	Location	Latitude	Longitude	Minimum	Stream	Mean	Potential barrier to fish movement
							elevation (m)	length (km)	gradient (%)	
1	Benito Creek	RG	CO	Rio Grande NF	37° 56' 30"	106° 48' 08"	3281	2.6	13.3	Steep gradient (~ 36%)
2	Cony Creek	GB	CO	RMNP	40° 11' 30"	105° 35' 27"	2904	4.9 ^a	10.5	Steep cascades (gradient ~ 25%)
3	Cottonwood Creek	GB	CO	Pike-San Isabel NF	38° 06' 00"	105° 31' 14"	2470	5.3	10.8	Dry stream channel (l > 1.0 km)
4	Doctor Creek	RG	NM	Santa Fe NF	35° 46' 07"	105° 42' 19"	2520	3.5	20.2	Waterfall (h = 1.4 m)
5	East Middle Creek	RG	CO	Rio Grande NF	38° 19' 07"	106° 16' 45"	2939	3.5	14.8	Steep gradient (~22%)

Table 2.1. Continued.

6	Fern Creek	GB	CO	RMNP	40° 20' 38"	105° 40' 13"	2818	2.4 ^a	17.0	Waterfall (h ~ 10 m)
7	Greenhorn Creek	GB	CO	Pike-San Isabel NF	37° 55' 23"	105° 01' 44"	3107	3.4	16.6	Series of falls (h ≥ 2.1 m)
8	Hourglass Creek	GB	CO	Arapaho-Roosevelt NF	40° 34' 60"	105° 38' 36"	2861	2.8	12.3	Fish weir (h = 1.0 m)
9	Jacks Creek	RG	NM	Santa Fe NF	35° 49' 39"	105° 39' 22"	2531	11.0	11.5	Waterfall (h = 1.9 m)
10	Little Medano Creek	RG	CO	Great Sand Dunes National Monument	37° 47' 59"	105° 31' 04"	2588	3.1	16.5	Dry stream channel (l = 1.7 km)
11	Little Ute Creek	RG	CO	Private land	37° 33' 49"	105° 26' 29"	3262	2.2	16.0	Series of falls (h ≥ 3.0 m)
12	May Creek	GB	CO	Arapaho-Roosevelt NF	40° 36' 26"	105° 47' 44"	2718	4.7	12.2	Steep gradient (~22%)
13	Medano Creek and tributaries	RG	CO	Great Sand Dunes National Monument	37° 45' 59"	105° 30' 21"	2502	20.5	13.2	Dry stream channel (l > 1.0 km)

Table 2.1. Continued.

14	Nabor Creek	RG	CO/ NM	NM State Wildlife Area and private land	36° 58' 37"	106° 37' 39"	2545	5.6 ^a	12.7	Rock gabion (h = 1.4 m)
15	North Fork Big Thompson River	GB	CO	RMNP	40° 30' 11"	105° 35' 60"	3293	1.8 ^a	13.6	Waterfall (h = 1.6 m)
16	Ouzel Creek	GB	CO	RMNP	40° 11' 39"	105° 38' 17"	3159	1.0	17.3	Series of falls (h ≥ 3.0 m)
17	Pecos River	RG	NM	Santa Fe NF	35° 56' 13"	105° 33' 28"	3178	5.8	8.4	Waterfall (h ~ 9 m)
18	Powderhouse Creek	RG	NM	Carson NF	36° 51' 58"	105° 16' 20"	2928	4.3	10.6	Wooden dam (h = 1.5 m)
19	Rio Cebolla	RG	NM	Santa Fe NF	35° 57' 16"	106° 40' 17"	2495	6.7 ^a	8.8	Earthen dam and culvert (h = 4.0 m)
20	Rio de los Pinos	RG	CO	Rio Grande NF	37° 04' 55"	106° 29' 47"	3206	3.1	12.8	Waterfall (h ~ 15 m)
21	Roaring River	GB	CO	RMNP	40° 25' 53"	105° 38' 22"	2881	6.9 ^a	16.4	Steep cascades (gradient ~ 49%)

Table 2.1. Concluded.

22	Rough Canyon/ Rhodes Gulch	RG	CO	Rio Grande NF	37° 23' 15"	106° 24' 35"	2738	4.9	13.4	Mining pollutants in receiving waters
23	San Francisco Creek	RG	CO	Private land and Rio Grande NF	37° 40' 02"	106° 19' 57"	2400	12.4	9.1	Water temperatures too warm for cutthroat trout
24	Sheep Creek	GB	CO	Arapaho-Roosevelt NF	40° 39' 37"	105° 44' 45"	2828	11.6	9.5	Steep cascades (gradient ~ 45%)
25	Unknown Creek	RG	CO	Rio Grande NF	37° 55' 39"	106° 42' 28"	3125	3.7	7.3	Steep gradient (~ 21%)
26	West Creek	GB	CO	RMNP	40° 27' 06"	105° 31' 01"	2496	5.0	17.0	Waterfall (h = 6.9 m)
27	Williams Gulch	GB	CO	Arapaho-Roosevelt NF	40° 42' 42"	105° 45' 56"	2758	4.1	7.3	Steep cascades (gradient ~ 43%)
28	West Fork San Francisco Creek	RG	CO	Rio Grande NF	37° 33' 22"	106° 23' 44"	2890	4.6	14.8	Rock gabion (h = 0.4 m)

^a These stream habitats included one or more lakes that support cutthroat trout.

Table 2.2. Habitat factors included in stream-scale models calculated from variables measured during basin-wide field surveys of translocation streams.

Habitat Variable	Definition
Stream length	Length (km) of stream surveyed from the fish movement barrier upstream to the end of pool habitat (< 2.0 m bankfull width), measured from a 1:24,000 USGS topographic quadrangle.
Number of pools	Total number of pools at least ½ channel width long with residual depths ≥ 18 cm. Pools were identified according to Hawkins et al. (1993) as channel geomorphic units formed by interactions among discharge, sediment load, and channel resistance to flow.
Bankfull pool width	Grand mean bankfull width (m) of all pools, calculated from measures at the downstream, center, and upstream ends of each pool, at the height where the water surface is level with the floodplain (Dunne and Leopold 1978).
Deep pools	Number of pools with residual depth ≥ 30 cm, calculated from the maximum depth minus the maximum tail crest depth measured at the downstream hydraulic control that forms the pool (Lisle 1987). This depth criterion was based on the median residual depth of all pools surveyed in all streams.

Table 2.2. Concluded.

Large woody debris	Number of pools with at least one piece of large woody debris, which was at least 15 cm in diameter for 3 m of length (adapted from Richmond and Fausch 1995) and at least partially in or suspended over the bankfull channel (including pieces forming pools).
Boulders	Number of pools with at least one boulder, which was >50 cm in diameter in all dimensions and within the bankfull channel, including those forming the stream bank if they protruded into the pool.
Undercut bank	Number of pools with at least 0.2 m of undercut bank, which was at least 10 cm undercut, no more than 15 cm above the water surface, and had a minimum water depth of 10 cm (adapted from Fausch and Northcote 1992).
Clean gravel	Number of pools with at least 25% area as clean gravel (6-63 mm in diameter, free from silt) in the downstream quarter of the pool, estimated visually.

Table 2.3. Basin-scale habitat factors measured for each translocation stream using a Geographic Information System and digital data derived from USGS digital elevation models. Specific commands from the Arc/Info[®] hydrological modeling tools (ESRI 1995) are listed in capitals letters.

Habitat Variable	Definition
Watershed area	Total upslope area (km ²) contributing flow to the basin outlet, calculated as planimetric watershed area (ZONALGEOMETRY).
Main channel length	Surface length (km) measured along the main channel (excluding tributaries) from the basin outlet to the headwaters. The GIS defined channels in cells where flow accumulation exceeded 1000 upslope cells, and incorporated changes in elevation in its calculation (SURFACELENGTH).
Total channel length	Surface length (km) computed by summing the length of all stream segments within the drainage basin, including tributary streams, which may also provide trout habitat. Channels were defined as for main channel length (SURFACELENGTH).
Drainage density	The total length of stream channels per unit basin area (km/km ²), calculated as total channel length divided by watershed area.
Latitude	The latitude of the basin outlet in UTM coordinates (m; CELLVALUE).
Elevation	Minimum elevation above sea level (m), measured at the basin outlet (ZONALSTATS).

Table 2.3. Concluded.

Basin relief	The difference between the highest and lowest elevations occurring within a basin (m; ZONALSTATS).
Basin slope	The percentage change in elevation quantified for each basin as mean percent rise (ZONALSTATS).
Basin aspect	The mean direction of the drainage basin measured as compass degrees either clockwise or counterclockwise from North. Data were restricted to values between 0° and 90°, which correspond to a North-South versus East-West aspect, respectively (ZONALSTATS).
Stream slope	The percentage change in elevation quantified for each stream network as mean percent rise (ZONALSTATS).
Stream aspect	The mean direction of the stream network measured as compass degrees either clockwise or counterclockwise from North. Data were restricted to values between 0° and 90°, which correspond to a North-South versus East-West aspect, respectively (ZONALSTATS).

