Rio Grande Cutthroat Trout (*Oncorhynchus clarkii virginalis*): A Technical Conservation Assessment

Prepared for the USDA Forest Service, Rocky Mountain Region, Species Conservation Project

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Victoria L. Pritchard, Ph.D. and David E. Cowley, Ph.D.
Department of Fishery and Wildlife Sciences,
New Mexico State University,
Las Cruces, NM 88003

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AUTHORS' BIOGRAPHIES

Dr. Victoria L. Pritchard obtained her PhD in fish behavior and quantitative genetics from the University of Leeds, U.K. She has recently completed a program of post-doctoral research within the Department of Fishery and Wildlife Sciences at New Mexico State University, examining population genetic structure and levels of non-native introgression in Rio Grande cutthroat trout.

Dr. David Cowley is an assistant professor in the Department of Fishery and Wildlife Sciences at New Mexico State University. He has worked with Rio Grande cutthroat trout in New Mexico since 1993. Dr. Cowley's research currently is focused on factors that structure fish communities at local, reach, and landscape scales. He and his students are studying how fish communities respond to the introduction of exotic species, flow intermittency, river regulation, and drought.

COVER ILLUSTRATION CREDIT

Rio Grande cutthroat trout (*Oncorhynchus clarkii virginalis*). Illustration by © Joseph Tomelleri.

SUMMARY OF KEY COMPONENTS FOR CONSERVATION OF THE RIO GRANDE CUTTHROAT TROUT

Status

The Rio Grande cutthroat trout (*Oncorhynchus clarkii virginalis*) is a subspecies of cutthroat trout occurring in the Rio Grande, Pecos, and Canadian drainages of southern Colorado and northern New Mexico. The subspecies has declined precipitously in the past two centuries, and it is currently believed to occupy a fraction of its previous native range. Approximately 200 self-reproducing populations phenotypically corresponding to Rio Grande cutthroat trout are known to exist. The majority of these occur on USDA Forest Service (USFS) lands within the Rocky Mountain (Region 2) and Southwest (Region 3) regions, which include the Rio Grande and the Carson and Santa Fe national forests, respectively. Populations are spatially restricted, highly fragmented, and primarily confined to headwater streams, which in some cases may represent marginal trout habitat. Many populations contain genetic material from non-native trout taxa.

Federal protection for the Rio Grande cutthroat trout under the Endangered Species Act was ruled 'not warranted' in 2002. However the subspecies is recognized as a species of special concern in both Colorado and New Mexico, and as a sensitive species within USFS Regions 2 and 3 and by the Bureau of Land Management in Colorado. The Nature Conservancy assigns the Rio Grande cutthroat trout a Global Heritage Status Rank of G4T3, which means that on a global basis, while the species is apparently secure, the subspecies is vulnerable. The subspecies has been the subject of multiple activities intended to improve its status in the past few decades, and as a result the current population trend appears to be stable. However, maintenance of this trend requires ongoing active management.

Primary Threats

The primary threat to Rio Grande cutthroat trout today is the presence of non-native trout, which have been introduced in vast numbers into New Mexico and Colorado over the past century and now occupy most suitable habitat within the subspecies' native range. Rainbow trout (*Oncorhynchus mykiss*) and non-native subspecies of cutthroat trout (*O. clarkii* spp.) cause loss of Rio Grande cutthroat trout populations via hybridization while brook charr (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) appear to cause population declines via predation or competitive exclusion. As a result of this threat, extant Rio Grande cutthroat trout populations require protection from the incursions of non-native trout by natural or artificial migration barriers. Such barriers also act to prevent gene flow between these extant populations.

In their present distribution, Rio Grande cutthroat trout are also at risk from anthropogenic and natural habitat disturbance, disease transmission, and the negative effects of population fragmentation. Anthropogenic habitat disturbance is believed to have been one cause of Rio Grande cutthroat trout decline in the late 19th and early 20th centuries. Grazing, logging, mining, road construction, and water extraction have all been demonstrated to impact cutthroat trout habitat. Natural events that may negatively affect Rio Grande cutthroat trout populations include wildfires and anchor ice formation. The headwater streams to which the subspecies is generally restricted are often characterized by extreme and fluctuating physical environments, and habitats are not easily re-colonized following local population extinctions. Rio Grande cutthroat trout are also highly susceptible to whirling disease, which has been introduced into several drainages occupied by the subspecies and is present in at least one population. The small size and isolation of many extant populations means that they are at increased vulnerability to extinction as a result of demographic stochasticity and reductions in fitness due to population genetic processes.

Primary Conservation Elements, Management Implications and Considerations

Populations of Rio Grande cutthroat trout require protection from non-native trout, introduced diseases, and habitat degradation. The security of the subspecies will be improved by eliminating co-existing, non-native trout; expanding the quantity and quality of habitat available to existing populations; creating new, self-sustaining populations within the historic range; re-establishing gene flow between isolated populations; and appropriately developing and using broodstocks. Management decisions will be informed by, among other things, knowledge of a population's genetic purity and abundance, presence of non-native trout, habitat characteristics, and the outcome of scientific studies.

Protection of Rio Grande cutthroat trout from both non-native trout and disease can be achieved by isolating populations using migration barriers. In some cases, a sufficient natural or artificial barrier is present; otherwise a barrier can be constructed. Barriers require monitoring and maintenance to ensure that they continue to exclude unwanted fish. Protection of the subspecies from non-native trout and disease is additionally achieved via policies and regulations concerning fish stocking.

Established protocols are available to assess habitat condition for stream-dwelling salmonids. Aspects of habitat shown to be important for cutthroat trout include availability of cover and number of deep pools, availability of sediment-free spawning gravels and fry rearing habitat, and summer water temperatures. Several management tools are available to protect Rio Grande cutthroat trout habitat from anthropogenic impacts where these are deemed to be a threat. These tools may include regulation of grazing, management of timber harvest activities to protect riparian areas, correct maintenance of roads, and establishment and purchase of water rights. In addition, methods are available to restore and improve trout habitat where this is considered necessary.

Expansion of existing Rio Grande cutthroat trout populations and establishment of new populations can be achieved by translocating wild or hatchery-produced fish into suitable habitat or by creating conditions that allow natural re-colonization. In most cases, removal of non-native fish will be required before a new population of Rio Grande cutthroat trout can be established. This is most commonly achieved by using piscicides. Recommendations are available regarding the habitat attributes and stocking strategies that will maximize the chance of population establishment and persistence. Management agencies have established hatchery stocks of Rio Grande cutthroat trout to be used in population restoration efforts.

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INTRODUCTION

Goal of Assessment

This assessment is one of many being produced as part of the USDA Forest Service (USFS) Rocky Mountain Region (Region 2) Species Conservation Project. These assessments are intended to provide forest managers, research biologists and others with a thorough discussion of the biology, ecology, conservation status, and management of a taxon based on available scientific knowledge. An important purpose of the species assessments is to provide information that managers can use to make management decisions. However, these assessments do not seek to develop specific management recommendations. Instead, they present recommendations made elsewhere in regard to the management of the taxon. The assessment goals limit the scope of the work to critical summaries of scientific knowledge, discussion of broad implications of the knowledge, and outlines of information needs.

Scope of Assessment

This assessment examines the biology, ecology, conservation status, and management of the Rio Grande cutthroat (*Oncorhynchus clarkii virginalis*; **Figure 1**), a subspecies listed both as a sensitive species and as a Management Indicator Species (MIS) in USFS Region 2. The Rio Grande cutthroat trout has a contiguous native distribution over parts of USFS Regions 2 and 3. Since the subspecies occurs in similar habitat and is subject to similar threats and similar management activities throughout its range, this assessment utilizes information collected from both Regions. We provide more in-depth discussion of its status in Region 2 where this is applicable.

In producing the assessment, we relied on peer-reviewed scientific literature, non-peer-reviewed publications, research and management reports, data collected by management agencies, and occasionally personal communication from individuals knowledgeable in the field. There is relatively little information available, particularly in the peer-reviewed scientific literature, on the biology and ecology of Rio Grande cutthroat trout. However, a much larger body of scientific knowledge exists regarding inland cutthroat trout and stream-dwelling salmonids in general. Therefore, we include information collected from studies of other taxa where this is considered appropriate. Unpublished reports and data were important sources of information on the subspecies. We performed few new statistical analyses of our own; many data collected by management agencies on attributes of Rio Grande cutthroat trout populations have not yet been formally compiled or analyzed and may represent a further source of information that we have not exhaustively investigated.

Treatment of Uncertainty and Application and Interpretation Limits

Due to the relative paucity of available information regarding Rio Grande cutthroat trout, we extrapolate from studies of closely-related inland cutthroat trout subspecies and from studies of the habitat requirements of stream-dwelling salmonids in general, in order to draw conclusions regarding the biology, ecology, and management of the subspecies. While we believe such extrapolations to be largely valid, they may not be accurate in all cases. Similarly, conclusions drawn from data collected from a subset of Rio Grande cutthroat trout populations may not hold true throughout its entire range. Throughout this assessment we note whether

Figure 1. Rio Grande cutthroat trout. Photograph taken by David E. Cowley.

the information that we present is derived from studies of Rio Grande cutthroat trout or from studies of other salmonid taxa.

Treatment of This Document as a Web Publication

To facilitate the use of species assessments in the Species Conservation Project, they are being published on the Region 2 World Wide Web site (www.fs.fed.us/r2/projects/scp/assessments/index.shtml). Publication of the documents on the internet makes them available more rapidly than paper publication and facilitates their future revision.

Peer Review of This Document

Assessments developed for the Species Conservation Project have been peer-reviewed prior to their release on the Web. Peer review for this assessment was administered by the American Fisheries Society, which employed two recognized experts for this or related taxa. Additional peer reviews were provided by the Colorado Division of Wildlife (CDOW) and New Mexico Department of Game and Fish (NMDGF).

MANAGEMENT STATUS AND NATURAL HISTORY

Management Status

U.S. Fish and Wildlife Service

The U.S. Fish and Wildlife Service received a petition in 1998 to list the Rio Grande cutthroat trout under the Endangered Species Act. In a 90-day finding, the agency concluded that listing was not warranted. However, in 2001 a candidate status review was initiated in response to litigation appealing this decision and new information, particularly regarding the presence of whirling disease within the native range of the subspecies (U.S. Fish and Wildlife Service 2002). The results of this review were published in 2002, and it was again determined that listing of this taxon was not warranted (U.S. Fish and Wildlife Service 2002). In 2005, a petition for Review of Agency Action regarding the 'not warranted' decision was denied.

USDA Forest Service

The Rio Grande cutthroat trout occurs in the Rio Grande National Forest within USFS Region 2 (**Figure 2**) and in the Carson and Santa Fe national forests within USFS Region 3 (**Figure 3**). The subspecies is included in the Regional Forester's Sensitive Species List for both Region 2 and Region 3. Within the National Forest System, a sensitive species is a plant or animal whose population viability is identified as a concern by a Regional Forester because of significant current or predicted downward trends in abundance and/or habitat capability that would reduce its distribution (FSM 2670.5 (19)). Due to concerns with population viability and abundance, a sensitive species requires special management. Consequently, knowledge of its biology and ecology is critical. Sensitive species are considered in Biological Evaluations during project planning and analysis. The Rio Grande cutthroat trout is also classified as a MIS on the Rio Grande National Forest in Region 2 and on the Santa Fe and Carson national forests in Region 3. A species may be selected as a MIS for use in land management planning because changes in its population are believed to indicate the effects of management activities.

Bureau of Land Management

In Colorado, the Bureau of Land Management (BLM) lists the Rio Grande cutthroat trout on the State Director's Sensitive Species List. Policy states that the BLM should not fund, authorize, or implement any action that would contribute to taxa on this list becoming listed as a candidate, threatened, or endangered species under the Endangered Species Act. Environmental Assessments are required to analyze the effects of actions on species included on this list. In New Mexico, the BLM does not have a special status for the subspecies; however, in this state the BLM defers to the NMDGF on wildlife management issues.

State of Colorado

Currently, the Rio Grande cutthroat trout is recognized as a species of special concern in Colorado. The state previously listed the Rio Grande cutthroat trout as a state threatened species in 1973 and de-listed the subspecies in 1984 following achievement of the recovery goal to establish 10 stable Rio Grande cutthroat trout populations on public land in the state.

State of New Mexico

The Rio Grande cutthroat trout is designated a species of special management concern in New Mexico. The June 2003 conservation agreement for the range-wide preservation and management of Rio Grande cutthroat trout states, "Preservation and expansion of existing populations is a priority. Establishing metapopulations and monitoring fish health are crucial

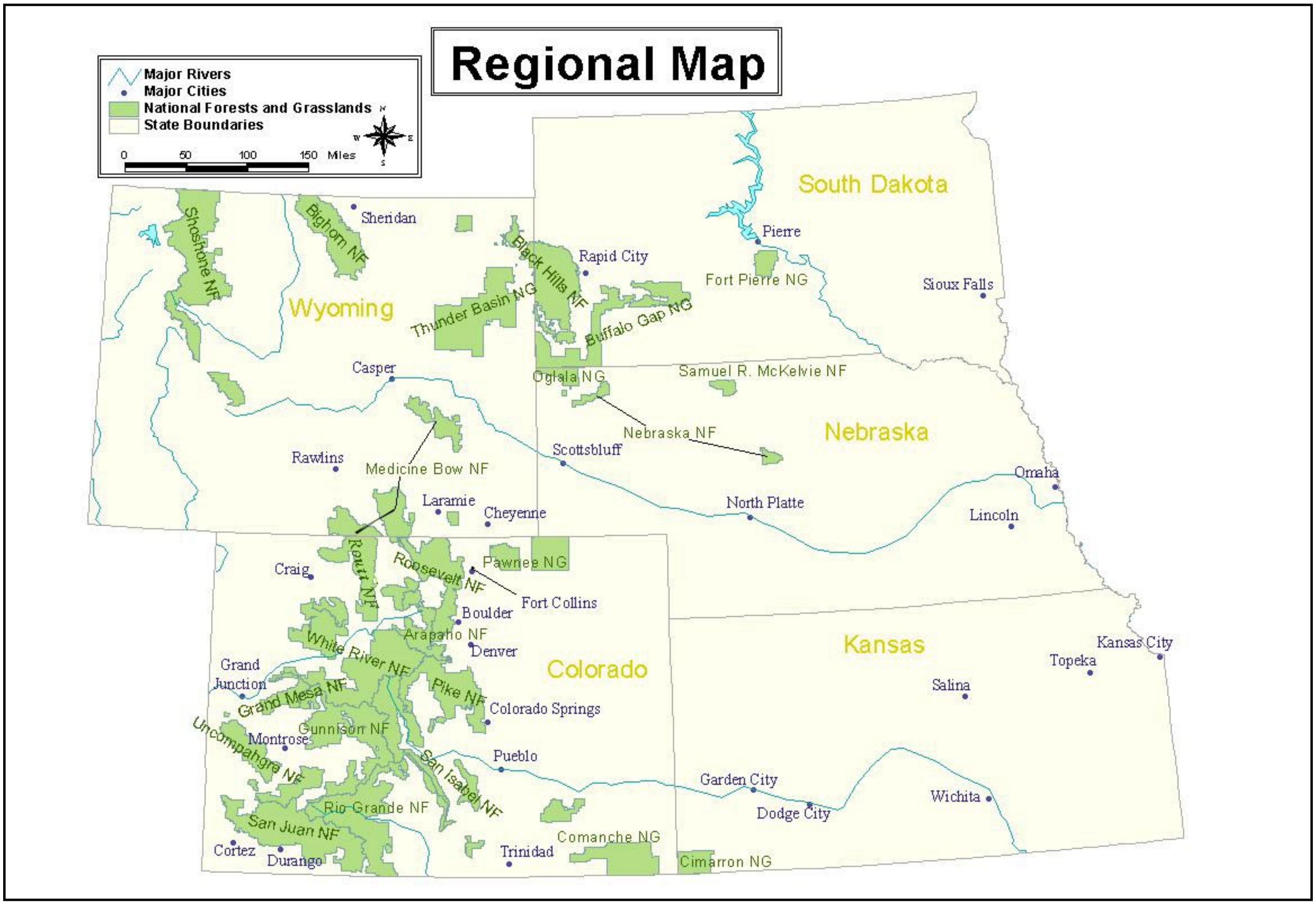

Figure 2. USDA Forest Service Rocky Mountain Region (Region 2) national forests and grasslands.

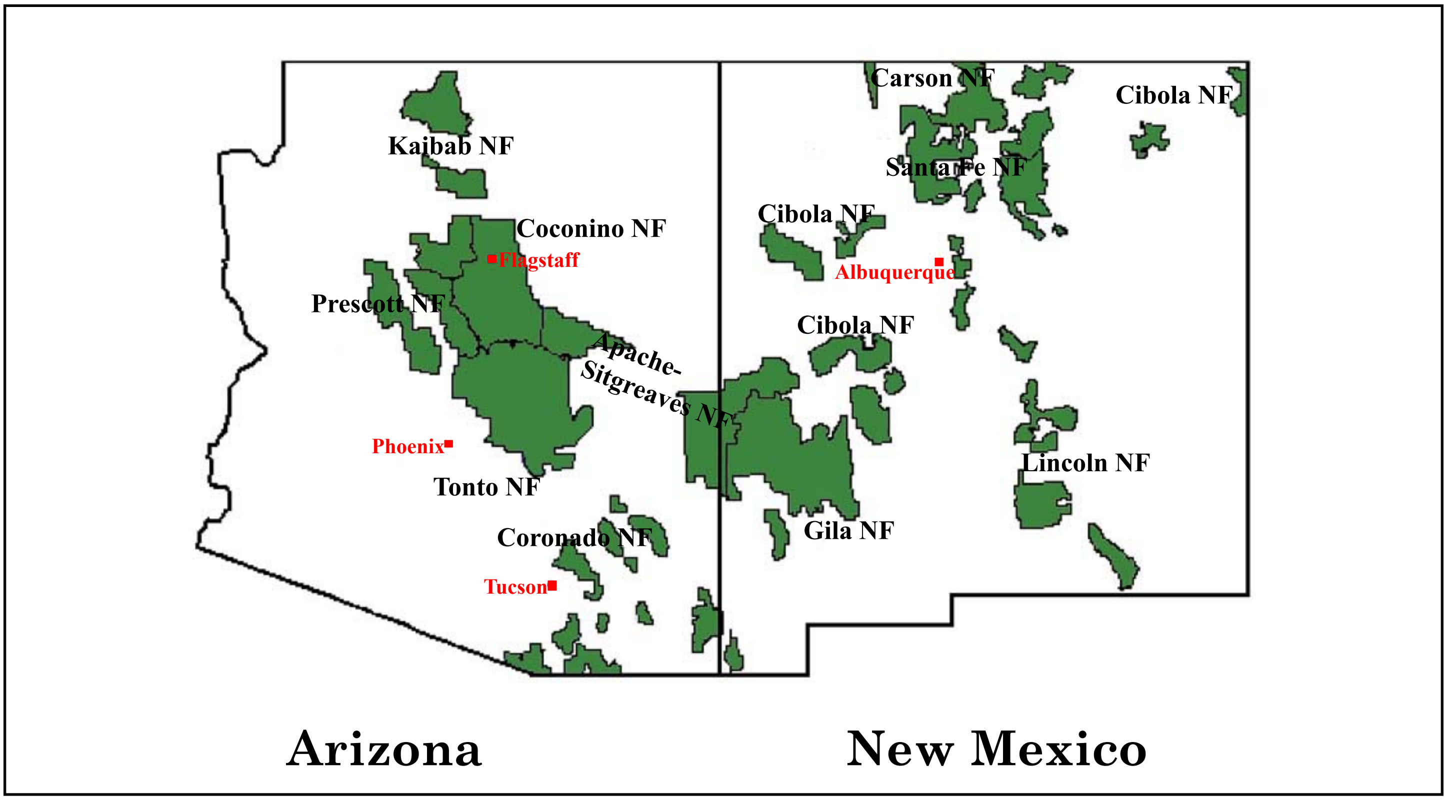

Figure 3. USDA Forest Service Southwest Region (Region 3) national forests and grassland in Arizona and New Mexico.

to support population viability and conservation efforts." Rio Grande cutthroat trout are currently considered a "species of greatest conservation need" in the New Mexico Department of Game and Fish's comprehensive wildlife conservation plan. (Patton personal communication 2006).

Other designations

The Nature Conservancy (http://natureserve.org/explorer) assigns the Rio Grande cutthroat trout a Global Heritage Status Rank of G4T3, which means that on a global basis, while the species is apparently secure, the subspecies is vulnerable. The subspecies is assigned a Subnational Conservation Status Rank of S3 (vulnerable) for Colorado and S2 (imperiled) for New Mexico.

Existing Regulatory Mechanisms, Management Plans and Conservation Strategies

USDA Forest Service

The Rio Grande, Carson, and Santa Fe national forests have developed forest-wide management goals and strategies intended to protect Rio Grande cutthroat trout (USDA Forest Service 1986, 1996, 2003). Land use management considerations as part of the Rio Grande National Forest land and resource management plan include riparian buffer maintenance and protection, sediment abatement, mining and logging restrictions, proper placement of recreational trails, minimal impact grazing strategies, quantitative habitat monitoring, and development of instream/riparian habitat restoration projects (USDA Forest Service 1996, 2003).

State of Colorado

CDOW has the authority and responsibility for the management of Rio Grande cutthroat trout on all Federal, State, and private land in Colorado. Management of the subspecies is guided by a state conservation plan, finalized in 2004 (Colorado Division of Wildlife 2004). This plan sets forth strategies to protect, monitor, and assess the status of existing Rio Grande cutthroat trout populations, to expand the range of the subspecies, and to restore degraded habitat. Native cutthroat trout populations in Colorado are protected by state regulations concerning stocking restrictions, fishing closures, harvest and gear restrictions, stream barriers to fish passage, and disease control. State and federal agencies no longer introduce non-native salmonids into existing populations of Rio Grande cutthroat trout, and the Colorado stocking permit system prevents private stocking of non-natives into waters occupied by the subspecies. In 33 Rio Grande cutthroat trout waters judged potentially vulnerable to depletion by angler harvest, fishing is restricted to catch-and-release with fly and lure only; otherwise daily bag limit for trout is four, with eight allowed in possession (Colorado Division of Wildlife 2005). Policies and regulations are in place to prevent the spread of whirling disease into Rio Grande cutthroat trout populations. Trout from hatcheries that test positive for whirling disease are no longer stocked into waters capable of supporting self-sustaining trout populations (Nehring 2006). The state has fish health inspection requirements for public and private hatcheries and fish rearing facilities, and fish imported from outside Colorado are required to have fish health certificates. Rio Grande cutthroat trout broodstock are tested for whirling disease infection prior to stocking into existing populations or restoration waters. A policy is also in place requiring the use of isolation or quarantine units while propagating native cutthroat stocks to decrease the risk of transmission of salmonid pathogens.

State of New Mexico

In New Mexico, the NMDGF has the authority and responsibility for the management of Rio Grande cutthroat trout on all Federal, State, and private land, and the authority to regulate those impairments to population viability of Rio Grande cutthroat trout that arise from sport fishing, stocking, and elk management (New Mexico Statutes Annotated 1978). The Department does not, however, have statutory authority to regulate consumptive water use, dam construction, grazing, mining, construction or maintenance of roads and trails, or timber harvest on land that it does not own (New Mexico Department of Game and Fish 2002). Management is guided by the 'Long Range Plan for the Management of Rio Grande Cutthroat Trout in New Mexico' (New Mexico Department of Game and Fish 2002). Sport fishing regulations set a daily bag limit of two cutthroat trout, and several Rio Grande cutthroat trout waters are currently designated catch-and-release only. NMDGF is also developing a Rio Grande cutthroat trout broodstock under the direction of a broodstock management plan (Cowley 1993) that is currently under revision (Cowley and Pritchard 2003). Policies and regulations are in place to prevent the spread of whirling disease into Rio Grande cutthroat trout populations; whirling disease-positive fish are destroyed.

Multi-party agreements

A conservation agreement for the range-wide preservation and management of Rio Grande cutthroat trout was signed in June 2003 by NMDGF, CDOW, USFS Regions 2 and 3, U.S. Fish and Wildlife Service Regions 2 and 6, National Park Service, BLM, and the Jicarilla Apache Nation. This agreement has the goal of "assuring the long-term persistence of the Rio Grande cutthroat trout subspecies within its historic range by preserving its genetic integrity, reducing habitat fragmentation, and providing sufficient suitable habitat to support adequate numbers of viable, self-sustaining populations." A primary objective of the agreement is to implement a formal process of cooperation, co-ordination, and data sharing amongst the signatory agencies. The initial duration of the Agreement is five years.

Management of introgressed populations

A Position Paper on genetic purity considerations associated with cutthroat trout management was developed co-operatively between seven state wildlife agencies at a meeting in Salt Lake City, Utah (Utah Division of Wildlife Resources 2000). This paper sets out recommendations for genetic analysis techniques for quantifying levels of introgression from non-native trout and recommends management approaches for dealing with populations with differing levels of genetic purity (see later discussion). This paper will hereafter be referred to as the 'Utah Position Paper'.

Biology and Ecology

There are rather few published studies pertaining to the biology and ecology of Rio Grande cutthroat trout. However, it is likely that many aspects of its ecology and life-history are similar to those of other interior cutthroat trout subspecies occupying stream habitats. This section therefore reviews what is known about cutthroat trout in general, with data included from Rio Grande cutthroat trout where they are available.

Systematics and general species description

The cutthroat trout (*Oncorhynchus clarkii*, Order Salmoniformes, Family Salmonidae) is a member of the genus *Oncorhynchus*, which also includes rainbow or steelhead trout (*O. mykiss*), golden trout (*O. chrysogaster*), Gila trout (*O. gilae*), and five species of Pacific salmon (*O. kisutch*, *O. tshawytscha*, *O. nerka*, *O. keta*, and *O. gorbushcha*). Along with rainbow trout, golden trout, and Gila trout, the species was formerly classified in the genus *Salmo*; hence literature prior to 1989 refers to the cutthroat trout as '*Salmo clarkii*'. The species name has also previously been spelled '*clarki*'. *Oncorhynchus clarkii* is a polytypic species, comprising 14 described subspecies and several distinct racial forms that are distributed across western North America. Taxonomic differences among inland subspecies of cutthroat trout are based on geographical location, chromosome number, and variation in coloration, spotting patterns, and various meristic characters (Behnke 2002). The Rio Grande cutthroat trout is closely related to Yellowstone (*O. c. bouvieri*), Bonneville (*O. c. utah*), Colorado River (*O. c. pleuriticus*), and greenback (*O. c. stomias*) cutthroat trouts (Allendorf and Leary 1988, Behnke 2002). It is thought to have arisen as a result of headwater transfer of ancestral trout populations from the Colorado River into the Rio Grande during the Pleistocene, probably less than 100,000 years ago (Behnke 2002). The first written report of the subspecies, in the upper Pecos River of New Mexico, comes from Francisco de Coronado's expedition of 1541 (Behnke 2002), and it was first formally described from Utah (Ute) Creek, a tributary to the Rio Grande in Costilla County, Colorado, in 1857 (Girard 1857).

In common with all cutthroat trout subspecies, the Rio Grande cutthroat trout possesses a red to orange slash in the gular fold beneath the lower jaw. The subspecies also exhibits relatively large, irregular shaped dark spots that are concentrated posterior to the dorsal fin, but may also occur anterior to the dorsal fin above the lateral line (Sublette et al. 1990). Individuals are generally colorful, with light rose to red-orange hues on the sides and pink or yellow-orange on the belly. Colors are brighter on breeding adults, especially males. In the high-elevation headwater streams to which they are primarily restricted today, Rio Grande cutthroat trout remain relatively small (adult length = 120 to 300 mm [4.7 to 11.8 inches]; New Mexico Department of Game and Fish unpublished data, Paroz 2005). However, the subspecies may grow more than 400 mm (16 inches) in length under hatchery conditions (New Mexico Department of Game and Fish unpublished data). The subspecies differs from the closely related but allopatric greenback cutthroat trout and Colorado River cutthroat trout by having fewer scales in the lateral series and more pyloric caecae (Behnke 1992), but there is overlap in these features between the subspecies. It has been noted by some authors that fish from the Pecos River drainage tend to have larger spots and more scales in the lateral series than those from the Rio Grande drainage (Sublette et al. 1990, Behnke 2002). Basibranchial teeth are poorly developed or absent in Rio Grande cutthroat trout.

Distribution and abundance

Rio Grande cutthroat trout are known to be native to the Rio Grande and Pecos River drainages of Colorado and New Mexico (Behnke 2002). They are also believed to be native to the Canadian River drainage of Colorado and New Mexico, but no early historical specimens or written accounts are available to verify this (Behnke 2002). The subspecies may have also previously occurred in Rio Grande and Pecos tributaries in Texas (Garrett and Matlock 1991) and possibly in the headwaters of the Rio Conchas in northern Mexico (Hendrickson et al. 2002). Today, apparently remnant populations of Rio Grande cutthroat trout occur in tributaries to the Rio Grande in Colorado and New Mexico; in the Carnero and Sanguache drainages in Colorado, which are geologically part of the Rio Grande system but drain into the San Luis closed basin; in tributaries to the Canadian River in Colorado and New Mexico; and in tributaries to the Pecos River in New Mexico (**Figure 4**).

The colonization routes by which Rio Grande cutthroat trout arrived at their present distribution are not well understood. It is likely that they were able to migrate between the Rio Grande and Pecos drainages via their confluence in southern Texas until the end of the Pleistocene; however neither of these rivers shares a confluence with the Canadian system. As alternative scenarios, Rio Grande cutthroat trout may have been transferred between the three river systems, and into the streams terminating in the San Luis closed basin, via pluvial lakes (Bachhuber 1989), erosional stream capture (Trotter 1987), or anthropogenic transport. During post-glacial warming of the climate, the subspecies is thought to have become confined to more northern and higher altitude waters as a result of temperature and habitat requirements. The Rio Grande cutthroat trout is the only native trout in the river systems in which it occurs.

While the entire range of Rio Grande cutthroat trout immediately prior to European settlement of the American West cannot be known for certain, probable limits to its distribution can be inferred from several pieces of information. First, studies have suggested that cutthroat trout are unable to survive in waters that exceed 24 °C (75 °F) for extended periods (Dickerson and Vinyard 1999, Johnstone and Rahel 2003). Rio Grande cutthroat trout could have once occupied lower elevation reaches of the Rio Grande or Pecos River during colder months, but current climatic conditions would not be favorable for sustaining permanent populations at these lower elevations. The southernmost proven occurrence of Rio Grande cutthroat trout is a specimen collected from the Rio Grande mainstem at San Ildefonso Pueblo, northern New Mexico, in 1874. Putative historic populations in the Black Range and Tularosa basin of southern New Mexico appear to have originated via stocking of various subspecies of *Oncorhynchus clarkii* (Pritchard and Cowley 2005). Second, Rio Grande cutthroat trout evolved as a member of a native fish assemblage that included longnose dace (*Rhinichthys cataractae*), flathead chub (*Platygobio gracilis*), Rio Grande chub (*Gila pandora*), and Rio Grande sucker (*Catostomus plebeius*) (Hatch et al. 1998). Most members of this native fish assemblage are gravel spawners whose downstream breeding limit generally coincides with the transition from degradation (erosion) to aggradation (deposition) of fine sediments. This transition typically occurs near the base of mountain ranges where stream gradient becomes flatter and current velocity slows, and approximately coincides with areas experiencing less than 150 frost-free days annually (New Mexico Department of Game and Fish 2002). Finally, as has been suggested for other subspecies of cutthroat trout, the upstream distribution of Rio Grande cutthroat trout may have been limited by stream gradient, temperature, and migration barriers (Kruse et al. 1997, Dunham et al. 1999).