Table 2.4. Values for habitat variables for the 28 study streams where cutthroat trout were translocated. Variables shown are used in the “best” models to predict probability of cutthroat trout translocation success. Total number of pools and number of pools with structure were highly correlated with number of deep pools ($r \geq 0.95$ for both), so are not shown here (see text and Appendix III).

Stream	Number of age-1 and older trout basin-wide	Mean July daily temperature (°C)	Mean bankfull pool width (m)	Number of deep pools (RD $\geq$ 30 cm)	Watershed area (km ²)	Basin aspect (degrees from N)
<i>Absent</i>						
Benito Creek	0	4.8	1.8	12	5.1	19
Doctor Creek	0	10.2	3.2	22	7.1	58
Hourglass Creek	0	4.2	3.8	94	9.5	64
Little Medano Cr.	3	7.1	2.1	35	16.2	10
Unknown Creek	1	6.5	1.0	2	8.5	32
W.F. San Francisco Cr.	0	9.5	2.8	13	7.2	25
Mean (SE)	-	7.1 (1.0)	2.5 (0.4)	30 (14)	8.9 (1.6)	35 (9)
<i>Low Population</i>						
Cottonwood Creek	67	7.1	2.6	131	9.2	41
Little Ute Creek	16	7.8	4.5	22	8.9	48
May Creek	65	8.4	3.3	144	12.5	59
Ouzel Creek	36	6.0	4.1	12	7.0	15
Pecos River	26	9.2	2.9	61	11.3	18

Table 2.4. Continued.

Rio de los Pinos	49	8.7	3.2	36	9.5	26
Rough C./ Rhodes G.	40	8.3	2.7	29	10.3	18
West Creek	82	7.1	4.1	117	25.3	67
Mean (SE)	-	7.8 (0.4)	3.4 (0.3)	69 (19)	11.7 (2.0)	36 (7)
<i>High Population</i>						
Cony Creek	129 ^{a, b}	9.0	5.4	146	14.5	17
East Middle Creek	117 ^b	9.7	3.2	46	14.2	43
Fern Creek	61 ^b	7.7	4.5	56	6.8	17
Greenhorn Creek	173	8.4	2.9	121	7.3	17
Jacks Creek	322 ^c	9.8	3.6	197	18.4	4
Medano Creek	1278	10.5	4.1	361	77.8	8
Nabor Creek	719 ^b	14.5	3.8	74	11.9	23
N.F. Big Thompson R.	112 ^b	8.1	3.1	18	8.3	47
Rio Cebolla	778 ^c	14.6	3.0	69	37.1	20
Roaring River	124 ^a	9.9	4.9	21	14.6	14
San Francisco Creek	193 ^a	8.8	3.9	88	42.5	14
Sheep Creek	661	7.8	4.1	318	36.1	24
Williams Gulch	319	10.8	2.3	105	9.0	32

Table 2.4. Concluded.

Mean (SE)	-	10.0 (0.6)	3.8 (0.2)	125 (30)	22.9 (5.7)	22 (4)
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Test Translocation Site

Powderhouse Creek	—	10.0	2.2	5	10.0	27
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^a Minimum abundance of age-1 and older trout was probably underestimated due to deep, turbulent, or turbid water.

^b Minimum abundance of age-1 and older trout was probably underestimated due to a lake or large beaver pond where it was not possible to make accurate visual counts.

^c Minimum abundance of age-1 and older trout was probably underestimated due to thick riparian or aquatic vegetation.

Table 2.5. Statistical models predicting cutthroat trout translocation success from summer water temperatures measured using thermographs, water temperature and stream-scale habitat collected during field surveys, basin-scale habitat measured from digital data, and a combination of stream- and basin-scale factors from the “best” models. Akaike’s Information Criterion corrected for small sample size (AICc) and AIC weights (*w*) were used to select the “best approximating” models from each set of *a priori* candidate models. Each set of models is nested but arranged according to their AICc ranking.

Model	AICc	<i>w</i>
<i>Summer Water Temperature</i>		
July through August	51.420	0.289
July	51.656	0.256
August	52.416	0.175
June through August	55.365	0.040
Intercept only	61.014	0.002
June	61.476	0.002
<i>Water Temperature and Stream-Scale Habitat</i>		
Summer temperature, pool width, number of deep pools	45.518	0.286
Summer temperature, pool width	46.678	0.162
Summer temperature, pool width, number of all pools	46.997	0.138
Summer temperature, pool width, number of pools with structure	47.298	0.118
Summer temperature, pool width, stream length	48.231	0.074
Summer temperature, pool width, number of pools with clean gravel	48.354	0.070

Table 2.5. Continued.

Summer temperature, number of deep pools	48.726	0.058
Summer temperature, number of all pools	50.904	0.020
Pool width, number of all pools	51.376	0.015
Summer temperature	51.656	0.013
Pool width, stream length	51.791	0.013
Summer temperature, stream length	52.006	0.011
Pool width, number of deep pools	53.412	0.006
Pool width, number of pools with clean gravel	54.171	0.004
Pool width, number of pools with structure	54.207	0.004
Number of all pools	55.146	0.002
Stream length	55.636	0.002
Pool width	55.656	0.002
Intercepts only	61.014	0.000
Winter temperature	63.274	0.000

Basin-Scale Habitat

Watershed area, basin aspect	56.173	0.201
Watershed area	56.437	0.177
Watershed area, basin relief	57.029	0.131

Table 2.5. Continued.

Watershed area, latitude	58.046	0.079
Watershed area, basin relief, latitude	58.806	0.054
Watershed area, elevation	58.894	0.052
Main channel length	58.907	0.051
Watershed area, main channel length	59.086	0.047
Watershed area, basin slope	59.124	0.046
Watershed area, basin aspect, elevation	59.165	0.045
Watershed area, elevation, latitude	60.805	0.020
Main channel length, elevation	60.959	0.018
Intercepts only	61.014	0.018
Main channel length, stream slope	61.531	0.014
Main channel length, stream aspect	61.655	0.013
Watershed area, elevation, latitude, elevation*latitude	62.121	0.010
Main channel length, elevation, latitude	62.392	0.009
Drainage density	63.143	0.006
Main channel length, stream aspect, elevation	63.951	0.004
Main channel length, elevation, latitude, elevation*latitude	64.102	0.004

Table 2.5. Concluded.

<i>Stream- and Basin-Scale Habitat</i>		
Summer temperature, pool width, deep pools	45.518	0.366
Summer temperature, pool width	46.678	0.205
Summer temperature, width, deep pools, watershed area, basin aspect	47.005	0.174
Summer temperature, pool width, watershed area	48.420	0.086
Summer temperature, width, deep pools, watershed area	48.847	0.069
Summer temperature, watershed area	50.816	0.026
Summer temperature, deep pools, watershed area	51.240	0.021
Pool width, deep pools, watershed area, basin aspect	51.557	0.018
Summer temperature	51.656	0.017
Pool width, watershed area, basin aspect	53.298	0.007
Pool width, watershed area	54.145	0.005
Pool width, deep pools, watershed area	54.992	0.003
Watershed area	56.437	0.002
Watershed area, basin aspect	56.173	0.002
Intercepts only	61.014	0.000

Table 2.6. Maximum likelihood estimates from ordinal polytomous logistic regression of intercept and slope parameters for the four “best approximating” models predicting cutthroat trout translocation success from water temperature and stream-scale habitat collected during field surveys. Standard errors are in parentheses. All models are significant at $P = 0.0001$.

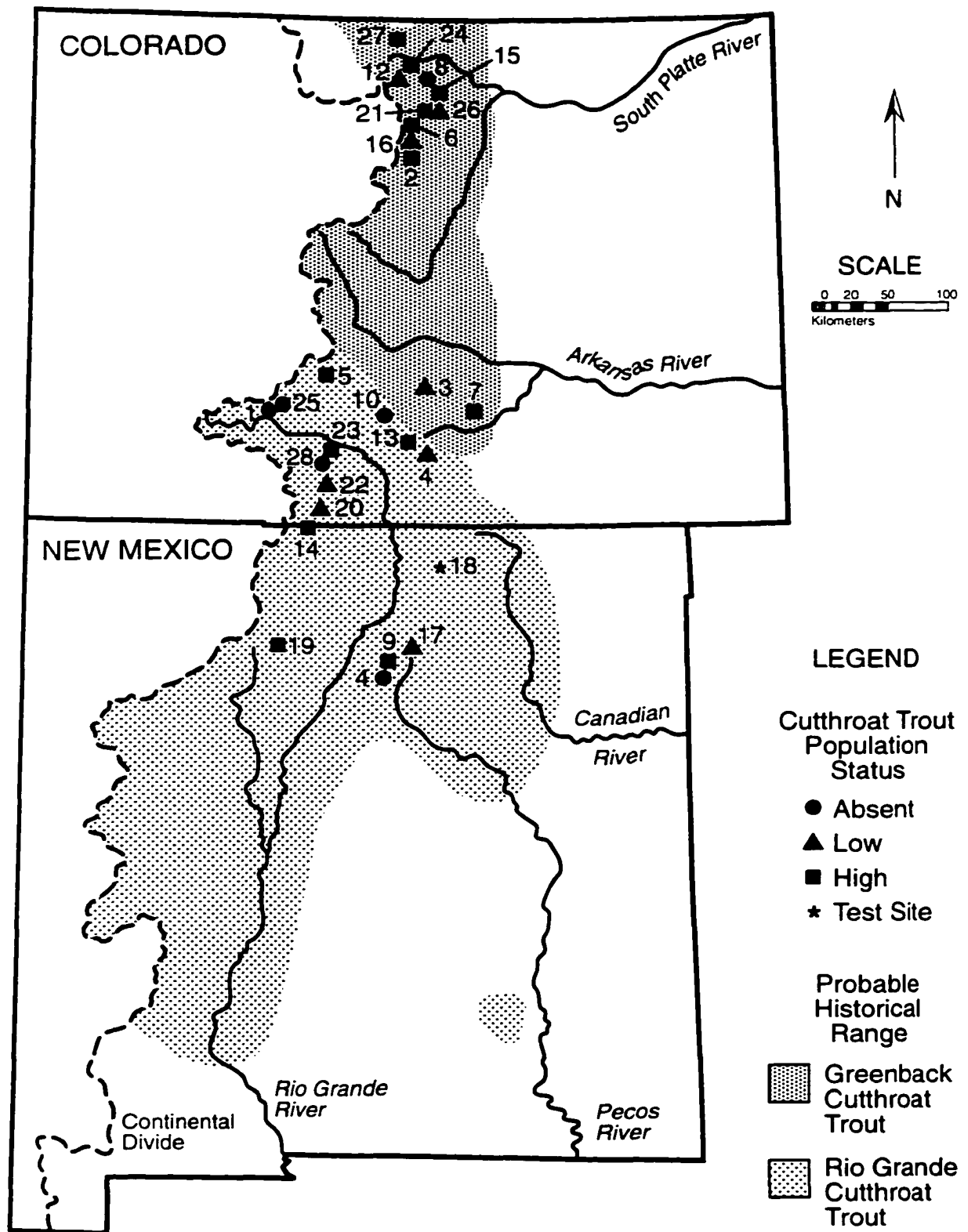
Model			Mean July		Number of pools ^a
	Intercept 1	Intercept 2	temperature	Pool width	
Summer temperature, pool width, deep pools ^a	11.454 (4.460)	14.077 (4.868)	-0.891 (0.355)	-1.451 (0.713)	-0.017 (0.010)
Summer temperature, pool width	10.345 (3.853)	12.688 (4.179)	-0.889 (0.336)	-1.460 (0.646)	- -
Summer temperature, pool width, all pools ^a	10.369 (4.052)	12.846 (4.398)	-0.765 (0.334)	-1.433 (0.676)	-0.008 (0.006)
Summer temperature, pool width, pools with structure ^a	10.277 (3.970)	12.749 (4.327)	-0.816 (0.330)	-1.352 (0.668)	-0.012 (0.009)

^a Three models included number of pools that met specific criteria (see text and Appendix III).

Table 2.7. Maximum likelihood estimates from ordinal polytomous logistic regression of intercept and slope parameters for the four “best approximating” models predicting cutthroat trout translocation success from basin-scale habitat measured from digital data (see Appendix IV). Standard errors are in parentheses. All models are significant at $P \leq 0.016$.

Model	Intercept 1	Intercept 2	Watershed	Basin	Basin	Latitude
			area	aspect	relief	
Watershed area, basin aspect	-0.867 (1.164)	0.829 (1.145)	-0.144 (0.081)	0.039 (0.024)	- -	- -
Watershed area	0.251 (0.869)	1.804 (0.916)	-0.123 (0.069)	- -	- -	- -
Watershed area, basin relief	-1.177 (1.366)	0.478 (1.337)	-0.176 (0.096)	- -	0.002 (0.002)	- -
Watershed area, latitude	9.901 (9.184)	11.529 (9.265)	-0.125 (0.068)	- -	- -	-0.000002 (0.000002)

Figure 2.1. The approximate historical range of greenback and Rio Grande cutthroat trout (adapted from Behnke 1992; Stumpff and Cooper 1996) and location of the 28 study streams where cutthroat trout were translocated. Stream numbers correspond to those in Table 2.1.



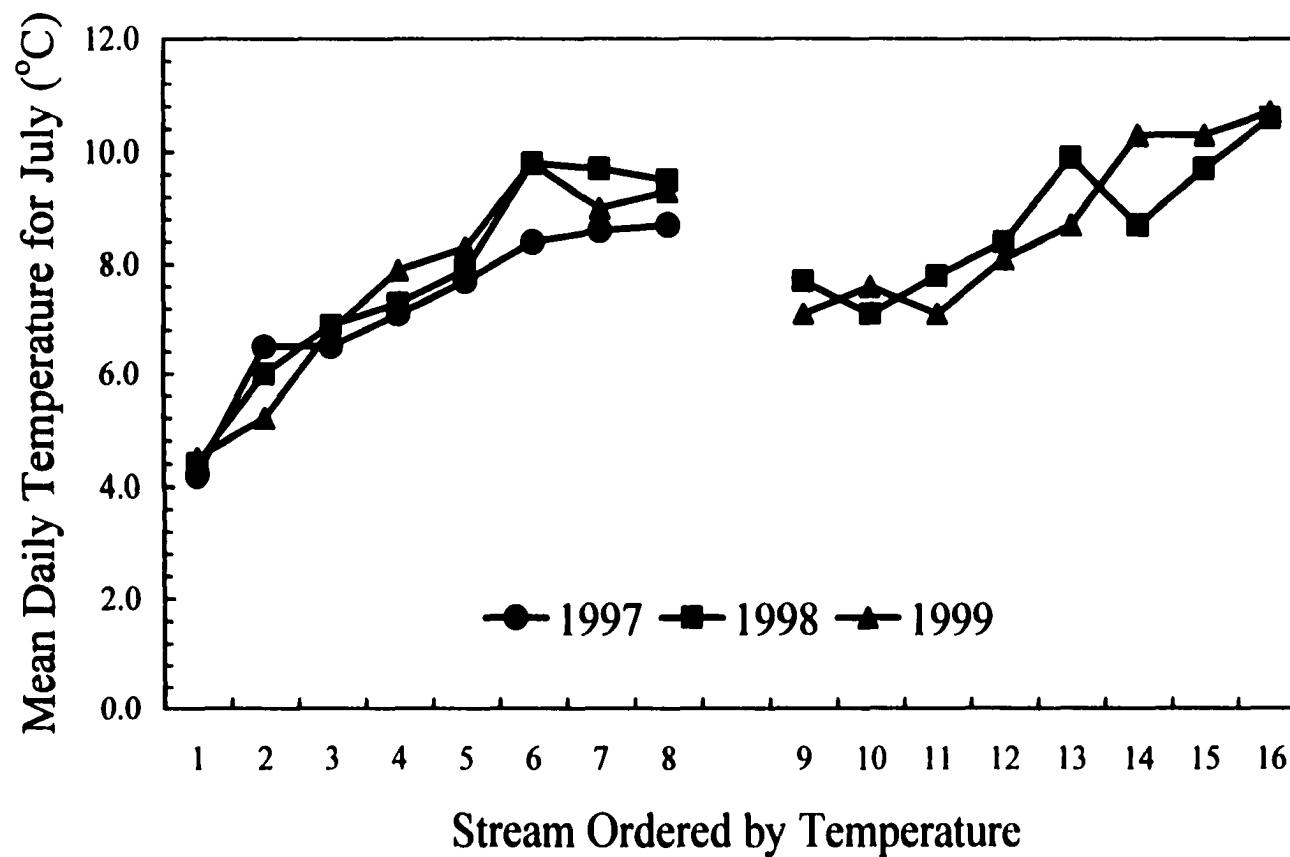


Figure 2.2. Mean daily water temperature (degrees C) for July for the 16 study streams with two or more years of data, ordered by temperature. There was no significant difference in July temperatures among years ($P = 0.15$ by ANOVA).