The distribution of the Rio Grande cutthroat trout has declined over the last 150 years as a result of a number of anthropogenic factors, including the introduction of non-native trout, habitat destruction, and over-fishing. Today the subspecies is primarily restricted to headwater streams, with some introduced populations also occurring in high-altitude lakes. NMDGF and CDOW document the presence of naturally reproducing populations of trout that phenotypically resemble Rio Grande cutthroat trout in approximately 200 water bodies (**Figure 4**; New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004). The exact number of such populations that currently exist is not known, both because undocumented extirpations are expected to have occurred and because some areas have not been exhaustively surveyed for the presence of cutthroat trout (New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004). At least 40 percent of these populations are known or suspected to contain genetic material from rainbow trout or non-native cutthroat trout (see later discussion; New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004, Douglas and Douglas 2005, Pritchard and Cowley 2005). Thirty-six naturally reproducing populations have been created by successful introduction or re-introduction of Rio Grande cutthroat trout into suitable habitat. In addition,

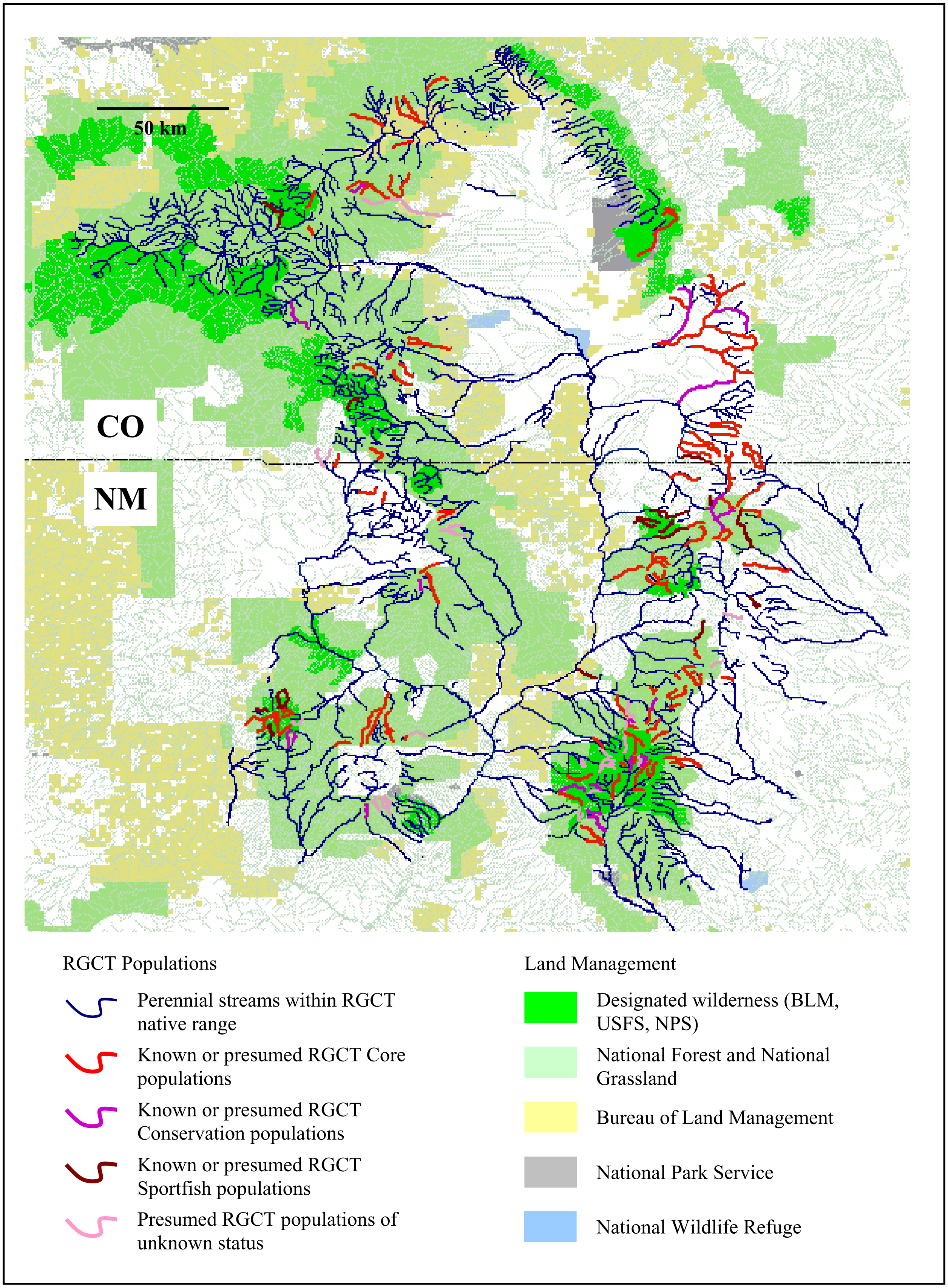

Figure 4. Current and presumed historic distribution of Rio Grande cutthroat trout in Colorado and New Mexico.

CDOW maintains 83 ‘recreation populations’ via periodical stocking of pure Rio Grande cutthroat trout from hatchery stocks (Colorado Division of Wildlife 2004), primarily into isolated high-elevation lakes where natural reproduction is unlikely to occur. Of the documented naturally reproducing Rio Grande cutthroat trout populations, approximately three quarters occur on public lands, primarily within the Carson, Rio Grande and Santa Fe National Forests, and approximately a quarter within designated wilderness areas.

Stumpff and Cooper (1996) estimate that Rio Grande cutthroat trout currently occupy around 10 percent of their original native range. In order to investigate this assertion in more detail we used a Geographic Information System approach to compare the length of perennial stream currently occupied by naturally reproducing populations of Rio Grande cutthroat trout with the length of perennial stream expected to be available to the subspecies historically (**Figure 4**). Hydrography data (1:100,000 scale) for New Mexico and Colorado were obtained from the USGS National Hydrography Dataset (NHD; http://nhd.usgs.gov). Probable limits to the subspecies’ native range at the start of the 19th century were inferred from observations of current distribution, historic records, information discussed above and maps provided in New Mexico Department of Game and Fish (2002) and Behnke (2002). Data on current distribution was obtained from New Mexico Department of Game and Fish (2002) and Colorado Division of Wildlife (2004). We included all known stream-dwelling Rio Grande cutthroat trout Core and Conservation populations (examined using genetic markers, see later discussion), suspected Core and Conservation populations (those in tributaries to Core and Conservation populations and/or examined using meristics only), and populations whose genetic status was unknown. We excluded any populations with sufficient levels of non-native introgression to be termed ‘Sportfish’. While we did not specifically include lakes in this analysis, most lakes occupied by Rio Grande cutthroat trout are part of occupied stream systems, and their length therefore made up a portion of estimated stream length. Using the program Arcview 3.3, we estimated just over 10,000 km (6,200 mi.) of stream habitat historically available to Rio Grande cutthroat trout, compared with approximately 1,150 km (713 mi.) of habitat currently supporting the subspecies. This suggests that the Rio Grande cutthroat trout currently occupies just over 11 percent of its original native range.

Although some of the perennial streams that we included in this analysis may not have historically supported Rio Grande cutthroat trout due to factors such as impassable waterfalls and poor habitat, we believe that the figure of 11 percent may overestimate the true proportion of historical habitat currently occupied for several reasons. First, in our analysis we were unable to take into account differences in stream width. Rio Grande cutthroat trout have primarily been eliminated from wider, higher-order streams and rivers and remain in narrower, lower-order streams. Thus, the area of habitat lost is expected to be greater than our estimate of 89 percent based on stream length alone. Second, the native range of Rio Grande cutthroat trout probably extended to many streams coded as ‘intermittent’ in the NHD dataset. A number of such ‘intermittent’ streams currently support extant populations, and many previously perennial streams are now ‘intermittent’ in Colorado and New Mexico as a result of water extraction. Third, in estimating the historic range of Rio Grande cutthroat trout, we ignored anecdotal reports of the subspecies occurring in the Black Range and Sacramento Mountains of southern New Mexico and in Pecos tributaries in Texas. Finally, in our analysis we included both Rio Grande cutthroat trout populations known to have low levels of non-native introgression and those whose genetic status was unknown, some of which have never been surveyed or have not been surveyed in the last decade. If we include only known or suspected ‘Core’ populations in our analysis, the estimated proportion of historical habitat currently occupied is reduced to just over 8 percent.

The distribution of Rio Grande cutthroat trout today is also highly fragmented; New Mexico Department of Game and Fish (2002) and Colorado Division of Wildlife (2004) document a mean occupied stream length of 7.6 km (4.7 mi.; range = <1 to 27.4 km or <1 to 17.6 mi.), and most populations are isolated from one another by migration barriers or the presence of intervening populations of non-native trout.

The abundance of Rio Grande cutthroat trout varies widely from population to population. For example, surveys of 56 Rio Grande cutthroat trout streams in Colorado, performed between 1986 and 2004, estimate mean per-population adult densities varying over a hundred-fold, from a minimum of 42 fish per hectare to a maximum of 4622 fish per hectare (17 to 1872 fish per acre; Colorado Division of Wildlife unpublished data). Similarly, surveys of 47 locations in New Mexico, where non-native trout were not present, estimate densities of Rio Grande cutthroat trout with length >80 mm (3.1 inches) varying from 238 per hectare to 12,818 per hectare (96 to 5189 per acre; Paroz 2005). Young et al. (2005) found that the densities of Colorado

River cutthroat trout in high-elevation streams were positively associated with amount of occupied habitat, so that population size increased as a function of the square of stream length. Kruse et al. (2001) found that density of Yellowstone cutthroat trout also increased with occupied stream length.

Several studies have demonstrated that abundance and size-class composition of cutthroat trout in a single stream can also vary widely from year to year (e.g., Benson 1960, Platts and Nelson 1988, House 1995, Schlosser 1995). This variation in abundance may be due to both environmental changes and stochastic population processes (Railsback et al. 2002). Results from 10 Rio Grande cutthroat trout populations in Colorado unaffected by competition from non-natives typically show a two to three-fold variation in adult density estimates between survey years. However, part of this variation is likely to be due to sampling error, and variation in estimated densities within a stream between years is generally much lower than variation in estimated densities between streams (Colorado Division of Wildlife unpublished data).

Population trend

As a result of ongoing management activities, the range-wide abundance and distribution of Rio Grande cutthroat trout appear to be stable, and the security of the subspecies has greatly improved over the past four decades. Since the 1970's, the subspecies has been re-introduced into numerous areas of suitable habitat within its native range, and on-going work focuses on securing, protecting, and improving habitat for extant populations (New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004). In addition, genetically pure populations of Rio Grande cutthroat trout continue to be identified (New Mexico Department of Game and Fish unpublished data). However, historic populations of Rio Grande cutthroat trout have continued to be lost over this time period (Harig and Fausch 1996, Alves 1996-2004, New Mexico Department of Game and Fish unpublished data), and the majority of extant populations remain vulnerable, in particular to invasion by non-native trout and the impact of low stream flows. The maintenance of a stable or increasing population trend for Rio Grande cutthroat trout requires continued active management. Under current conditions, if such management activities were to cease, the subspecies would be expected to resume a declining trend as a result of invasion of populations by non-native salmonids, stochastic environmental events, and the demographic and genetic factors associated with small, isolated populations.

Activity patterns and movements

No studies have been performed on the activity patterns and movements of Rio Grande cutthroat trout, but their habits are likely to reflect those of closely related taxa occurring in similar habitats. Although resident, stream-dwelling salmonids have previously been considered to be sedentary, recent studies have demonstrated that a large proportion of individuals may move frequently (Rodriguez 2002). Trout may move for a variety of different reasons (e.g., to escape adverse environmental conditions, predators, or competitors for resources; to complete different stages in their life history; to find breeding opportunities) (Hilderbrand and Kershner 2004b, Schrank and Rahel 2006). Although most movements are over short distances (several tens or hundreds of meters or yards), a few individuals may disperse much further (Rodriguez 2002, Colyer et al. 2005, Schrank and Rahel 2006). Movement of reproductively mature individuals between populations can maintain genetic diversity when the effective size of each population is small (see later discussion; Jensen et al. 2005).

Bonneville (*Oncorhynchus clarkii utah*), westslope (*O. c. lewisi*), and Yellowstone cutthroat trout occupying large interconnected river systems have been shown to migrate several hundred or thousand meters (several hundred yards to several miles) to suitable over-wintering areas (e.g., Brown 1999, Zurdstadt and Stephan 2004, Colyer et al. 2005). The fish generally move little during winter, aggregating in deep pools, beaver ponds, or areas of ground-water upwelling where they are able to avoid anchor-ice formation (Brown and Mackay 1995a, Harper and Farag 2004, but see Colyer et al. 2005). Subsequently, in the spring they migrate again to suitable spawning areas, primarily lower-order tributaries or main-stem or side channel spawning grounds (Brown and Mackay 1995b, Schmetterling 2001).

In contrast, a study of Colorado River cutthroat trout in montane stream habitat that is typical for Rio Grande cutthroat trout found no movement between summer and winter habitats, although, as in other studies, fish activity did decrease over winter. The lack of autumnal migration in this system may be because water temperatures in these high-elevation streams are cool year-round or because such habitats are insulated by snow cover during the winter and therefore are at little risk from anchor-ice formation (Lindstrom and Hubert 2004). Conversely, Young (1996) found that Colorado River cutthroat trout in the same habitat showed substantial summer movement, in the region of several hundred to several thousand meters. Much of this movement again appeared to be associated with migration to and from spawning habitat; however, trout continued to exhibit home range sizes of several hundred meters in the post-spawning period. Schmetterling and Adams (2004) found that westslope cutthroat trout in montane streams were also mobile during the summer; median distance moved was 91 m (83 yd.), but several individuals were observed to move further than 1.2 km (0.75 mi.).

Schmetterling and Adams (2004) suggest that cutthroat trout in smaller streams may need to move more extensively than those in larger water bodies because the various habitat types required by trout are more widely dispersed in such systems. This suggestion is supported by several studies showing that trout move less where stream channels are more complex, for example as a result of increased levels of large woody debris (Harvey et al. 1999, Roni and Quinn 2001). Schrank and Rahel (2006) found that summer movements of Bonneville cutthroat trout depended on fish size, with larger individuals remaining in deep pools and smaller individuals moving more frequently. In contrast, Hilderbrand and Kersher (2004) found that larger individuals of the same subspecies tended to move more frequently. An important reason for movement in cutthroat trout may be access to food resources; several studies have suggested that more mobile cutthroat trout may exhibit better condition, or greater improvement in condition, than less mobile individuals (Roni and Quinn 2001, Hilderbrand and Kershner 2004, Schrank and Rahel 2006).

As well as seasonal movement patterns, cutthroat trout may also exhibit small scale diurnal movements. Schmetterling and Adams (2004) and Hilderbrand and Kershner (2000b) reported that westslope and Bonneville cutthroat trout tended to remain in low velocity areas at night, moving into higher velocity areas during the day in order to feed. Harvey et al. (1999) found that adult stream-dwelling coastal cutthroat trout tended to occupy habitats providing cover during the day and moved to more open habitats at night. Young (1996), however, found no such diurnal movement patterns in Colorado River cutthroat trout.