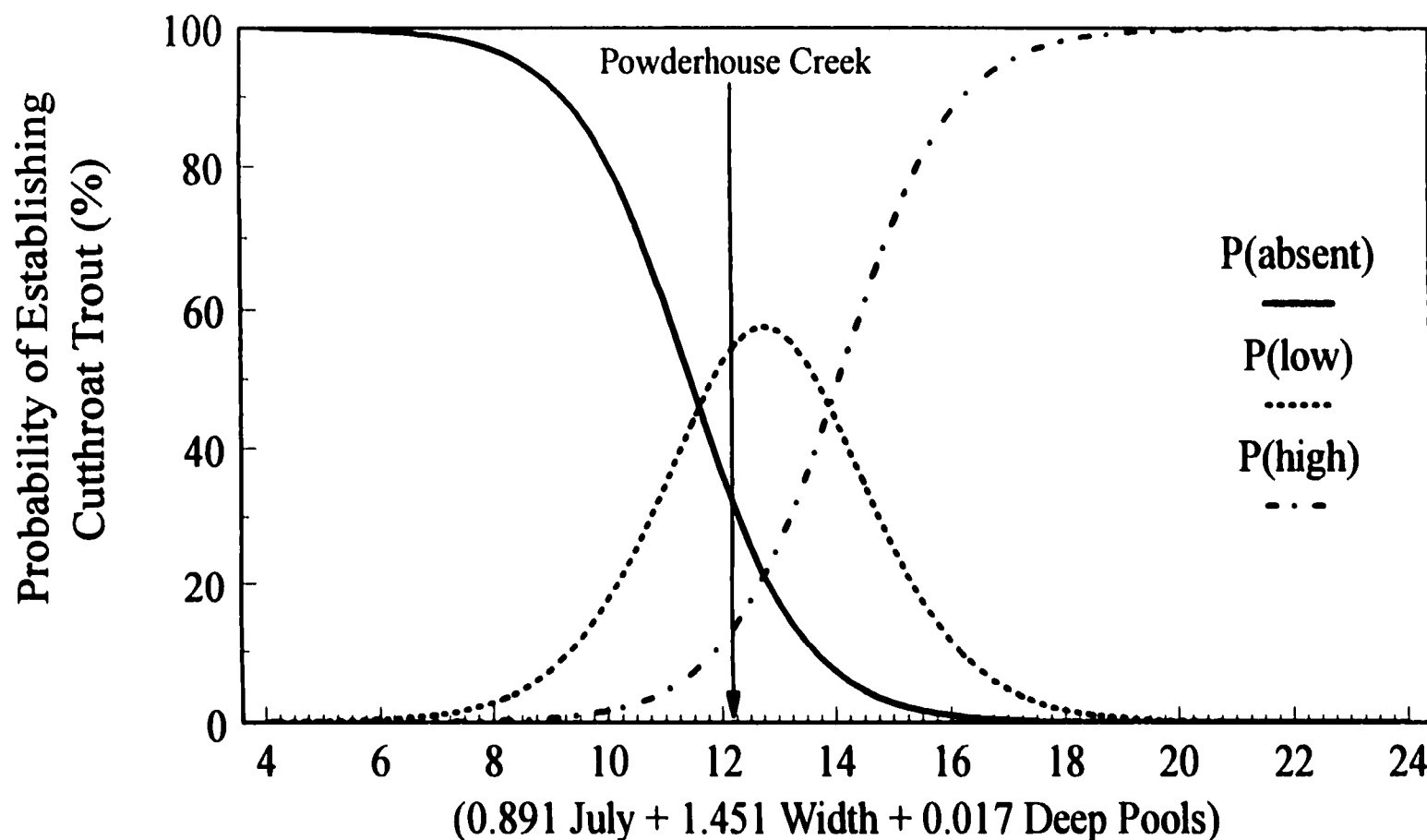


Figure 2.3. The "best" model for predicting success of translocated cutthroat trout populations from stream-scale habitat attributes. Curves show the predicted probability of translocation success based on a polytomous logistic regression function (shown on the abscissa) including mean daily water temperature for July (degrees C), mean pool bankfull width (m), and total number of deep pools (residual depth >30 cm). This model can be used to estimate the probability that a future translocation site will be successful. For example, predicted probabilities for Powderhouse Creek were only 13% for supporting high numbers of cutthroat trout, 54% for supporting low numbers, and 33% for supporting no cutthroat trout (shown by the arrow).

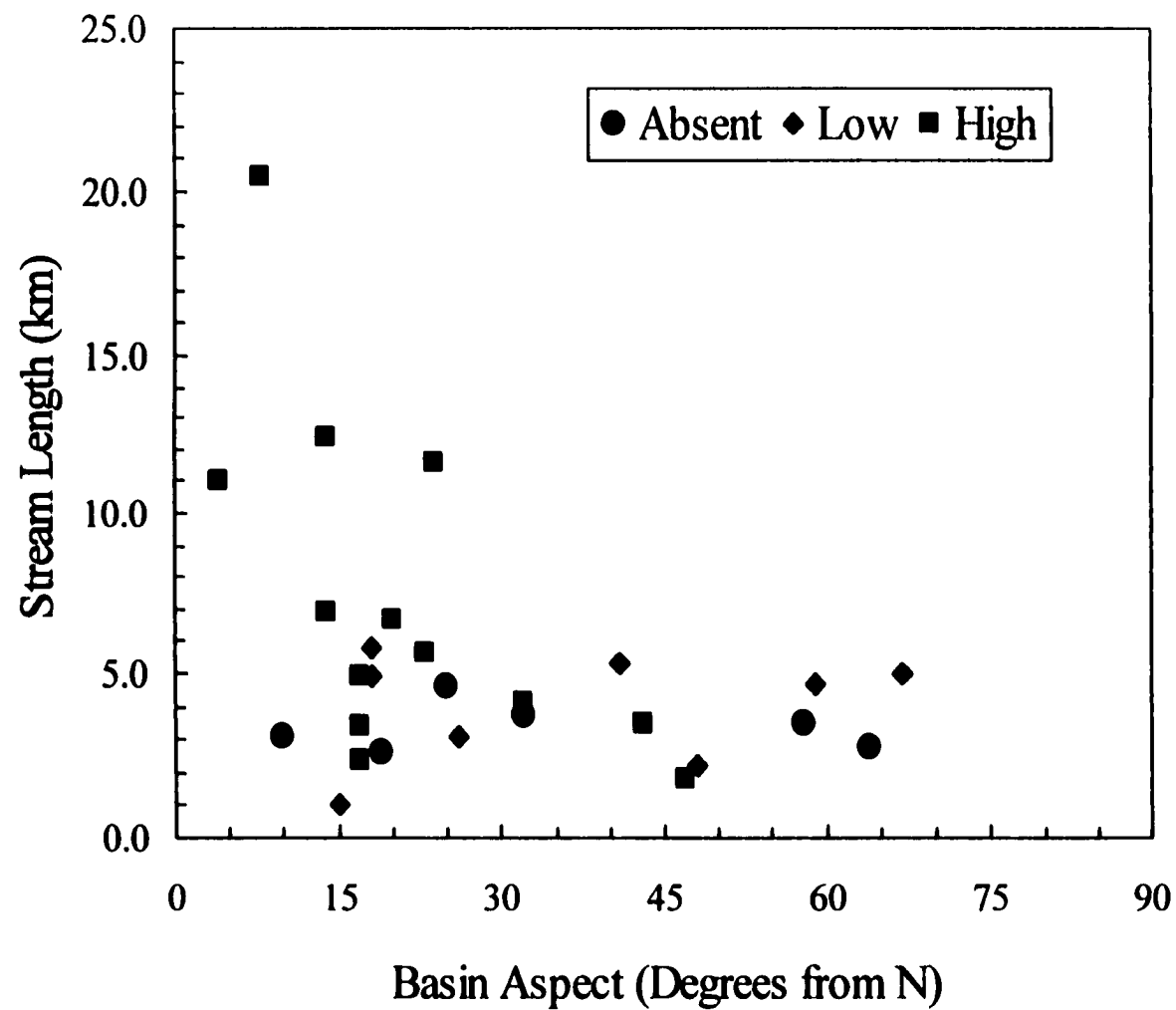


Figure 2.4. Basin aspect versus stream length (km), measured as the distance from the downstream fish movement barrier to the upstream end of pool habitat (bankfull width < 2.0 m). Symbols show population status of the translocation streams. Four large basins with high numbers of cutthroat trout that trend North-South likely caused spurious statistical significance for basin aspect in the logistic regression model.

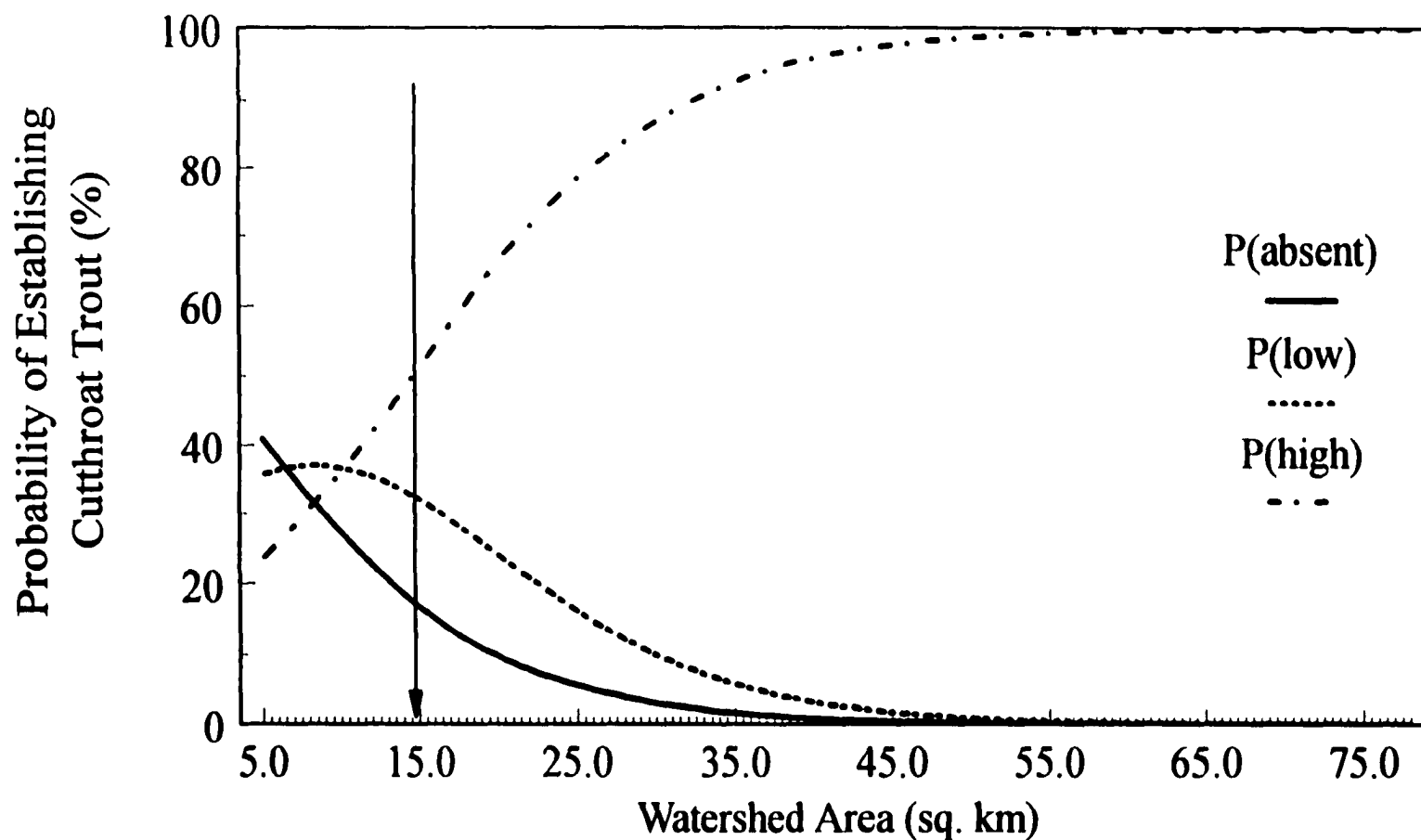


Figure 2.5. The "best" model for predicting success of translocated cutthroat trout populations from basin-scale habitat attributes. Curves show the predicted probability of translocation success based on a polytomous logistic regression function of watershed area (square kilometers). Translocations have greater than a 50% chance of establishing high numbers of cutthroat trout in watersheds greater than 14.7 square kilometers (shown by the arrow) .

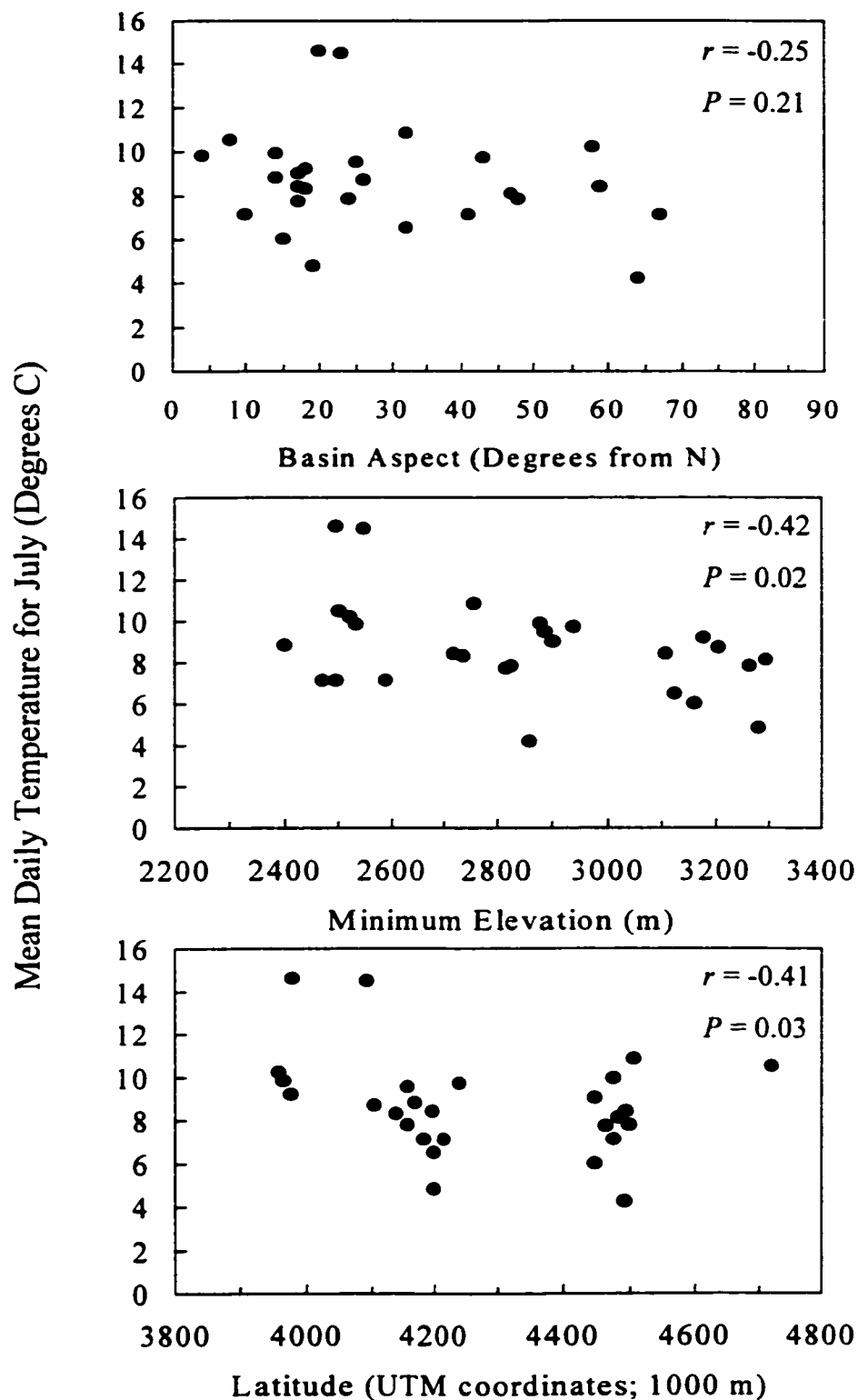


Figure 2.6. The relationship between aspect (degrees from N), elevation (m), and latitude (Universal Transmercator coordinates in 1000 m) and July water temperatures (degrees C) measured with thermographs for 27 cutthroat trout translocation streams.