It is possible that, historically, Rio Grande cutthroat trout exhibited a variety of different migration patterns and levels of vagility. These would have been influenced by demographic, genetic, and environmental factors and are expected to have varied both within and between populations. Westslope cutthroat trout and Yellowstone cutthroat trout, for example, exhibit both a river-dwelling 'fluvial' form that migrates into smaller streams to spawn, and a 'stream-resident' form that completes its entire life cycle in these smaller tributaries (Behnke 2002). Any such 'fluvial' life history strategy occurring within Rio Grande cutthroat trout would have been lost with the extirpation of the subspecies from the Rio Grande mainstem and fragmentation of habitat.

Habitat

Prior to the arrival of Europeans in the American West, Rio Grande cutthroat trout probably occupied a variety of fluvial habitats, ranging from first-order streams to the Rio Grande mainstem. Today, however the subspecies is excluded from most suitable habitat by the presence of non-native trout and is primarily restricted to small, high-elevation streams (commonly channel type A3/4, B3/4, C3/4 and E3/4; Rosgen 1996) and lakes. Such water bodies may be sub-optimal in a number of aspects; they may suffer extreme and fluctuating environmental conditions (Novinger and Rahel 2003), lack some habitat types important for cutthroat trout survival and reproduction (Harig and Fausch 2002), and provide insufficient refuge from natural and anthropogenic habitat disturbance. Rio Grande cutthroat trout are expected to exhibit similar habitat requirements to other trout taxa. In general, cutthroat trout appear to be more generalist in their habitat use than certain other *Oncorhynchus* species (Bisson et al. 1988). Cutthroat trout may also be better able to exploit higher gradient habitats than certain other salmonid species, including brown trout and brook charr (Bozek and Hubert 1992). Quist and Hubert (2005) found that, in the absence of co-existing non-natives, densities of cutthroat trout in the Salt River watershed were positively related to stream gradient. Latterell et al. (2003) found coastal cutthroat trout to occur frequently in streams with a 10 percent gradient and to be able to access channels with up to a 22 percent gradient. Dunham et al. (1999) showed Lahontan cutthroat trout to utilize a similar range of stream gradients, and Rio Grande cutthroat trout in New Mexico also occur in streams with mean gradients up to 20 percent (Pritchard et al. submitted).

Trout require several different habitat types according to life stage and season. A scarcity of any of these habitat types is expected to limit cutthroat trout abundance (Bjorn and Reiser 1991). Areas of suitable gravels that are well-oxygenated by flowing water and relatively free of fine sediment are needed for successful spawning and egg development (see later discussion; Magee et al. 1996). Following emergence, cutthroat trout fry move to areas of slow-moving, shallow water (velocities generally <0.06 m/s, depths generally <20 cm; Moore and Gregory 1988b, Bozek and Rahel 1991, 1992) such as margins, backwaters, and side channels (‘lateral habitats’), or small, low velocity pools created by physical obstructions in riffle areas (Moore and Gregory 1988a, b, Bozek and Rahel 1991, Rosenfeld et al. 2000, Hubert and Joyce 2005). Detrital loads and hence number of benthic invertebrates are frequently high in such areas (Moore and Gregory 1988a, b). The fry establish individual territories in these habitats, generally near a source of cover such as aquatic plants or overhanging vegetation, and remain in them for several months (Moore and Gregory 1988a, b, Hubert and Joyce 2005). The availability of such rearing habitat may in some cases be a limiting factor for survival of age 0 cutthroat trout; Moore and Gregory (1988a) demonstrated a positive correlation between numbers of cutthroat trout fry and area of lateral habitat in a third order stream in the Cascade Mountains, Oregon. Bozek and Rahel (1991), in contrast, found no relationship between density of Colorado River cutthroat trout young and the amount of suitable rearing habitat in the North Fork Little Snake River, Wyoming, perhaps because numbers of fry were limited by the availability of spawning gravels. Juvenile cutthroat trout may use stream substrate as cover during winter; hence high levels of fine sediment may reduce overwinter survival (McIntyre and Rieman 1995).

As cutthroat trout increase in size (e.g. >50 mm), they move back into higher velocity waters in the main stream channel (Moore and Gregory 1988a, b). Older trout in streams primarily utilize pools, and, to a lesser extent, riffle areas, rarely being found in rapids and cascades (Herger et al. 1996, Young et al. 1998). Numerous studies have demonstrated deep pools to be important for cutthroat trout, both as summer and overwintering habitat (e.g., Spangler and Scarnecchia 2001, Dare et al. 2002). Inhabiting pools is energetically less costly for trout than remaining in higher-velocity riffle areas (Rosenfeld and Boss 2001). Pools provide a refuge against elevated summer temperatures, terrestrial predators, and winter ice formation. In general, salmonids favor pools created by large woody debris, boulders, or lateral scour beneath stream banks (Bisson et al. 1988, Griffith and Smith 1993). Large woody debris appears to be particularly important for pool formation in high-elevation streams (Fausch et al. 1995). Cover such as that provided by undercut stream banks is generally considered to be another important element of salmonid habitat (e.g., McMahon and Hartman 1989), but it may be less important to cutthroat trout than to other stream-dwelling trout. Horan et al. (2000) found greater densities of Colorado River cutthroat trout in survey sites with higher percentages of undercut bank. Young (1996), however, observed that habitat created by large woody debris appeared to be more important to this subspecies than habitat created by undercutting, and noted that both juvenile and adult Colorado River cutthroat trout appeared to use cover infrequently. Wilzbach (1985) suggested that prey availability was more important than cover for juvenile coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) in high-elevation streams.

Water temperature is also an important component of cutthroat trout habitat. With low spring and summer temperatures, spawning will be delayed, and fry may not reach a sufficient size to survive the following winter (Scarnecchia and Bergersen 1986). Harig and Fausch (2002) found that summer water temperature (>7.8 °C mean daily temperature for July), in combination with pool width and number of pools, was the best predictor of the success of transplanted greenback and Rio Grande cutthroat trout populations in high-elevation streams in New Mexico and Colorado. Young and Guenther-Gloss (2004) correspondingly found the abundance of juvenile greenback cutthroat trout in 12 streams to be positively correlated with summer stream temperature. Peterson et al. (2004a) suggest that low water temperature caused recruitment failure in Colorado River cutthroat trout populations restricted to high-elevation streams. Conversely, high summer water temperatures may lead to trout mortality as a result of heat stress. Most salmonids are in danger at temperatures above 23 to 25 °C (Bjornn and Reisner 1991). Meeuwig et al. (2004), for example, demonstrated reduced feeding and growth of Lahontan cutthroat trout (*Oncorhynchus clarkii henshawi*) at 24 °C compared to 12 °C and 18 °C. Isaak and Hubert (2004) found that peak cutthroat trout biomass and density within the Salt River drainage of Idaho and Wyoming occurred where mean summer water temperature approximated 12 °C, declining as mean temperature increased or decreased. Dunham et al. (2003) found that the downstream distributional limit of Lahontan cutthroat trout in streams in Nevada and Oregon corresponded with a mean July air temperature of 18 °C. Water temperature may also influence the outcome of competitive interactions between cutthroat trout and non-native salmonids (see later discussion; Dunham et al. 2002). Water temperature may therefore be one factor determining the probability of invasion of a cutthroat trout populations by non-natives. McHugh and Budy (2005) suggest that the observed altitudinal segregation of brown trout and Bonneville cutthroat trout in streams (de la Hoz Franco and Budy 2005) may occur because brown trout are more limited by colder water temperatures. Water temperature can vary markedly within and among adjacent streams as a result of local landscape characteristics such as riparian vegetation and channel morphology (Sloat et al. 2002, Gardner et al. 2003).

A number of anthropogenic activities have been demonstrated to negatively impact habitat quality for trout. These include grazing, logging, road and trail construction, mining, and water diversion (Meehan 1991, Stumpff and Cooper 1996). Such activities can have a number of interacting effects on Rio Grande cutthroat trout habitat (e.g., changes in channel morphology, elevated summer water temperatures, increased deposition of fine sediments, reduction in stream flow and water pollution). Excessive grazing pressure, for example, can reduce bank stability via removal of riparian vegetation and mechanical damage; this in turn can lead to widening of the stream channel, reducing the number of deep pools, increasing fine sedimentation, and causing more extreme fluctuations in water temperature as a result of changes in channel morphology and reduced shading (Platts 1991). Riparian areas commonly offer flatter terrain, improved forage quality, and increased water availability compared to other range habitats; they therefore may be disproportionately used by livestock (e.g., Platts and Nelson 1985). Multiple studies have reported habitat degradation resulting from grazing pressure, decreases in trout abundance with grazing or increases in trout abundance with cessation of grazing (e.g., Platts 1991, Knapp and Matthews 1996). Timber harvest can similarly impact riparian vegetation and hence stream morphology, habitat conditions, and availability of food (Chamberlin et al. 1991, Wipfli 1997). Removal of timber adjacent to the stream will also remove a source of large woody debris, which is important in structuring stream morphology, causing the retention of sediments and organic matter, and providing nutrient inputs. Large woody debris from conifers persists longer than that from deciduous species and is therefore considered to be particularly important in generating stable habitat for salmonids (Gregory et al. 1991). Timber management activities, such as clear-cutting, will additionally affect basin-wide hydrologic and erosional processes, and use of forest chemicals, for example for disease control (Norris et al. 1991), can impact aquatic ecosystems. Road construction and improper road maintenance are also associated with changes in hydrologic and erosional processes and often cause increased deposition of fine sediment in streams (Furniss et al. 1991, Eaglin and Hubert 1993). Poorly-designed culverts under roads can act as barriers to fish movement. Trail construction and use of off-road vehicles and pack animals can also cause local-scale changes in drainage processes and increase deposition of fine sediment (Clark and Gibbons 1991). Mining and associated activity can similarly result in changes in hydrologic and erosional processes and hence changes in channel morphology and increased sedimentation, however mining is particularly associated with chemical pollution of water bodies (Nelson et al. 1991). Meehan (1991) discusses the influence of forest and rangeland management practices on salmonid habitat in more detail. Independent of local management activities, the high elevation habitats to which Rio Grande cutthroat trout are restricted may also be vulnerable to acidification (Farag et al. 1993) as a result of air pollution. Laboratory studies have demonstrated detrimental effects of lowered pH and associated elevated aluminum levels on early life stages of Yellowstone and greenback cutthroat trout (Farag et al. 1993, Woodward et al. 1991). The projected global warming trend is also expected to have some impact on Rio Grande cutthroat trout populations, although the direction of this impact is unclear. Cutthroat trout may benefit from warmer temperatures in headwater streams, however projected changes in precipitation may be detrimental (Cooney et al. 2005).

As a result of anthropogenic impacts, habitat quality for trout may be significantly reduced outside wilderness areas. Kershner et al. (1997), for example, compared wilderness and non-wilderness stream reaches in the Uinta Mountains and documented poorer habitat quality and correspondingly lower densities and condition of adult Colorado River cutthroat trout in streams outside the wilderness boundary. However, not all forest use activities are expected to have negative impacts on trout habitat, and in certain cases positive effects have been documented. Wilzbach et al. (1986), for example, found increased growth rates in cutthroat trout in logged compared to non-logged areas, apparently as a result of greater prey abundance and increased foraging efficiency as a result of more surface light. In many cases, invasion of Rio Grande cutthroat trout habitat by non-native trout appears to have been prevented by the presence of mining pollution, water extraction activities or road or rail crossings downstream from the extant Rio Grande cutthroat trout population (**Table 1**; Alves 1994 - 2004, New Mexico Department of Game and Fish unpublished data).

Natural processes that may impact current cutthroat trout habitat include beaver activity, flash floods, drought and forest health problems that impact watershed vegetation. Beaver dams may benefit Rio Grande cutthroat trout by creating suitable over-wintering habitat and providing barriers to the movement of non-native trout, however

Table 1. Name, origin, land ownership, and genetic management category for self-sustaining Rio Grande cutthroat trout populations within USDA Forest Service Region 2 that are documented in the Colorado Division of Wildlife Conservation Plan (Colorado Division of Wildlife 2004). Populations wholly or partially occurring within the Rio Grande National Forest are shown in bold.

						Management Category[4]					
Sub-basin	**Drainage**	**Water body**	**Origin[1]**	**Genetic origin of transplant[2]**	**Land ownership[3]**	**CP**	**PINES**	**BiAMs (RT)**	**Allozymes**	**mtDNA**	**Meristics**
Alamosa-Trinchera	Cat	**Cat Creek**	**Historic**		**USFS**	**Core**				**Core**	**A-**
Alamosa-Trinchera	Cat	**Cat Creek North Fork**	**Historic**		**USFS**	**Core**					
Alamosa-Trinchera	Cat	**Cat Creek South Fork**	**Historic**		**USFS**	**Core**					
Alamosa-Trinchera	Culebra	Alamosito Creek	Historic		Private	Core	Core	Core			A
Alamosa-Trinchera	Culebra	Cuates Creek	Historic		Private	Core	Core	Core			A
Alamosa-Trinchera	Culebra	Culebra Creek North Fork	Historic		Private	Core					A-
Alamosa-Trinchera	Culebra	Jaroso Creek	Historic		Private	Core	Core	Core		Core	A
Alamosa-Trinchera	Culebra	Torcido Creek	Historic		Private	Core	Core	Core		Core	A
Alamosa-Trinchera	Culebra	Vallejos #2	Historic		Private	Core	Core	Core			A
Alamosa-Trinchera	Culebra	Vallejos North Fork	Historic		Private	Core					A
Alamosa-Trinchera	Culebra	Willow Creek	Historic		Private	Core	Sport	Sport			
Alamosa-Trinchera	Jim	**Jim Creek**	**Renovation 1976**	**Jaroso, Torcido, WI**	**USFS/ Private/ STLB**	**Core**	**Core**	**Core**			
Alamosa-Trinchera	Jim	**Jim Creek South Fork**	**Renovation 1976**	**Jaroso, Torcido, WI**	**USFS/ STLB**	**Core**					
Alamosa-Trinchera	Jim	**Torsido Creek**	**Renovation 1977**	**Torcido, WI**	**USFS**	**Core**					
Alamosa-Trinchera	Rough Canyon	**Rhodes Gulch**	**Transplant 1980s**	**WI**	**USFS**	**Core**		**Core/Cons**	**Core**		
Alamosa-Trinchera	Rough Canyon	**Rough Canyon**	**Transplant 1980s**	**Osier, Placer, WI**	**USFS**	**Core**		**Core/Cons**	**Core**		

Table 1 (cont.).

						Management Category[4]					
Sub-basin	**Drainage**	**Water body**	**Origin[1]**	**Genetic origin of transplant[2]**	**Land ownership[3]**	**CP**	**PINES**	**BiAMs (RT)**	**Allozymes**	**mtDNA**	**Meristics**
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Grayback Canyon	Historic		Private	Core	Core	Core			
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Little Ute Creek	Transplant 1978	Placer, WI	Private	Core	Core	Core		Core	
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Placer Creek	Historic		Private	Core	Cons		Core	Core	A-/B
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Placer Creek Middle Fork	Historic		Private	Core			Core		
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Placer Creek South Fork	Historic		Private	Core			Core		
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Sangre de Cristo Creek	Historic		Private	Core			Core	Core	A
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Ute Creek	Historic		Private	Core					A
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Ute Lake Lower	Transplant 1978	Placer, WI	Private	Core					
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Ute Lake Upper	Transplant 1978	Placer, WI	Private	Core					
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	Wagon Creek	Historic		Private	Core	Cons		Core		
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	West Indian Creek	Historic		Private	Core	Core	Core	Core	Core	A
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	West Indian Creek North Fork	Historic		Private	Core					

Table 1 (cont.).