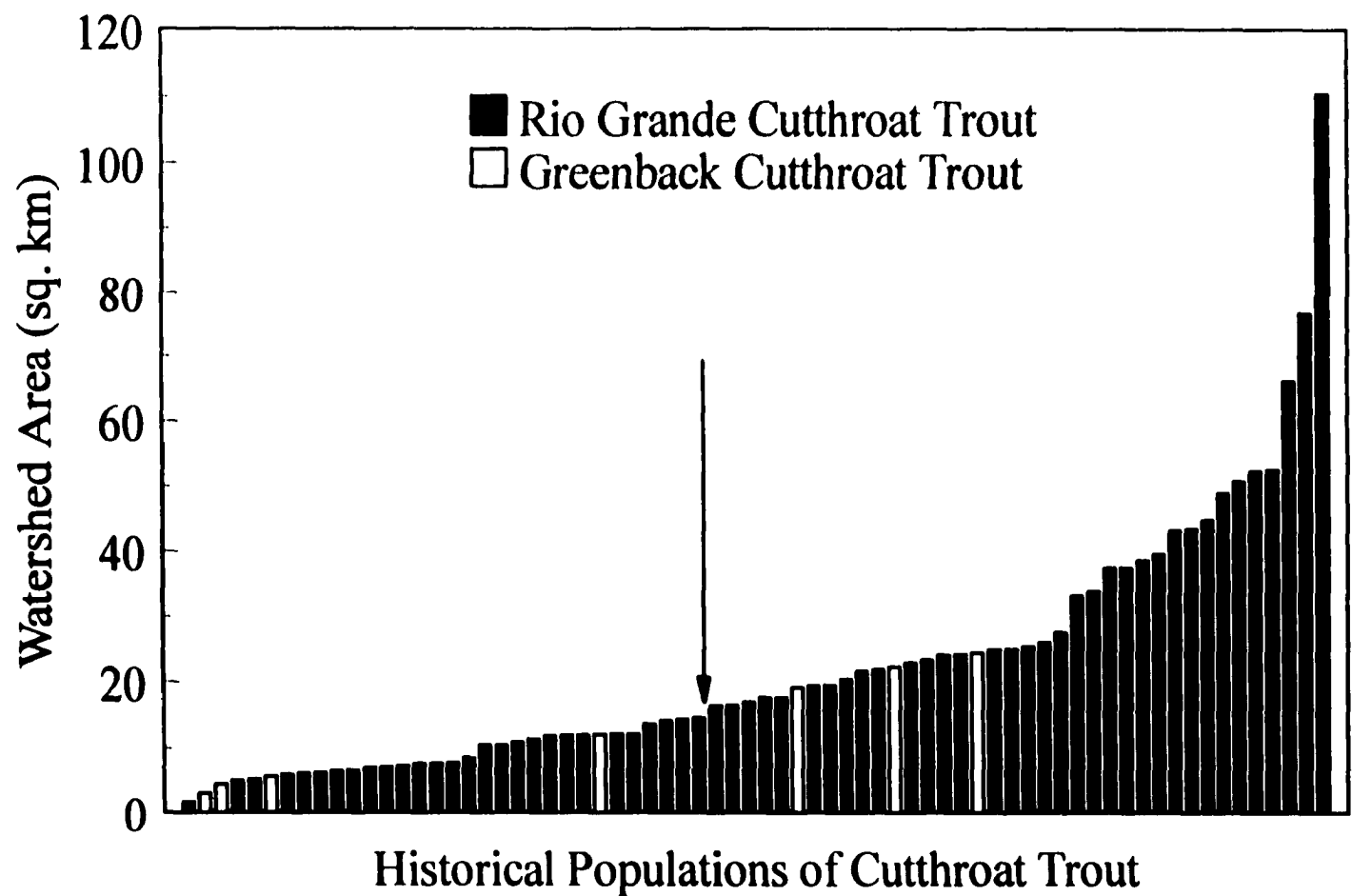


Figure 2.7. The distribution of watershed area (square kilometers) for 70 historical stream populations of greenback and Rio Grande cutthroat trout. Thirty-two populations in watersheds < 14.7 square kilometers (shown by the arrow) may be at greater risk from extirpation due to insufficient habitat (see text).

APPENDIX I:
GREENBACK CUTTHROAT TROUT TRANSLOCATION PROJECTS

Appendix I. Greenback cutthroat trout translocation projects carried out prior to 1995 (USFWS 1998 and personal communications) in the South Platte (SP) and Arkansas (AR) drainage basins. Locations include Arapaho-Roosevelt National Forest (ARNF), Fort Carson Military Base, Leadville National Fish Hatchery (LNFH), Rocky Mountain National Park (RMNP), Pike-San Isabel National Forest (PSINF), and city water sources. Chemical treatment is the date of treatment with rotenone or antimycin to remove nonnative salmonids prior to introduction of cutthroat trout. No chemical treatment indicates that the habitat was barren of fish, and netted indicates that nonnative salmonids were removed by gillnet prior to introduction. Initial stocking includes the source of fish (W = wild, H = hatchery) and the year of first introduction.

No.	Greenback cutthroat trout translocation water	Drain-age	Location	Stream (km)	Lake (ha)	Median elevation (m)	Potential barrier to fish movement	Chemical treatment	Initial stocking
<i>Successful Translocations</i>									
1	North Fork Big Thompson, Lake Louise, and ponds	SP	RMNP	1.7	13.1	3320	Waterfall	None	W: 1971
2	Bear Lake	SP	RMNP	0.1	4.5	2890	Steep cascades	1975	W: 1975
3	Williams Gulch	SP	ARNF	2.0	0.0	2900	Steep cascades	None	H: 1981
4	Sheep Creek	SP	ARNF	11.2	0.0	2970	Steep cascades	1981	H: 1982
5	Fern Lake and Fern Creek	SP	RMNP	1.4	3.7	2850	Fern Falls	1982	H: 1982
6	Odessa Lake	SP	RMNP	0.2	4.5	3030	Steep cascades	None	H: 1984
7	Lawn Lake	SP	RMNP	0.9	9.5	3350	Steep cascades	1983	H: 1984
8	Roaring River	SP	RMNP	6.5	0.0	3080	Horseshoe Falls	1983	H: 1984

Appendix I. Continued.

9	Lower Hutcheson Lake	SP	RMNP	1.0	1.7	3260	Steep cascades	1987	H: 1989
10	Pear Lake	SP	RMNP	1.2	6.1	3200	Steep cascades	1988	H: 1989
11	Sandbeach Lake	SP	RMNP	0.1	4.0	3140	Steep cascades	1988	H: 1989
12	Spruce Lake	SP	RMNP	0.3	1.5	2940	Steep cascades	None	H: 1991
13	Cony Creek	SP	RMNP	3.5	0.0	3040	Calypso Cascades	1988	H: 1989
14	Boehmer Reservoir	AR	Colorado Springs	0.0	10.1	3500	Reservoir spillway Height = 18 m	1984	H: 1985

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Unsuccessful Translocations for Reasons Apparently Related to Habitat Size or Quality

15	May Creek	SP	ARNF	1.7	0.0	3040	Steep cascades	None	W: 1980
16	Loomis Lake	SP	RMNP	0.4	1.5	3120	Steep cascades	None	H: 1991
17	West Creek	SP	RMNP	2.4	0.0	2960	West Falls	1978	W: 1979
18	Ouzel Creek (below Bluebird Lake)	SP	RMNP	1.4	0.0	3180	Steep cascades	1980	H: 1981
19	Big Crystal Lake	SP	RMNP	0.0	10.0	3510	Waterfall	None	H: 1984
20	Hourglass Creek	SP	ARNF	2.0	0.0	2990	Gage station and fish weir	None	W: 1965

Appendix I. Continued.

21	Lily Lake	SP	RMNP	0.0	5.0	2610	Cement dam	Netted	H: 1986
22	North Fork Jackson Creek	SP	PSINF	0.3	0.0	2950	Intermittent channel	1984	H: 1987
23	Cottonwood Creek	AR	PSINF	6.4	0.0	3150	Dry channel	None	H: 1983
24	Greenhorn Creek	AR	PSINF	3.2	0.0	3270	Waterfalls	None	H: 1988

Unsuccessful Translocations Due to Reinvasion by Nonnative Salmonids

25	George Creek	SP	ARNF	12.7	0.0	2570	Rock gabion Height = 1 - 2 m	1982	H: 1983
26	Black Hollow Creek	SP	ARNF	5.2	0.0	2540	Wooden structure	1967, 1979	W: 1967
27	Bruno Gulch	SP	PSINF	9.0	1.4	3060	Wooden structure Height = 2.4 m	1985	H: 1987
28	Pennock Creek	SP	ARNF	8.0	0.0	2760	Rock gabion Height = 1.4 m	1985	H: 1986
29	Cornelius Creek	SP	ARNF	6.9	0.0	2500	Rock gabion Height = 1 - 2 m	1982	H: 1983
30	Ouzel Lake and Ouzel Creek (above Ouzel Falls)	SP	RMNP	4.7	2.6	3000	Ouzel Falls	1980	H: 1981

Appendix I. Concluded.

31	Hidden Valley Creek and beaver ponds	SP	RMNP	1.6	2.5	2760	Steep cascades	1973	H: 1973
32	Lake Fork complex	AR	PSINF	4.4	11.3	3290	Wooden structure Height = 2 - 3 m	1986	H: 1987
33	Rock Creek complex	AR	LNHF, PSINF	10.3	8.3	3290	Water diversion	1990	H: 1991
34	Husted Lake	SP	RMNP	0.1	4.1	3380	Steep cascades	1986, 1990	H: 1991
35	North Fork Big Thompson River and Lost Lake	SP	RMNP	3.0	3.7	3140	Lost Falls	1986	H: 1987

Unsuccessful Translocations Due to Metal Pollution (36) and Bird Predation (37)

36	Bard Creek	SP	ARNF	6.1	0.0	2960	Steep cascades	None	H: 1982
37	Lytle and Duck Pond	AR	Fort Carson	1.0	2.1	1890	No outlet stream	None	W: 1981

APPENDIX II:
AGENCY STOCKING RECORDS FOR GREENBACK AND RIO GRANDE
CUTTHROAT TROUT TRANSLOCATION STREAMS

Appendix II. Unpublished data from natural resource agencies on the stocking frequency, source and number of fish, and approximate size of individuals translocated to the 28 greenback (GB) or Rio Grande (RG) cutthroat trout study streams (Harig and Fausch 1996; Y. Paroz, New Mexico Department of Game and Fish, personal communication; unpublished reports from the Colorado Division of Wildlife and U.S. Fish and Wildlife Service). Dashes indicate that data were unavailable.

Stream	Sub-species	Number of stocking events	Source of fish	Approximate number of fish stocked	Approximate size of individuals (cm)
<i>Absent</i>					
Benito Creek	RG	2	Wild	275	8 – 15
Doctor Creek	RG	Natural ^a	Wild	--	--
Hourglass Creek	GB	4	Wild and hatchery	1686	3 – 15
Little Medano Creek	RG	2	Wild	400	10 – 15
Unknown Creek	RG	1	Wild	130	8
W.F. San Francisco Creek	RG	1	Wild	65	13 – 15
<i>Low Population</i>					
Cottonwood Creek	GB	7	--	--	--
Little Ute Creek	RG	3	Wild	522	2 – 15
May Creek	GB	5	Wild and hatchery	6255	2 – 15
Ouzel Creek	GB	3	Hatchery	--	2 – 8
Pecos River	RG	2	Wild	63	--
Rio de los Pinos	RG	1	Wild	300	5 – 25

Appendix II. Concluded.