						Management Category[4]					
Sub-basin	**Drainage**	**Water body**	**Origin[1]**	**Genetic origin of transplant[2]**	**Land ownership[3]**	**CP**	**PINES**	**BiAMs (RT)**	**Allozymes**	**mtDNA**	**Meristics**
Alamosa-Trinchera	Sangre de Cristo, Ute, West Indian	West Indian Creek South Fork	Historic		Private	Core	Core	Core			
Alamosa-Trinchera	San Francisco	San Francisco Creek	Renovation 1979	Placer, Torcido, WI	Private	Core	Core	Core			
Alamosa-Trinchera	San Francisco	**San Francisco Creek Middle Fork**	**Renovation 1981**	**Torcido, WI**	**USFS/ Private**	**Core**	**Core**	**Core**			
Alamosa-Trinchera	San Francisco	**San Francisco Lake Upper West**	**Renovation 1980**	**Broodstock**	**USFS**	**Core**					
Alamosa-Trinchera	Trinchera	Deep Canyon	Renovation 1977	WI	Private	Core					
Alamosa-Trinchera	Trinchera	Trinchera #2	Renovation 1977	WI	Private	Core					A-
Alamosa-Trinchera	Trinchera	Trinchera Creek North Fork	Renovation 1977	WI	Private	Core					
Alamosa-Trinchera	Trinchera	Trinchera Creek South Fork	Renovation 1977	WI	Private	Core					
Chama	Chama	Nabor Creek	Renovation 1982	WI	Private	Core					
Chama	Chama	Native Lake	Transplant 1987	WI	Private	Core					
Chama	Chama	Rio Chamita	Historic		Private	Cons					B+
Chama	Chama	Sexto Creek	Historic		Private	Cons					B
Chama	Chama	**Wolf Creek**	**Historic**		**USFS/ Private**	**Core**	**Core**				**A/B**
Conejos	Conejos	**Big Lake**	**Renovation 2005**	**Broodstock**	**USFS**	**Core**					
Conejos	Conejos	**Canyon Verde**	**Historic**		**USFS**	**Cons**	**Sport**	**Cons**			**B**
Conejos	Conejos	**Conejos Lake Fork**	**Renovation 1977-2005**	**WI, Broodstock**	**USFS/ Private**	**Core**	**Core**	**Core**			
Conejos	Conejos	**Rock Lake**	**Renovation 1977**	**WI**	**USFS Wilderness**	**Core**					
Conejos	Los Pinos	**Cascade Creek**	**Historic**		**USFS**	**Core**	**Core**	**Core**			**A/A-**
Conejos	Los Pinos	**Osier Creek**	**Historic**		**USFS/ Private**	**Core**	**Core**	**Core**		**Core**	**A**

Table 1 (cont.).

						Management Category[4]					
Sub-basin	**Drainage**	**Water body**	**Origin[1]**	**Genetic origin of transplant[2]**	**Land ownership[3]**	**CP**	**PINES**	**BiAMs (RT)**	**Allozymes**	**mtDNA**	**Meristics**
Conejos	Los Pinos	**Rio de los Pinos #2**	**Transplant 1984**	**WI**	**USFS**	**Core**					
Rio Grande Headwaters	Rio Grande headwaters	**Pass Creek**	**Historic**		**USFS**	**Cons**					**B+**
Rio Grande headwaters	Rio Grande headwaters	**Pass Creek West Fork**	**Stocking into Reservoir**	**Broodstock**	**USFS/ Private**	**Core**	**Core**		**Core**		
Rio Grande headwaters	Rio Grande headwaters	**West Alder Creek**	**Historic**		**USFS**	**Cons**	**Core**	**Core**			**B**
Rio Grande headwaters	Rio Grande headwaters	**West Bellows Creek**	**Historic**		**USFS**	**Cons**	**Sport**	**Cons**		**Sport**	**C**
San Luis	San Luis	Medano Creek	Renovation 1987	Osier, Placer, WI	NPS	Core	Core	Core			
San Luis	San Luis	Medano Lake	Transplant 1987-1994	Broodstock	NPS	Core			Core		
San Luis	San Luis	Little Medano Creek	Renovation 1987	Placer	NPS	Core					
San Luis	San Luis	Medano Hudson Branch	Renovation 1987	Osier, Placer, WI	NPS	Core	Core	Core			
Sanguache	Carnero	Carnero Creek	Historic		BLM/ Private	Cons					
Sanguache	Carnero	**Carnero Creek Middle Fork**	**Historic**		**USFS/ Private**	**Cons**	**Core**		**Core**	**Cons**	**A-**
Sanguache	Carnero	**Carnero Creek North Fork**	**Historic**		**USFS/ BLM/ Private**	**Core**	**Core**		**Core**	**Core**	**A-**
Sanguache	Carnero	**Carnero Creek South Fork**	**Historic**		**USFS/ Private**	**Cons**	**Core**		**Core**	**Cons**	**B/C**
Sanguache	Carnero	**Cave Creek**	**Historic**		**USFS/ Private**	**Core**	**Cons**		**Core**		**A-**
Sanguache	Carnero	**Miners Creek**	**Historic**		**USFS**	**Core**	**Cons**	**Core**		**Core**	**A**
Sanguache	Carnero	**Prong Creek**	**Historic**		**USFS**	**Core**	**Sport**		**Core**		
Sanguache	Jack's	**Cross Creek**	**Historic**		**USFS/ BLM/ Private**	**Core**	**Core**		**Core**		
Sanguache	Jack's	**Jack's Creek**	**Historic**		**USFS/ Private**	**Core**	**Core**		**Core**		
Sanguache	Sanguache	**Big Springs Creek**	**Renovation 2003**	**Broodstock**	**USFS**	**Core**					
Sanguache	Sanguache	**East Pass Creek**	**Historic**		**USFS**	**Core**	**Core**		**Core**	**Core**	**A**

Table 1 (concluded).

Sub-basin	Drainage	Water body	Origin[1]	Genetic origin of transplant[2]	Land ownership[3]	Management Category[4] CP	PINES	BiAMs (RT)	Allozymes	mtDNA	Meristics
Sanguache	Sanguache	**Middle Creek East**	**Renovation 1980s**	**Osier, Placer**	**USFS**	**Core**					
Sanguache	Sanguache	**Tuttle Creek**	**Transplant 1977**	**WI**	**USFS/ BLM/ Private**	**Core**				**Core**	**A**
Sanguache	Sanguache South Fork	**Deep Creek**	**Historic**		**USFS**	**Unknown**					
Sanguache	Sanguache South Fork	**Unknown Creek**	**Transplant 1985**	**Osier**	**USFS Wilderness**	**Core**					
Sanguache	Sanguache South Fork	**Wannamaker Creek**	**Historic**		**USFS/ Private**	**Cons**	**Sport**				**B+**
Sanguache	Sanguache South Fork	**Whale Creek**	**Historic**		**USFS**	**Cons**	**Core**	**Core**			**B-**
Upper Rio Grande	Costilla	Costilla Creek East Fork	Renovation 2002	Broodstock	Private	Core			Core		
Upper Rio Grande	Costilla	Costilla Creek West Fork	Renovation 2002	Broodstock	Private	Core			Core		
Upper Rio Grande	Costilla	Glacier Lake	Renovation 2002	Broodstock	Private	Core			Core		
Canadian	Ricardo	Fish Creek	Historic		Private	Unknown					
Canadian	Ricardo	Little Vermejo Creek	Historic		Private	Unknown	Core	Core			
Canadian	Ricardo	Ricardo Creek	Historic		Private	Core				Core	A

[1]Renovation = a population created in a water body from which fish had previously been removed, Transplant = a population created in a water body that was previously fishless.

[2]WI = West Indian Creek

[3]USFS = USDA Forest Service, STLB = State Trust Land Board, NPS = National Park Service, BLM = Bureau of Land Management

[4]Management categories of populations as noted in the Colorado Division of Wildlife Conservation Plan are shown under CP. In some cases these categories may have since been revised. Subsequent columns show management categories as determined using meristics and/or several different types of genetic markers, abbreviated as follows: PINEs=polymorphic interspersed nuclear elements, BiAMs (RT) = biallelic markers used to assess level of rainbow trout introgression only, mtDNA = mitochondrial DNA. Core = a population considered to have <1 percent non-native introgression, Cons = a population considered to have <10 percent non-native introgression, Sport = a population with >10 percent non-native introgression

extensive beaver activity may also result in the loss of spawning gravels.

Food habits

Cutthroat trout are opportunistic foragers, primarily feeding on invertebrates. Cutthroat trout fry utilize the invertebrate assemblages characteristic of shallow, slow velocity rearing habitats; Moore and Gregory (1988b), for example, found the diet of cutthroat trout fry in a stream in the Cascade Mountains to consist primarily of chironomid midge larvae, mayflies (*Ephemeroptera*), and ostracods. Older stream-dwelling cutthroat trout are primarily drift foragers, waiting in open water for prey items to pass. Diet studies of Rio Grande cutthroat trout and Colorado River cutthroat trout (Bozek et al. 1994, Young et al. 1997, New Mexico Department of Game and Fish unpublished data) have found midge larvae (*Diptera*), caddisflies (*Tricoptera*), and mayflies (*Ephemeroptera*) to be important diet components. Larger prey items appear to be preferentially selected (Wilzbach et al. 1986, Bozek et al. 1994, Hilderbrand and Kershner 2004a). As individuals grow, they tend to utilize a wider size range and variety of food items, and they may exhibit more benthic feeding (Skinner 1985). Piscivory has not been demonstrated in Rio Grande cutthroat trout, but Rinne (1995) speculates that the young of other native fish taxa may be a component of adult diet.

Availability of food for a Rio Grande cutthroat trout population is affected by stream channel morphology, competition with other fish (Griffith 1988, Shemai 2004), condition of the riparian corridor, deposition of fine sediments on the stream bottom, hydrology, and water quality. Alterations in these elements will modify the stream character, altering the total abundance of food items and the composition of the aquatic macroinvertebrate community (Rosenberg and Resh 1993), and they may also potentially affect the foraging efficiency of resident trout (e.g., Wilzbach et al. 1986). At higher elevations where many extant populations of Rio Grande cutthroat trout are found, streams are typically less productive than those at lower elevations, and leaves and dead wood from riparian vegetation are the primary sources of energy for aquatic invertebrates (Sublette et al. 1990). Deciduous plant tissue, which decomposes more rapidly than coniferous plant tissue, may be a particularly important nutrient source (Romero et al. 2005). At certain times of the year a large proportion of the diet may come from terrestrial invertebrates, the availability of which will also depend on the riparian vegetation (Wipfli 1997, Romero et al. 2005). Bozek et al. (1994) suggest that food may be limiting for adult Colorado River cutthroat trout in montane streams and that there may be high competition for food between adults and younger size classes. This may cause immigration of individuals out of such streams in search of better feeding opportunities (see earlier discussion; Hilderbrand and Kershner 2004b).

Breeding biology

In common with other inland cutthroat trout subspecies, male Rio Grande cutthroat trout typically mature sexually at 2 or 3 years whereas females usually mature at 3 years (Irving 1954, Drummond and McKinney 1965, Young 1995a, New Mexico Department of Game and Fish unpublished data). However, time of maturation is expected to vary between individuals and populations, and it may depend more strongly on fish length than chronological age (Meyer et al. 2003). In colder headwater streams, trout tend to mature at a smaller size than they do at lower elevations with higher water temperatures (Behnke and Zarn 1976, Meyer et al. 2003). Based upon examination of scale annuli and length frequency histograms for individuals from five populations in New Mexico, D. Cowley (unpublished data) suggests a lower size limit of approximately 120 mm (4.7 inches) for age 2 and 150 mm (5.9 inches) for age 3 Rio Grande cutthroat trout. When surveying Rio Grande cutthroat trout populations, CDOW considers all individuals >120 mm (4.7 inches) to be adults (Colorado Division of Wildlife 2004). Paroz (2005) considers all individuals >140 mm (5.5 inches) to be adult.

Rio Grande cutthroat trout spawn on the descending limb of the spring snowmelt hydrograph, typically from the middle of May to July. It is not known exactly what factors influence the timing of spawning, but Stumpff (1988) suggests that water temperature may be important. Data from field spawns demonstrate that, as for other salmonids, the number of eggs produced by Rio Grande cutthroat trout depends upon female size (**Figure 5**; D. Cowley unpublished data, New Mexico Department of Game and Fish unpublished data). The eggs are deposited into a gravel nest, or redd, located in areas exposed to flowing water such as stream riffles (Sublette et al. 1990, Young 1995a). Redds have a unique morphology that appears to optimize physical conditions for egg incubation (Chapman 1988). The location of cutthroat trout redds appears to be influenced by fish density, water temperature, flow velocity, water depth, and the availability of suitable spawning substrate (Magee et al. 1996). However, the

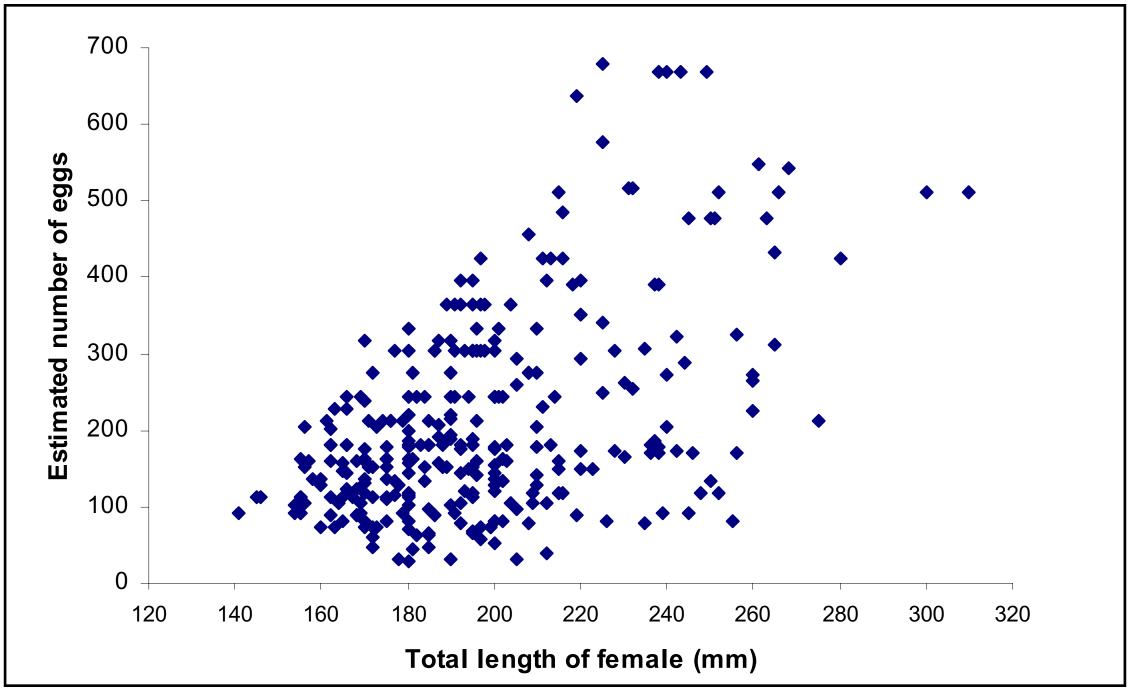

Figure 5. Relationship of egg number to female length in nine wild populations of Rio Grande cutthroat trout in New Mexico.

range of conditions under which they are constructed is fairly broad (Thurow and King 1994). Cutthroat trout have been observed to use spawning substrates ranging from <1 to 110 mm (<0.04 to 4.29 inches) in diameter, but optimum gravel size range appears to be somewhere between 12 and 85 mm (0.5 and 3.3 inches; Thurow and King 1994, Schmetterling 2000, Harig and Fausch 2002). Salmonids remove fine sediment from the substrate during redd construction (Young et al. 1989). The number of adult trout in a population able to spawn successfully in a given year is expected to be limited by the availability of suitable spawning habitat and will depend upon redd size and spacing. Redd size generally varies with fish size (Ottaway et al. 1981), and territorial behavior of spawning fish may maintain additional distance between redds (Thurow and King 1994). Although individual Rio Grande cutthroat trout are believed to spawn in multiple years, post-spawning mortality, as in other cutthroat trout subspecies, may be high (Schmetterling 2001, De Rito 2004, Schrank and Rahel 2004). Observations during Rio Grande cutthroat trout field spawns in New Mexico suggest that most adults within a population are capable of spawning each year (Paroz personal communication 2005). However, notes on a Rio Grande cutthroat trout broodstock maintained by NMDGF between the 1930's and 1970's suggest that adults in this broodstock may have only been capable of spawning once every two years (New Mexico Department of Game and Fish 1951). A similar biennial pattern of spawning has been reported in some populations of westslope cutthroat trout (McIntyre and Rieman 1995).

Cutthroat trout do not exhibit parental care. Depending upon temperature, subspecies, and population, the eggs deposited in the redd hatch within 3 to 7 weeks (Young 1995a, Behnke 2002). The juvenile trout ('alevins') then remain within the gravel of the redd for a further 2 to 3 weeks until the yolk sac is absorbed (Young 1995a, Behnke 2002), after which they emerge to begin actively feeding. Successful embryonic development requires sediment-free gravel beds that have a continuous flow of well-oxygenated water, and accumulation of fine sediments in the redd can significantly reduce hatch rate as a result of reducing oxygenation of the eggs (Irving and Bjornn 1984). Weaver and Fraley (1993), for example, examined the influence of substrate particle size on hatch rate of westslope cutthroat trout eggs. Where particle size was >6.35 mm (0.25 inches), hatch rate averaged 76 percent; as proportion of fine sediments increased, hatch rate declined, averaging only 4 percent in the treatment with the greatest proportion of fines. Fine sediment deposition in a stream is a frequent result of land disturbance such as grazing, logging, and road construction (Magee et al. 1996). Magee et al. (1996) suggest that some salmonid populations may be able adapt to elevated levels of fine sediment; for example, van den Berghe and Gross (1984) report that smaller salmonid eggs exhibit improved survival in highly sedimented substrates. In contrast, Einum et al. (2002) found the opposite to be true for brown trout; survival was higher for larger eggs when dissolved oxygen levels were low.

Virtually nothing is known about mate choice and mating success in Rio Grande cutthroat trout. As has been observed in other salmonid taxa, females may compete for suitable spawning sites while males may compete for access to females (McLean et al. 2005). Salmonid females frequently prefer larger males (e.g., Berejikian et al. 2000), and size assortative mating may occur (Hanson and Smith 1967). Such phenomena are expected to result in skewed reproductive success that can generate an effective population size (N_e; see later discussion) much lower than the census number of reproductive-aged fish in the population. Similarly, N_e will also be affected by sex ratio. Unpublished data collected by New Mexico Department of Game and Fish from several wild Rio Grande cutthroat trout populations suggest a mean male: female sex ratio of approximately 1.28:1 (Patten personal communication 2006), which is similar to mean sex ratios reported for other cutthroat trout subspecies (e.g., Meyer et al. 2003). However, actual sex ratio may vary widely from stream to stream (Young 1995b).

As is the case for other cutthroat trout (e.g., Weigel et al. 2003), Rio Grande cutthroat trout spawn in the same habitat and at the same time of year as introduced rainbow trout and non-native cutthroat trout. In general, there appear to be no behavioural or physical barriers to hybridization between inland cutthroat trout and introduced rainbow trout. In some cases, however, temporal separation of spawning may limit gene exchange between the two species (fluvial Yellowstone cutthroat trout; Henderson et al. 2000, De Rito 2004). Hybrid offspring are fertile. Allendorf et al. (2004) present some evidence suggesting that the hybrid offspring of matings between westslope cutthroat trout and rainbow trout may have a fitness disadvantage compared to pure westslope cutthroat. However, there is no evidence that such a fitness disadvantage is limiting the spread of non-native introgression into cutthroat trout populations. Rubidge and Taylor (2005) and Hitt et al. (2003), for example, demonstrate rapid spread of rainbow trout hybridization through westslope cutthroat trout populations, perhaps due to hybrid vigor (see later discussion) or because individuals containing genetic material from rainbow trout exhibit increased movement rates (Ellstrand and Schierenbeck 2000, Allendorf et al. 2004). Ongoing hybridization will cause a cutthroat trout population to be replaced firstly by a hybrid swarm containing genetic material from native and non-native trout and ultimately by a population phenotypically corresponding to the non-native species (e.g., Hitt et al. 2003). Incorporation of even small amounts of non-native genetic material into a cutthroat trout population may have long-term fitness consequences, even if there is a short-term fitness advantage (see later discussion; Allendorf et al. 2004). Currently hybridization with non-natives is one of the primary threats to the continued survival of the Rio Grande cutthroat trout.

Demography

As previously detailed, extant pure Rio Grande cutthroat populations are confined to relatively short reaches of headwater stream, or in some cases small, high-elevation lakes. They are frequently protected from the influence of non-native trout by migration barriers, which also prevent movement of cutthroat trout between populations. This contrasts with the presumed historical situation, in which cutthroat trout are expected to have existed in spatially connected and numerically large populations. Opportunities for population growth and dispersal of adults and young are highly limited in the current situation. Such small, isolated populations have an elevated extinction risk as a result of demographic and genetic stochasticity.

Genetic characteristics and concerns

Several genetic phenomena occur as a result of population fragmentation and need to be considered when assessing management alternatives for Rio Grande cutthroat trout. In addition, genetic processes, such as local adaptation, which are expected to occur in populations in their natural state, may require consideration. Ryman and Utter (1987) and Hallerman (2003) discuss the application of population genetics theory to fisheries management in more detail.

Random genetic drift, effective population size and inbreeding depression: In any closed population, such as a Rio Grande cutthroat trout population isolated above a migration barrier, genetic variation will be lost over time as a result of random genetic drift. The rate of loss will be greater the smaller a population's 'effective size' (N_e; Wright 1931). In an 'idealized population' with equal sex ratio, equal probability of reproductive success for each adult, random mating, non-overlapping generations and constant population size, N_e equals the size of the adult population, N_{adult} (Hallerman 2003). Most real populations, including Rio Grande cutthroat trout, do not conform to this ideal, and N_e is commonly substantially smaller than N_{adult} (Frankham 1995). For example, Palm et al. (2003) and Jensen et al. (2005) estimate N_e/N_{adult} ratios for stream dwelling brown trout of <0.2 to <0.5 and 0.22 to 0.24 respectively. Several approaches are available to estimate N_e of a population (e.g., Cabellero 1999, Luikart and Cornuet 1999), but no such study has yet been completed for any inland cutthroat trout subspecies. In the absence of such data, we suggest that half the number of adult cutthroat trout present should be considered the maximum N_e for that population, with true N_e probably being much lower (Young and Harig 2001).

Recently isolated populations of Rio Grande cutthroat trout with low N_e face at least two genetic threats, which may increase their vulnerability to extinction (Frankham 2005). Firstly, loss of genetic variability reduces the capacity of a population to adapt to environmental changes. Secondly, an effect known as 'inbreeding depression' can occur, whereby fitness of a population declines as a result of increased homozygosity of individuals (Keller and Waller 2002). Inbreeding depression appears to be primarily due to recessive deleterious alleles being expressed in the homozygous state. While loss of genetic diversity typically has cumulative impacts over the long term, inbreeding depression can very quickly increase the extinction risk of a population (Frankham 2005). It is frequently proposed, as a rule-of-thumb, that a minimum N_e of 500 is required in order for a population to maintain its historical level of genetic variation, with a minimum N_e of 50 being required in order for a population to avoid inbreeding depression in the short term (commonly known as the "50/500 rule"; Franklin 1980, Frankel and Soulé 1981). However, there is debate over these figures. Lande (1995), for example, suggests that a minimum N_e of 5000 may be required in order for a population to maintain adaptive potential in the long term. Conversely, it has been hypothesized that populations that undergo gradual inbreeding for a period of time may 'purge' their deleterious recessive alleles and hence may become less susceptible to the negative effects of inbreeding (Crnokrak and Barrett 2002, Keller and Waller 2002). A number of authors argue that demographic processes are likely to drive small populations to extinction before genetic processes have an important effect (e.g., Lande 1988).

Introducing new genetic material to a population suffering from inbreeding depression can cause a rapid short-term rise in population fitness, an effect known as 'outbreeding enhancement', 'hybrid vigor', or 'genetic rescue' (Tallmon et al. 2004).

Subpopulation differentiation and outbreeding depression: Where little or no gene flow occurs between populations, for example where trout are isolated in different river systems, genetic divergence will occur as a result of selection within the local environment and/or random genetic drift. Selection will favor alleles that confer a fitness advantage in the physical environment within which an individual occurs, and combinations of alleles at different loci that function better together within that physical environment. Over time, this is expected to give rise to locally co-adapted complexes of genes (Dobzhansky 1937, Wallace 1991). When populations that have diverged as a result of selection (this is known as 'adaptive divergence') interbreed, the hybrid descendents are expected to be less fit than their parents in their parents' native environment. This phenomenon, known as 'outbreeding depression', occurs both because alleles adaptive in the native environment are replaced by alleles not adaptive to that environment and because co-adapted gene complexes are disrupted. This latter process means that outbreeding depression may occur even where two genetically isolated populations are adapted to identical environments (e.g., Gharrett and Smoker 1991). Outbreeding depression may be masked in the first few hybrid generations by the effect of outbreeding enhancement, a phenomenon that may allow the spread of hybridization through a population of cutthroat trout even though this hybridization may ultimately have a fitness cost (Allendorf et al. 2004).

Unfortunately, current scientific knowledge is unable to predict with accuracy when hybridization between two populations is likely to result in outbreeding depression. Generally, outbreeding depression is considered more likely when populations are more geographically isolated from one another, are more genetically divergent, or appear to be adapted to different environmental conditions (Hallerman 2003).

Treatment of introgressed populations: Incorporation of even small amounts of genetic material from non-native taxa may cause outbreeding depression, and hence reduced fitness and increased risk of extinction, in populations of native trout. For this reason, most conservation plans focus on identifying and protecting pure cutthroat trout, even though individuals containing low levels (e.g., <20 percent) of non-native introgression may be morphologically and behaviorally indistinguishable from pure fish (U.S. Fish and Wildlife Service 2003) and exhibit no apparent reduction in fitness. Numerous populations of Rio Grande cutthroat trout are known to contain genetic material from non-native *Oncorhynchus* taxa (New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004); in many cases this genetic material may have been incorporated many generations ago and the populations continue to persist, with no ill-effects being documented. Nevertheless, outbreeding depression remains a theoretical risk if these introgressed populations are allowed to interbreed with pure populations (Allendorf et al. 2004).

Despite this concern, populations containing low levels of non-native introgression can still represent an important resource in the conservation of threatened taxa. They may, for example, contain native genetic diversity that is not represented in extant pure populations (Peacock and Kirchoff 2004), or exhibit unique ecological characteristics or life-history strategies. Generally, the fewer the number of pure populations remaining, the greater the conservation value of such hybridized populations (Allendorf et al. 2004). The Utah Position Paper (Utah Division of Wildlife Resources 2000) places cutthroat trout populations into three different management categories, primarily according to levels of introgression calculated using diagnostic genetic markers.

'Core Conservation Populations' are self-sustaining cutthroat trout populations that exhibit <1 percent introgression from non-native trout. These populations are considered to contain primarily pure native cutthroat trout; a boundary level of '<1% introgression' is necessary because complete absence of introgression cannot statistically be proven without sampling the entire population. Core Conservation Populations have the highest conservation priority and are the primary source of gametes and individuals for transplants and broodstock development.

'Conservation Populations' are self-sustaining populations that correspond phenotypically to pure native cutthroat trout but exhibit low levels of introgression from non-native taxa. While most agencies currently include populations with the arbitrary value of <10 percent non-native introgression within this category, populations with higher levels of introgression may also be classed as 'Conservation Populations' if they exhibit phenotypic, ecological, behavioral, or genetic characteristics deemed worthy of protection (Utah Division of Wildlife Resources 2000). 'Conservation Populations' also have high conservation priority, but they are not utilized as sources of gametes for broodstock and in cases may be targeted for management actions intended to convert them to 'Core Conservation Population' status.

'Sportfish populations' ('recreation populations', Colorado Division of Wildlife 2004; 'primary restoration populations', New Mexico Department of Game and Fish 2002) are populations of trout that either exhibit greater levels of introgression than are acceptable in 'Conservation Populations' of cutthroat trout or are not-self sustaining, for example populations created by stocking in high-elevation lakes for recreation purposes. 'Sportfish Populations' are generally subject to the same management as non-native trout populations.

Outbreeding depression and population supplementation: Outbreeding depression is also a potential risk when cutthroat trout are moved between genetically divergent populations, for example as part of a supplementation program. For this reason the Utah Position Paper recommends that fish are not introduced into Conservation or Core Conservation Populations unless deemed absolutely necessary, for example to rescue a population from documented inbreeding depression (Utah Division of Wildlife 2000). It is unknown whether any Rio Grande cutthroat trout population has had the opportunity to undergo adaptive divergence to the point where outbreeding depression may be a significant problem. Observation of movement rates in other cutthroat trout subspecies suggests that many populations within the Rio Grande drainage may have exchanged genes prior to anthropogenic population fragmentation. In contrast, Rio Grande cutthroat trout in the Rio Grande, Pecos, and Canadian drainages and in the San Luis closed basin may have been naturally isolated from one another for thousands of years. Large numbers of hatchery-reared Rio Grande cutthroat trout, however, were stocked between the three river drainages in the mid 20th century (New Mexico Department of Game and Fish unpublished data). This anthropogenic migration may have swamped out any adaptive divergence that was previously present between these drainages.

Genetic characteristics of Rio Grande cutthroat trout populations: A recent study has investigated the population genetics of Rio Grande cutthroat trout in New Mexico using highly variable nuclear genetic markers ('microsatellites'; Pritchard and Cowley 2005, Pritchard et al. submitted). This study did not include Rio Grande cutthroat trout populations from Colorado, but the patterns observed are expected to be true for the subspecies over its entire range. Results show that populations vary in the amount of genetic diversity that they contain. As expected, those occurring above natural migration barriers tend to be less diverse than those not isolated by such barriers. Genetic diversity shows no significant relationship to habitat or population size; two of the largest Rio Grande cutthroat trout populations in New Mexico (Canones Creek and Polvadera Creek) are also the least genetically diverse. Stumpff (1998) previously noted very low trout densities in these two populations, apparently due to habitat degradation, and the observed low genetic diversity may therefore reflect recent population bottlenecks. Several other Rio Grande cutthroat trout populations in New Mexico exhibit genetic evidence for recent bottlenecks (Pritchard and Cowley 2005, Pritchard et al. submitted).