Rough Canyon/ Rhodes Gulch	RG	1	Wild	230	8
West Creek	GB	4	Wild and hatchery	4158	2 – 15

High Population

Cony Creek	GB	1	Hatchery	5000	2 – 5
East Middle Creek	RG	1	Wild	515	10 – 15
Fern Creek	GB	3	Hatchery	>36,000	2 – 5
Greenhorn Creek	GB	1	--	--	--
Jacks Creek	RG	2	Wild	219	--
Medano Creek	RG	3	Wild and broodstock	3152	13
Nabor Creek	RG	1	Wild	736	8 – 15
N.F. Big Thompson River	GB	2	Wild and hatchery	4000	3 – 13
Rio Cebolla	RG	2	Wild	865	--
Roaring River	GB	4	Hatchery	>20,000	2 – 5
San Francisco Creek	RG	2	Wild	182	8 – 15
Sheep Creek	GB	5	Hatchery	>15,000	3 – 10
Williams Gulch	GB	4	Hatchery	4109	2 – 15

Test Translocation Site

Powderhouse Creek	RG	1	Wild	340	--
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- ^a Prior to treatment with rotenone, Rio Grande cutthroat trout in Doctor Creek were moved upstream above an upper barrier to fish movement (number of fish is unknown). It was assumed that these fish would move downstream and repopulate the stream after treatment (Y. Paroz, New Mexico Department of Game and Fish, personal communication). The status of the cutthroat trout above the upper barrier is unknown, but I did not see any in a ~ 100 meter reach just upstream of the barrier during my habitat survey.

APPENDIX III:
VALUES FOR REMAINING STREAM-SCALE HABITAT VARIABLES

Appendix III. Values for additional stream-scale habitat variables quantified from field surveys of the 28 study streams where Rio Grande or greenback cutthroat trout were translocated.

Stream	Number of pools	Number of pools with structure	Number of pools with clean gravel	Mean daily temperature for Dec - Feb (°C)
<i>Absent</i>				
Benito Creek	51	8	0	1.4
Doctor Creek	71	40	34	0.2
Hourglass Creek	154	100	72	0.3
Little Medano Cr.	47	36	10	0.4
Unknown Creek	45	10	6	0.8
W.F. San Francisco Cr.	67	33	1	0.5
Mean (SE)	73 (17)	38 (14)	21 (11)	0.6 (0.2)
<i>Low Population</i>				
Cottonwood Creek	192	86	72	1.7
Little Ute Creek	39	16	20	0.2
May Creek	325	174	207	0.2
Ouzel Creek	24	15	7	0.6
Pecos River	148	56	65	0.2
Rio de los Pinos	118	71	84	0.7

Appendix III. Continued.

Rough C./ Rhodes G.	141	71	64	0.2
West Creek	138	114	19	0.3
Mean (SE)	141 (33)	75 (18)	67 (22)	0.5 (0.2)
<i>High Population</i>				
Cony Creek	227	153	116	1.1
East Middle Creek	162	78	155	0.2
Fern Creek	84	58	47	0.2
Greenhorn Creek	151	74	27	0.3
Jacks Creek	346	217	155	0.5
Medano Creek	571	313	286	0.3
Nabor Creek	215	108	114	0.4
N.F. Big Thompson R.	37	21	0	0.3
Rio Cebolla	214	44	33	2.6
Roaring River	81	35	51	0.0
San Francisco Creek	217	88	54	0.2
Sheep Creek	490	302	218	0.0
Williams Gulch	173	118	85	0.0
Mean (SE)	228 (43)	124 (27)	103 (23)	0.5 (0.2)

Appendix III. Concluded.

<i>Test Translocation Site</i>				
Powderhouse Creek	76	42	73	0.0

APPENDIX IV:
VALUES FOR REMAINING BASIN-SCALE HABITAT VARIABLES

Appendix IV. Values for additional basin-scale habitat variables measured with a Geographic Information System from digital data for the 28 study streams where Rio Grande or greenback cutthroat trout were translocated.

Stream	Main channel length (km)	Total stream length (km)	Drainage density (km/km ²)	Basin relief (m)	Mean basin slope (%)	Stream aspect (degrees from N)
<i>Absent</i>						
Benito Creek	2.8	5.4	1.1	864	35.9	66
Doctor Creek	5.1	5.2	0.7	1175	42.3	81
Hourglass Creek	4.5	5.0	0.5	1006	28.0	88
Little Medano Cr.	7.2	14.4	0.9	1479	47.0	14
Unknown Creek	4.5	4.5	0.5	627	25.4	67
W.F. San Francisco Cr.	7.0	7.0	1.0	985	34.2	14
Mean (SE)	5.2 (0.7)	6.9 (1.5)	0.8 (0.1)	1023 (118)	35.5 (3.4)	55 (13)
<i>Low Population</i>						
Cottonwood Creek	11.1	11.3	1.2	1620	43.1	67
Little Ute Creek	2.5	5.5	0.6	1094	50.0	50
May Creek	6.4	9.8	0.8	885	16.9	88
Ouzel Creek	3.0	3.4	0.5	849	52.9	52
Pecos River	5.6	6.9	0.6	667	23.4	1
Rio de los Pinos	3.4	4.7	0.5	504	20.0	4

Appendix VI. Continued.

Rough C./ Rhodes G.	4.6	9.2	0.9	922	22.1	80
West Creek	9.6	18.4	0.7	1584	31.6	77
Mean (SE)	5.8 (1.1)	8.7 (1.7)	0.7 (0.1)	1016 (142)	32.5 (5.0)	52 (12)
<i>High Population</i>						
Cony Creek	7.4	9.2	0.6	1103	37.6	75
East Middle Creek	5.2	9.3	0.7	1095	34.6	53
Fern Creek	4.3	4.7	0.7	942	50.1	84
Greenhorn Creek	2.7	4.7	0.7	571	19.0	67
Jacks Creek	11.2	11.2	0.6	1290	26.2	1
Medano Creek	20.2	67.8	0.9	1565	43.6	16
Nabor Creek	6.3	8.0	0.7	916	27.5	23
N.F. Big Thompson R.	2.0	5.2	0.6	854	39.0	89
Rio Cebolla	14.9	29.1	0.8	638	22.5	35
Roaring River	6.9	8.7	0.6	1279	45.3	22
San Francisco Creek	27.0	40.6	1.0	1629	27.6	30
Sheep Creek	15.5	25.9	0.7	764	17.4	39
Williams Gulch	5.1	6.7	0.7	339	12.1	60
Mean (SE)	9.9 (2.1)	17.8 (5.2)	0.7 (0.0)	999 (105)	31.0 (3.3)	46 (8)

Appendix VI. Concluded.

<i>Test Translocation Site</i>							
Powderhouse Creek	5.0	5.5	0.6	910	28.6	74	

APPENDIX V:
VALUES FOR BASIN-SCALE HABITAT VARIABLES FOR HISTORICAL
STREAM POPULATIONS OF GREENBACK AND RIO GRANDE
CUTTHROAT TROUT

Appendix V. Values for basin-scale habitat variables measured with a Geographic Information System from digital data for 70 historical stream populations of greenback and Rio Grande cutthroat trout.

Stream	Latitude	Longitude	Watershed area (km ²)	Main channel length (km)	Total stream length (km)	Drainage density (km/km ²)	Basin relief (m)	Min- imum elevation (m)	Mean basin slope (%)	Mean channel slope (%)	Basin aspect (degrees from N)	Stream aspect (degrees from N)
<i>Greenback Cutthroat Trout Populations</i>												
Cascade Cr	37° 38' 52"	105° 24' 07"	4.4	2.1	2.3	0.5	697	3068	33.2	16.8	48	75
Como Cr	40° 00' 59"	105° 30' 51"	5.6	2.9	2.9	0.5	469	2695	15.9	8.6	63	53
Hague Cr	40° 28' 57"	105° 40' 22"	3.0	2.3	2.8	1.0	756	3377	53.2	12.9	0	72
Hunters Cr	40° 12' 47"	105° 34' 21"	12.0	6.9	8.0	0.7	1463	2865	42.8	12.2	21	51
Poudre River	40° 32' 54"	105° 36' 29"	22.2	7.5	13.8	0.6	1322	2782	35.0	16.7	21	16
Roaring Cr	40° 43' 11"	105° 44' 27"	24.3	11.1	15.8	0.6	819	2532	22.9	10.7	2	7
South Apache Cr	37° 50' 26"	104° 55' 44"	19.0	11.4	17.4	0.9	1619	2084	49.2	18.1	35	55
<i>Rio Grande Cutthroat Trout Populations in Colorado</i>												
Bennett Cr	37° 50' 03"	107° 07' 55"	11.9	7.4	9.1	0.8	882	2914	20.2	11.6	31	50
Cascade Cr	37° 00' 52"	106° 21' 41"	6.4	3.9	4.6	0.7	348	2984	14.5	8.7	19	32
Cat Cr	37° 23' 06"	106° 14' 21"	76.6	20.1	65.6	0.9	1123	2517	27.4	13.2	20	28
Cuates Cr	37° 01' 32"	105° 22' 51"	11.9	8.1	9.5	0.8	1245	2644	36.9	15.1	31	83

Appendix V. Continued.

East Pass Cr	38° 09' 55"	106° 27' 21"	37.3	12.3	28.9	0.8	755	2597	21.5	7.5	39	45
Jaroso Cr	37° 02' 59"	105° 22' 28"	21.8	11.8	20.0	0.9	1270	2642	30.2	11.8	13	57
Johns Cr & Bear Cr	38° 00' 29"	106° 39' 24"	38.5	18.6	24.4	0.6	1089	2855	25.2	8.9	18	9
Middle Fork Carnero Cr	37° 54' 31"	106° 23' 49"	48.9	16.4	38.0	0.8	864	2639	22.7	8.0	42	51
North Fork Carnero Cr	37° 54' 21"	106° 23' 24"	65.9	13.7	49.7	0.8	788	2630	22.6	8.4	7	17
Osier Cr	37° 05' 19"	106° 18' 11"	33.1	9.1	19.6	0.6	669	2615	29.3	10.0	24	42
Torcido Cr	37° 04' 28"	105° 21' 16"	16.3	9.2	12.4	0.8	936	2653	30.2	11.7	20	57
Willow Cr	37° 01' 09"	105° 22' 52"	21.6	8.9	17.3	0.8	1275	2643	34.5	16.0	46	79
Wolf Cr	37° 01' 18"	106° 27' 57"	10.4	2.7	7.7	0.7	544	2960	21.4	14.5	47	40

Rio Grande Cutthroat Trout Populations in New Mexico

Agua Piedra Cr	36° 08' 01"	105° 31' 35"	16.9	6.9	14.8	0.9	1235	2575	29.1	14.5	52	85
Alamitos Cr	36° 03' 03"	105° 28' 20"	17.5	6.6	11.3	0.6	856	2962	20.6	9.2	51	89
Rito Angostura	36° 06' 24"	105° 29' 11"	25.3	12.1	22.2	0.9	1221	2698	26.7	12.1	49	73
Rito Azul	35° 56' 58"	105° 38' 24"	6.1	3.4	3.4	0.6	541	3109	23.6	14.4	76	71
Bitter Cr	36° 42' 22"	105° 24' 03"	27.4	13.9	18.9	0.7	770	2642	31.2	9.8	14	22
Cabresto Cr	36° 44' 05"	105° 30' 27"	52.5	18.5	34.4	0.7	1085	2562	38.5	13.8	12	46

Appendix V. Continued.

Rito Café	36° 01' 20"	106° 42' 26"	16.4	6.5	12.2	0.7	427	2657	16.2	7.6	26	38
Rio Capulin	35° 52' 04"	105° 47' 26"	10.8	7.7	8.2	0.8	1086	2750	37.1	17.3	42	65
Canjilon Cr	36° 32' 07"	106° 19' 08"	13.6	7.6	9.2	0.7	534	2800	16.4	8.7	23	8
Canones Cr	36° 06' 06"	106° 30' 04"	37.3	12.3	27.4	0.7	715	2513	22.7	9.5	8	27
Chihuahueros Cr	36° 07' 00"	106° 28' 00"	20.3	13.6	18.7	0.9	669	2523	26.4	9.6	6	26
Rito de los Chimayosos	35° 57' 23"	105° 36' 56"	7.0	1.5	3.2	0.5	710	3277	33.9	11.4	30	48
Rio Chiquito	36° 18' 47"	105° 25' 02"	24.0	9.7	14.2	0.6	739	2731	27.6	11.1	6	18
Columbine Cr	36° 40' 34"	105° 30' 54"	43.1	20.5	30.9	0.7	1456	2414	47.7	19.4	2	33
Comanche Cr	36° 49' 53"	105° 19' 03"	110.3	16.4	87.4	0.8	1093	2728	23.7	7.7	6	14
Dalton Canyon	35° 40' 39"	105° 45' 20"	12.1	5.9	9.2	0.8	752	2401	40.8	14.8	13	33
El Rito	36° 32' 52"	106° 16' 45"	24.8	8.3	20.8	0.8	424	2831	16.6	9.1	43	37
Frijoles Cr	36° 15' 52"	105° 24' 00"	6.9	3.2	4.1	0.6	630	3002	26.7	14.1	13	7
Gavilan Canyon	36° 35' 42"	105° 28' 42"	7.7	4.8	5.3	0.7	985	2738	50.7	24.1	6	1
Indian Cr	35° 44' 07"	105° 42' 41"	7.2	5.3	7.2	1.0	947	2523	34.1	12.4	44	47
Indian Cr	36° 10' 22"	105° 36' 47"	1.7	1.8	1.8	1.1	478	2362	21.0	13.6	76	48
La Cueva Canyon	36° 50' 24"	105° 18' 32"	8.5	4.2	7.0	0.8	763	2751	30.2	11.9	32	19
La Jara Cr	36° 07' 24"	106° 54' 56"	14.1	6.3	13.0	0.9	812	2399	28.9	16.8	39	52
Little Blue Cr	36° 14' 04"	105° 14' 10"	10.4	5.5	7.8	0.7	893	2482	33.0	11.9	67	64

Appendix V. Continued.

Little Costilla Cr	36° 47' 47"	105° 17' 46"	14.3	9.6	10.4	0.7	1034	2787	27.2	10.3	38	49
Lake Fork Cabresto Cr	36° 45' 01"	105° 29' 28"	19.4	6.6	12.0	0.6	1066	2798	41.7	18.2	3	17
Macho Canyon	35° 42' 21"	105° 43' 56"	22.9	9.2	15.6	0.7	920	2451	38.8	15.9	16	34
McCrystal Cr	36° 47' 31"	105° 07' 56"	24.1	14.8	19.6	0.8	1296	2539	22.3	8.6	48	62
Rio Nambe	3° 50' 36"	105° 50' 43"	33.8	8.6	23.1	0.7	1478	2347	39.1	16.4	29	52
Rio Nutrias	36° 49' 08"	106° 13' 19"	5.0	4.3	4.3	0.9	367	2785	20.8	8.8	1	79
Rito de la Olla	36° 16' 22"	105° 25' 00"	17.5	10.6	11.7	0.7	764	2868	28.3	12.2	17	37
Rito del Padre & Rito Maestas	35° 55' 56"	105° 35' 22"	12.1	7.4	7.4	0.6	750	3060	19.6	11.8	18	3
Palociento Cr	36° 15' 38"	105° 26' 40"	6.5	3.5	5.3	0.8	874	2768	45.5	22.8	3	28
Peralta Canyon	35° 41' 06"	106° 27' 24"	43.4	17.4	31.5	0.7	1164	1864	41.2	14.8	20	24
Rito de los Pinos	36° 06' 05"	105° 54' 24"	5.9	3.7	4.6	0.8	699	2477	38.0	19.3	50	39
Policarpio Canyon	36° 08' 20"	105° 27' 11"	7.6	3.6	4.6	0.6	463	2772	20.2	12.9	39	51
Polvadera Cr	36° 06' 59"	106° 26' 20"	50.8	32.4	41.5	0.8	1022	2396	28.7	12.4	8	13
Rito la Presa	36° 10' 05"	105° 27' 07"	23.3	9.8	15.3	0.7	838	2804	32.6	13.3	3	12
Rio Puerco	36° 03' 19"	106° 53' 07"	14.5	11.5	13.3	0.9	613	2615	17.4	9.9	21	22
Rio Quemado	36° 01' 00"	105° 47' 29"	52.3	24.5	42.7	0.8	1654	2334	37.0	15.3	19	38
Rito Resumidero & Oso Cr	36° 06' 03"	106° 44' 16"	24.9	8.4	21.6	0.9	592	2643	15.5	10.4	54	61

Appendix V. Concluded.

Saloz Cr	36° 12' 19"	105° 31' 16"	11.3	3.9	6.3	0.6	771	2656	34.7	15.5	15	53
Sardinas Canyon	36° 11' 12"	105° 27' 20"	7.6	5.1	5.1	0.7	669	2973	33.9	16.9	30	30
San Cristobal Cr	36° 37' 00"	105° 36' 22"	11.8	6.1	8.3	0.7	1178	2509	43.3	16.9	56	89
South Fork Rio Hondo	36° 34' 24"	105° 30' 28"	19.3	7.2	12.6	0.7	1356	2559	54.4	25.2	15	63
Tienditas Cr	36° 22' 18"	105° 22' 27"	39.5	9.1	29.9	0.8	776	2541	22.3	11.3	6	5
Rio de Truchas	36° 05' 09"	105° 52' 19"	44.8	21.0	43.3	1.0	1477	2141	19.8	7.9	18	59
Rio Valdez	35° 56' 13"	105° 32' 15"	5.2	3.1	3.1	0.6	345	3293	14.5	9.6	49	39
Vidal Cr	36° 45' 24"	105° 16' 06"	25.9	8.0	15.4	0.6	532	2845	17.7	4.2	32	8
Yerba Cr	36° 34' 11"	105° 31' 04"	6.2	3.7	4.3	0.7	1083	2522	46.9	21.2	45	40