Individual Rio Grande cutthroat trout populations, even those geographically adjacent, tend to be highly genetically differentiated from one another (global F_{st} = 0.4; , Pritchard and Cowley 2005, Pritchard et al. submitted). Similar levels of genetic differentiation have been observed in other stream-dwelling salmonids, particularly those fragmented by natural or artificial migration barriers (e.g., Carlsson and Nilsson 2001, Young et al. 2004). This level of genetic differentiation suggests that migration between Rio Grande cutthroat trout populations in their natural state may have been rather limited, and hence there may have been opportunity for adaptive divergence between these populations. However, the observed distribution of genetic variation within the Rio Grande drainage supports a model of at least some gene flow, rather than one of complete population isolation, and some populations may have exchanged migrants relatively recently. The recent effects of population fragmentation and associated population bottlenecks have probably contributed to the high levels of genetic differentiation observed (Hedrick 1999). As each isolated population only contains a small proportion of the total genetic diversity remaining within Rio Grande cutthroat trout as a whole, preservation of as many historic populations as possible, and maintenance of these populations at a sufficient size to minimize further loss of allelic diversity due to drift, are necessary if a management agency wishes to minimize further loss of genetic variation from the subspecies. Populations within the Pecos and the Rio Grande drainages are genetically more similar to populations within their own drainage than populations in the alternative drainage; hence fish stocking has not completely obscured the expected genetic divergence between these two drainages. Genetic analysis of the few populations remaining in the Canadian drainage has shown that some appear to be particularly genetically distinct compared to Rio Grande cutthroat trout populations in the Rio Grande and Pecos (Riddle and Yates 1990, Keeler-Foster 2003, Douglas and Douglas 2005, Pritchard and Cowley 2005). Genetic evidence does not support the hypothesis that Rio Grande cutthroat trout recently entered the Canadian system via stocking.

Life history characteristics

Figure 6 shows a life cycle diagram for Rio Grande cutthroat trout, based on data collected from populations of the subspecies in New Mexico (D. Cowley unpublished data, New Mexico Department of Game and Fish unpublished data). Age classes were determined by examining the scale annuli, and length frequency histograms in five populations and survival probabilities between age classes 1 and above were estimated from number of fish observed in each class. Age 0 is equivalent to young-of-the-year (YOY). Maximum estimated fish age was 8 years. Data were unavailable to estimate egg hatching rate and YOY-age 1 survival for Rio Grande cutthroat trout; therefore, values shown in the diagram are taken from studies of other subspecies (Magee et al. 1996, Knight et al. 1999). Number of eggs produced by each age class is estimated using fish length and fecundity data collected during Rio Grande cutthroat trout field spawns (**Figure 5**). These data show that number of eggs per female is significantly related to length and hence to age class. Based on observations of hatchery-reared Rio Grande cutthroat trout, and data from other cutthroat trout subspecies, a small number of individuals are expected to become sexually mature at the age of 2 (length >120 mm [4.7 inches]); however, most do not reach sexual maturity until their third year (length >150 mm [5.9 inches]) (Harig and Young 2001, New Mexico Department of Game and Fish unpublished data).

In many cases, immigration of fish into Rio Grande cutthroat trout populations is currently precluded by the presence of migration barriers. Emigration from these populations as a result of movement of trout downstream over these barriers, however, may be substantial. Such emigration from the population may increase in response to management activities such as electrofishing (Nordwall 1999, but see Young and Schmetterling 2004), competition for feeding territories (Nakano et al. 1992, Hilderbrand and Kershner 2004b), or adverse environmental conditions such as the formation of anchor ice (Jakober et al. 1998). Once a fish passes over the barrier, it is permanently lost from the population. Hence, if level of vagility has a genetic basis, there is expected to be strong selective

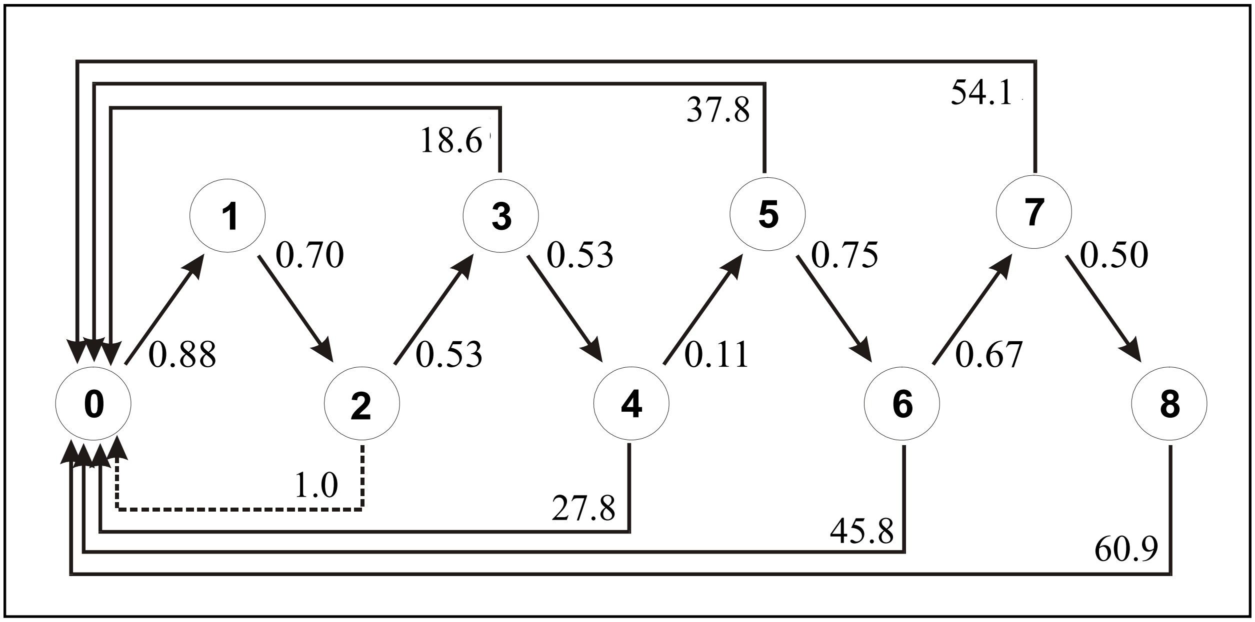

Figure 6. A lifecycle graph for Rio Grande cutthroat trout. Eight age classes are denoted by circles. Age 0 is equivalent to young-of-the-year. Arrows connecting age classes represent transitions from one age to the next, and the numbers adjacent to these arrows denote survival probabilities. Numbers next to arrows leading from an age class back to age 0 denote reproductive outputs (number of eggs*estimated egg hatching rate of 0.1). The dashed arrow from age 2 to age 0 indicates that a small percentage (about 9 percent) of age 2 females produce eggs.

pressure within these isolated Rio Grande cutthroat trout populations for a sedentary life-history strategy (Northcote 1992).

Population viability analyses: Several recent modeling studies have estimated the population size and amount of habitat required to support an isolated cutthroat trout population with sufficiently small chance of going extinct as a result of demographic processes and/or having a long-term N_e of <500. Such analyses serve to guide managers as to the population conditions they might strive to achieve, and help them to identify those populations that are most vulnerable and therefore might be prioritized for management activity. They cannot, however, be used as tools to predict the ultimate fate of a population, nor can they be used to provide exact numeric thresholds above or below which a population can be considered 'secure' or 'insecure' (Hilderbrand 2003).

McIntyre and Rieman (1995) and Young and Harig (2001) used a method described by Dennis et al. (1991) to estimate extinction risk for westslope cutthroat trout and greenback cutthroat trout, based on time-series analysis of population survey data. Both studies concluded that the risk of extinction as a result of stochastic population processes increased rapidly when the size of a population dropped below 2000 individuals.

Hilderbrand and Kershner (2000a) used data for Bonneville, Colorado River, and westslope cutthroat trout to calculate the minimum length of stream (<7 m [23 ft.] wide) required to support a cutthroat trout population with N_e >500, assumed in this case to correspond to population with >2500 individuals >75 mm (3 inches) in length (following Allendorf et al. 1997). They based their calculations on census population size, the fraction of the population assumed to remain after mortality or emigration and the density of fish per unit stream length. Using this approach and assuming a linear relationship of population size to stream length, they estimated that managing for a target effective population size of Ne = 500 requires at least 8 km (5 mi.) of stream at high fish abundance (0.3 fish per m) and 25 km (15.5 mi.) of stream at low abundance (0.1 fish per m). Young et al. (2005), using data from surveys of 31 Colorado River and greenback cutthroat trout populations in high-elevation streams, found that number of fish >75 mm (3 inches) did not increase linearly with stream length but instead could be related through the function: $(\text{population size})^{1/2} = 0.00508$ (stream length (m)) + 5.148. Based on this relationship, they also found that managing for the target N_e = 500 would require at least 8 km (5 mi.) of stream.

Hilderbrand (2003) used a stage-based matrix projection model, based on data collected from westslope and Colorado River cutthroat trout, to examine the effect of carrying capacity, alterations in vital rates, and immigration on the probability of population extinction over a 100-year time period. Carrying capacity in this model was defined as the maximum number of post-YOY individuals that the habitat could support. The model incorporated density dependence and a function that generated random fluctuations in survivorship in order to mimic the effect of environmental stochasticity. Results showed that, at low population sizes, relatively small changes in carrying capacity had a strong influence on probability of population persistence. For example, with a level of stochasticity that resulted in year-class failure in approximately one year out of 20, a population with a carrying capacity of 500 individuals had a 30 percent chance of extinction while one with a carrying capacity of 1000 had just over 10 percent risk of extinction. Where carrying capacity was 2000 or above, the chance of extinction within 100 years was <5 percent. Changing inter-annual survival rate had a strong effect on probability of population persistence. For example, with a 5 percent reduction in survival probabilities, even populations with a carrying capacity of 16000 individuals had <95 percent chance of persistence. Conversely, with a 5 percent increase in survival probabilities, even populations with 500 individuals had >95 percent chance of persistence. Changing levels of 'environmental stochasticity' also changed the likelihood of population persistence. In addition, probability of population extinction was decreased when a population was able to receive immigrants from another population. For example, a population with a carrying capacity of 1000 exhibited a three-fold decrease in extinction risk when it received an average of four subadult or adult immigrants per year.

Cowley (unpublished) developed a similar age-based matrix projection model, based on data collected for Rio Grande cutthroat trout populations in New Mexico (**Figure 5**, **Figure 6**), with data included from other cutthroat trout subspecies where necessary. This model examined both the probability of a population going extinct within 100 years and the probability of a population having a long-term N_e of >500. Carrying capacity was defined as the total number of individuals supported by the habitat, and N_e at any time point was assumed to be equal to N_{adult}. The model incorporated density dependence, demographic stochasticity in survival rates and reproductive outputs, and it included a chance effect, mimicking environmental stochasticity, causing total reproductive failure in a year. No immigration or emigration was assumed. Similar to Hilderbrand (2003), results demonstrated that population carrying capacity, changes in interannual survivorship, and the number of years in which complete reproductive failure occurred all had a strong effect on extinction probability. For example, with a YOY- year 1 survivorship rate of 0.9 and total reproductive failure in 2 years out of 10, a population with a carrying capacity of 250 had <10 percent chance of going extinct over 100 years. However, with complete reproductive failure in 4 or more years out of 10, even populations with the maximum carrying capacity of 25,000 had a >10 percent chance of extinction. Frequency of year-class failure also had a strong effect on probability of a population falling below the target N_e of 500. At the same high survivorship rate of 0.9 for YOY along with no reproductive failure, a carrying capacity of 2750 was needed in order for a long term N_e >500 to occur in over 90 percent of simulations; with complete reproductive failure 2 years in 10, a carrying capacity of 16,000 was needed, and with reproductive failure in 3 years out of 10, a carrying capacity >115,000 was required. Based upon the outcome of this model and data for Rio Grande cutthroat trout density in 37 streams in New Mexico, Cowley (unpublished) estimated minimum habitat sizes required to support a population of Rio Grande cutthroat trout with >90 percent chance of persisting for 100 years and of having a long term N_e >500. With a median fish density of 1,250 fish per ha, age 0 to age 1 survivorship of 0.9, and successful reproduction every year, minimum habitat size required to support such a population is 2.2 ha, which equates to approximately 7 to 22 km (4.4 to 13.8 mi.) of headwater stream in Colorado or New Mexico (Harig and Fausch 1996, Cowley unpublished data). Minimum habitat required increases with decreasing fish density, decreasing age 0 to age 1 survivorship, and increasing number of years where complete reproductive failure occurs. This minimum estimated habitat size assumes that sufficient habitat types are present to support Rio Grande cutthroat trout at all life stages.

Both Hilderbrand (2003) and Cowley (unpublished) noted that the outcome of their models was strongly influenced by the survivorship of early life stages (YOY-age 1 survival and age 1- age 2 survival). The lower the survivorship, the larger the population carrying capacity required to minimize the chance of population extinction and/or long-term N_e<500. It is notable that the negative impact of brook trout on cutthroat trout primarily occurs at the YOY stage. Removing non-natives, as is already practiced, and targeting habitat improvements to enhance YOY survival (Hilderbrand 2003) may therefore be ways of reducing the extinction risk of Rio Grande cutthroat trout populations. Such an outcome, however, also illustrates the limitations of such models: enumeration of YOY and therefore collection of data on YOY survival in Rio Grande cutthroat trout is difficult, making it hard for a management agency to know which model scenario best suits the true situation.

Implications of population viability analyses: Taken as a whole, the results of these different modeling efforts, based on data from different stream-dwelling cutthroat trout subspecies, suggest that managing for populations of Rio Grande cutthroat trout with a carrying capacity of several thousand individuals will help to minimize the chance of population extinction as a result of demographic processes and to minimize the genetic problems associated with small populations. If such population numbers cannot be achieved, however, even modest improvements in carrying capacity and/or YOY survivorship may improve a population's security. Since cutthroat trout abundance appears to increase as a function of the square of habitat length, even a relatively small downstream extension of habitat may cause a substantial expansion in population size (Young et al. 2001). Populations will be more vulnerable to extinction when variance in survivorship is high, for example in habitats characterized by extreme and fluctuating environmental conditions, when YOY survivorship is low, and when complete year-class failures occur.

Hilderbrand's (2003) model shows that the chance of population extinction may also be reduced where a population is able to receive immigrants from another population. This can be achieved, for example, by extending available habitat downstream to include the confluence of two or more tributaries containing Rio Grande cutthroat trout populations. The more tributaries that are connected, the smaller the likelihood of correlation between the population dynamics in each population and therefore the more likely immigration is to reduce the risk of population extinction (Hilderbrand 2003). Connecting multiple tributaries containing Rio Grande cutthroat trout populations is also expected to enable expression of mobile life-history strategies, reduce the risk of inbreeding depression within populations, and allow natural recolonization following local extinctions. Re-connection of isolated Rio Grande cutthroat trout populations to form larger 'metapopulations' is a stated management goal of both Colorado Division of Wildlife (2004) and New Mexico Department of Game and Fish (2002), and several such 'metapopulations' of Rio Grande cutthroat trout already exist in Colorado. However, in some cases, connection of isolated populations may be difficult to accomplish due to factors such as the presence of natural migration barriers, water rights issues, and difficulties in removing non-native trout. In these situations, simulation of migration by artificial translocation of trout is a potential management alternative (see later discussion).

Many isolated Rio Grande cutthroat trout populations (New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004) currently exhibit population sizes smaller than those recommended by Hilderbrand (2003) and occupy shorter stream lengths than those recommended by Hilderbrand and Kershner (2000). However, all of these populations remain important elements in the conservation of the subspecies. For some of these populations, there may be opportunities to increase carrying capacity and chance of long-term persistence by improving habitat quality, expanding available habitat downstream, and linking them with other isolated populations. The population viability analyses discussed here do not address the probability of population extinction as a result of factors such as disease, invasion by non-native trout, and catastrophic environmental events such as wildfires. However, larger, more genetically diverse populations inhabiting more complex habitats are expected to be more robust to such threats, and increasing connectivity between populations may allow fish to escape to refugia and habitat to be re-colonized naturally following a local extinction event.

Social pattern for spacing: Stream-living salmonids are known to defend feeding territories, and territorial behavior may limit population density (Grant et al. 1998). Trout occupying the same location frequently exhibit a dominance hierarchy, with larger individuals tending to be competitively dominant and hence monopolizing preferred feeding stations (e.g., Sabo and Pauley 1997). Territory size, and therefore population density, may be influenced by a number of factors (Grant et al. 1998), including body size (Grant and Kramer 1990), habitat complexity (Chapman 1966), and food density (Slaney and Northcote 1974). Defense of feeding and breeding territories by Rio Grande cutthroat trout will influence the carrying capacity of a stream and the reproductive success of individuals.

Community ecology

The Rio Grande cutthroat trout was a member of a historical fish assemblage that included dace, chubs, and suckers (Hatch et al. 1998). Little is presently known about the physical and environmental factors that structure montane fish communities. Shemai (2004) found evidence of competition between hatchery-reared Rio Grande cutthroat trout and Rio Grande sucker. This finding suggests that efforts to manage a self-sustaining native fish community may require more habitat that managing solely for Rio Grande cutthroat trout, but more research is required.

Predation

Terrestrial predators that may utilize Rio Grande cutthroat trout in their present range include black bear (*Ursus americanus*), raccoon (*Procyon lotor*), and garter snakes (*Thamnophis* spp.) (Rinne 1995). Historically, mink (*Mustela vision*) and river otters (*Lutra canadensis*) were probably major predators of Rio Grande cutthroat trout, but these species are largely or completely extirpated in northern New Mexico and southern Colorado. Piscivorous birds, in particular osprey (*Pandion haliaetus*), great blue heron (*Ardea herodias*), and belted kingfisher (*Ceryle alcyon*), are other potential predators of Rio Grande cutthroat trout. However, they rarely occur in the high elevation habitats to which the subspecies is now restricted. Young (1996) notes dipper (*Cinclus mexicanus*) predation on YOY Colorado River cutthroat trout. Terrestrial predators do not currently appear to be a significant threat to Rio Grande cutthroat trout populations. In their early life stages, Rio Grande cutthroat trout may suffer predation from aquatic macroinvertebrates and larger fish, including conspecifics. Predation on YOY by brown trout and brook trout has been suggested as a factor mediating displacement of cutthroat trout by these non-native species, but tests of this hypothesis have produced mixed results (Dunham et al. 2002). Novinger (2000) and Irving (1987), for example, observed predation on age 0 cutthroat trout by larger age 0 brook trout while Dunham et al. (2000) found no evidence for such predation.

Competition

Competitive exclusion has been suggested as another mechanism by which non-native trout might displace cutthroat trout (Dunham et al. 2002). Juvenile brook trout have been shown to reduce the feeding efficiency, growth, and survival of juvenile cutthroat trout in stream enclosure experiments (Thomas 1996, Novinger 2000). The outcome of competition may also partially be mediated by water temperature. Brook trout appear to be more physiologically tolerant to warmer water conditions (>20 °C [68 °F]) than cutthroat trout (De Staso and Rahel 1994, Novinger 2000). Alves (2003, 2004) notes several cases where Rio Grande cutthroat trout appear to have been extirpated from streams as a result of drought impacts while brook trout are still present, an observation that may at least partly be explained by brook trout being able to tolerate higher temperatures in remnant pools. De Staso and Rahel (1994) showed that brook trout and Colorado River cutthroat trout were equivalent competitors at 10 °C (50 °F), but that brook trout appeared to have a competitive advantage at 20 °C (68 °F). Correspondingly, brook trout may be more likely to invade where anthropogenic impacts on streams result in elevated water temperatures (Shepard 2004). The competitive advantage of brook trout appears to be primarily due to behavioral dominance, enabling them to exclude cutthroat trout from resources. This may be at least partly mediated by the size advantage that brook trout enjoy at the early life stages as a result of hatching in the autumn rather than in the spring. Brown trout also appear to be competitively dominant to cutthroat trout (Wang and White 1994, Shemai 2004). Shemai (2004) and McHugh and Budy (2005) found adult brown trout to have a significant negative impact on the condition of coexisting adult cutthroat trout in enclosure experiments. In the case of Rio Grande cutthroat trout, this impact appears to be due to brown trout excluding cutthroat trout from food resources (Shemai 2004). Paroz (2005), however, found no significant correlation between the body condition of Rio Grande cutthroat trout and the presence of brown trout in streams in New Mexico. In contrast to results from brook trout, McHugh and Budy (2005) found that temperature does not appear to mediate the outcome of competitive interactions between adult brown trout and Bonneville cutthroat trout.

Disease

Rio Grande cutthroat trout are susceptible to common salmonid diseases and parasites. As is true for all taxa, they are particularly vulnerable to pathogenic organisms introduced from outside their native range. This includes whirling disease, which is caused by the mycosporean *Myxobolus cerebralis* (Markiw 1992). Whirling disease was imported from Europe to North American in the 1950's and is now present in hatcheries and trout waters in Colorado and New Mexico. *Myxobolus cerebralis* has a two-stage life cycle with two obligate hosts: salmonid fish and aquatic oligochaetes of the genus *Tubifex*. The free swimming triactinomyxon stage of the parasite is released by *Tubifex* and infects fish via ingestion or attachment to the skin. The parasite then consumes fish cartilage, causing skeletal deformity and abnormal swimming behavior. Young fish, whose skeleton is primarily cartilage, are most severely affected. Spores released from infected fish are taken up again by *Tubifex*. Whirling disease can cause very high mortality in both native cutthroat trout and introduced rainbow trout and brook trout; however brown trout are significantly resistant to the disease

(Nehring 2006). Thompson et al. (1999) and DuBey (2006) found that YOY Rio Grande cutthroat trout infected with whirling disease suffered greater mortality than did similarly infected rainbow trout or brook trout. However, other studies have suggested that cutthroat trout in general are less susceptible to the effects of whirling disease than rainbow trout (Hedrick et al. 1999, Sipher and Bergersen 2005). Different subspecies of cutthroat trout appear to differ in their vulnerability to whirling disease, and there may also be geographical variation in susceptibility within subspecies (Wagner et al. 2001). Whirling disease has been implicated in the rapid decline of several rainbow trout populations within the Rio Grande drainage. Spores of *M. cerebralis* may persist in sediments for several decades, meaning that previously infected waters are poor candidates for Rio Grande cutthroat trout restorations.

Other exotic salmonid diseases that are known to infect cutthroat trout include bacterial kidney disease (*Renibacterium salmoninarum*), bacterial coldwater disease (*Flavobacterium psychrophilum*), and furunculosis (*Aeromonas salmonicida*).

Conservation

Threats

Non-native trout

Currently, the primary threat to the long-term persistence of Rio Grande cutthroat trout is the presence of non-native trout. Vast numbers of brook trout, brown trout, rainbow trout, Yellowstone cutthroat trout, and its fine-spotted Snake River form have been introduced into Colorado and New Mexico over the past 150 years. These large-scale introductions continue today, frequently immediately downstream from extant Rio Grande cutthroat trout populations (New Mexico Department of Game and Fish unpublished data). As a result, non-native trout now occur in self-sustaining or artificially sustained populations in the majority of waters that historically supported Rio Grande cutthroat trout. As previously discussed, Rio Grande cutthroat trout hybridize freely with rainbow trout and non-native cutthroat trout to produce fertile offspring. If left unchecked, this process leads to the irreversible loss of a Rio Grande cutthroat trout population. In contrast, brook trout and brown trout are fall spawners and therefore do not interbreed with Rio Grande cutthroat trout. However, brook trout invasion of streams is frequently associated with the decline of native cutthroat trout (Dunham et al. 2002). The processes involved are not completely understood (Peterson and Fausch 2003) but primarily seem to involve impacts by brook trout on cutthroat trout in the early life stages (age 0 and 1; Peterson et al. 2004a). As previously noted, competition for food and spring predation of brook trout fry on the smaller cutthroat trout fry may be important (Dunham et al. 2002). Some cutthroat trout populations may be able to co-exist with brook trout (Dunham et al. 2002), and in some cases it may be unclear whether the replacement of cutthroat trout by brook trout is due to a direct interaction between the two species or to an independent variable, such as anthropogenic habitat disturbance (Dunham et al. 2002). Spatial segregation is frequently observed between brook trout and cutthroat trout in streams, suggesting that factors such as low water temperatures or high stream gradients may limit brook trout invasion of some cutthroat populations (Dunham et al. 2002). The presence of brown trout is also associated with a decline in cutthroat trout populations, but this is less well documented in the scientific literature. Brown trout are more common than brook trout within Rio Grande cutthroat trout populations in New Mexico (New Mexico Department of Game and Fish 2002). Quist and Hubert (2005) found a negative relationship between the density of brown trout and/or brook trout and the density of cutthroat trout in the Salt River watershed of Wyoming. Similarly, Calamusso and Rinne (2004) and Paroz (2005) found significantly reduced densities of Rio Grande cutthroat trout in populations where they co-existed with brook trout or brown trout compared to populations where they did not. Paroz (2005) found an inverse relationship between the number of age 0 Rio Grande cutthroat trout and the number of brown trout in a population.

The overwhelming threat from non-native trout means that pure Rio Grande cutthroat trout populations require protection by natural or artificial migration barriers. Although construction, monitoring, and maintenance of such barriers is a stated management priority for all relevant agencies, many populations remain unprotected and barrier failure is a frequent occurrence (Alves 1996 - 2004). Policies are in also in place to prevent stocking of non-native trout into Rio Grande cutthroat trout populations on both public and private land. Nevertheless, all populations remain at risk from movement of non-native trout, either upstream over the migration barrier or from adjacent water bodies. There are recently documented cases, for example, where pure Rio Grande cutthroat trout have been replaced by rainbow-cutthroat hybrids as a result of private rainbow trout stocking in adjacent waters (e.g., Stumpff 1998). Illegal stocking of non-native trout into Rio Grande cutthroat trout populations and movement of fish by anglers also remain threats. Alves (2003) notes a recent case where a Rio Grande cutthroat trout population on private land (Willow Creek) was stocked with Yellowstone cutthroat trout. Colorado Division of Wildlife (2004) documents the presence of brook trout or brown trout in over half of 76 surveyed Rio Grande cutthroat trout populations in Colorado; additionally rainbow trout or non-native cutthroat subspecies were found within four of these populations. In most cases the presence of brook or brown trout in these Rio Grande cutthroat trout streams in Colorado appears to be associated with a decline in the native taxon.

Population fragmentation

Although protection of Rio Grande cutthroat trout populations by a migration barrier is an essential short-term conservation strategy, the resulting population isolation generates an alternative set of conservation concerns. These concerns are exacerbated where the habitat available upstream of the migration barrier is capable of supporting only low numbers of trout (N_e <500). As discussed, such small, isolated populations are expected to be at elevated risk of extinction as a result of demographic stochasticity and population genetic phenomena such as loss of genetic diversity, 'mutational meltdown', and inbreeding depression. Additionally, these small isolated populations are at increased risk of extinction as a result of anthropogenic or environmental disturbances and, once lost, cannot be re-colonized naturally. Studies of other salmonid taxa have suggested that populations recently isolated in small habitats tend to be lost more quickly than those isolated in larger habitats (Morita and Yamamoto 2002).

Anthropogenic habitat disturbance

Anthropogenic habitat disturbance, together with over-fishing, is believed to have contributed to the decline of Rio Grande cutthroat trout in the late 19^{th} and early 20^{th} centuries (Cowley 1993), and it remains a potential threat to Rio Grande cutthroat trout populations. Stumpff and Cooper (1996) note that out of 83 Rio Grande cutthroat trout populations surveyed for habitat quality in New Mexico and Colorado, only 6 percent occurred in habitat conditions classified as 'excellent', 47 percent had 'good' habitat, 41 percent 'fair' habitat, and 6 percent 'poor' habitat. Reduction in habitat quality was most often caused by grazing, with mining, logging, and road construction also affecting some populations. However, in most cases habitat degradation associated with anthropogenic activity currently appears to be minimal or localized, with sedimentation and grazing impacts being the major problems (U.S. Fish and Wildlife Service 2002). Timber harvest in national forests has declined appreciably over the past two decades, and construction of new roads is minimal; in addition grazing practices may be improving in some areas (U.S. Fish and Wildlife Service 2002). Water extraction for irrigation purposes and urban or recreational development may represent threats to current or potential cutthroat trout habitat in some areas (Colorado Division of Wildlife 2004). In addition to reducing the habitat area available to cutthroat trout as a result of reduced stream flow, water extraction activities can cause the entrapment of trout in associated structures such as diversion ditches (Schrank and Rahel 2004). Restoring aquatic and riparian habitat and designing land management activities to reduce impacts on Rio Grande cutthroat trout populations are priorities for management agencies; however coordination between agencies such as CDOW and the USFS is important to ensure that activities are directed towards Rio Grande cutthroat trout conservation.

Natural habitat disturbance

Natural habitat disturbance is also a potential threat. The high elevation streams to which Rio Grande cutthroat trout are currently restricted are often characterized by extreme and fluctuating environmental conditions (Novinger and Rahel 2003). Drought, ice formation, high volume water flows, and forest fires can severely impact populations, and population fragmentation means that natural re-colonization can no longer occur following such events. Rio Grande cutthroat trout in first- and second-order streams may previously have survived adverse conditions by migrating downstream and re-colonizing when conditions became more favorable, but the presence of migration barriers now precludes this. Reduction in stream flow volume as a result of drought reduces habitat area available to trout. In some cases, streams may dry out completely, particularly where water extraction activities are occurring. High volume snowmelt flows have the potential to reduce population size and recruitment by moving fish downstream past the migration barrier and scouring redds (Strange et al. 1992). However, since adult cutthroat trout appear to be relatively resistant to displacement by flooding (Harvey et al. 1999), and Rio Grande cutthroat trout spawn on the descending limb of the snowmelt hydrograph, the impact of such high flows may be minor. Wildfires are a frequent occurrence in forested watersheds, and although trout may survive the fire itself, subsequent ash flows or the entry of fire-retardant slurry into streams may eliminate entire populations. Several Rio

Grande cutthroat trout populations have been negatively impacted by fire or drought in the last 10 years (New Mexico Department of Game and Fish unpublished data, Alves 1996 – 2004, U.S. Fish and Wildlife Service 2002). However, both catastrophic fire and drought may also provide opportunities to reclaim waters for Rio Grande cutthroat trout, by eliminating populations of non-native fish. Fire risk can be reduced through fuels reduction and prescribed burns. Loss of riparian forest cover as a result of wildfires, blow-downs, insect damage, or disease can also lead to increased deposition of fine sediment into a stream, changes in channel morphology, and greater fluctuations in water temperature (Swanston 1991).

Over-utilization

Over-utilization for commercial, recreational, scientific, or educational purposes does not currently appear to threaten continued persistence of Rio Grande cutthroat trout. All angling for Rio Grande cutthroat trout is recreational only. Cutthroat trout in general appear more vulnerable to angling capture than non-native trout taxa (Behnke 1992). However, the majority of extant Rio Grande cutthroat trout populations contain relatively small fish, are located in remote headwater drainages with difficult access, and hence suffer relatively little fishing pressure. Rio Grande cutthroat trout waters in Colorado and New Mexico considered especially vulnerable to angling pressure are protected by special regulations (Colorado Division of Wildlife 2005, New Mexico Department of Game and Fish 2005). Scientific collections are regulated by the CDOW and NMDGF via a permit system. Modern methods of testing for genetic purity utilize small fin-clip samples and are therefore non-lethal. Disease testing requires sacrifice of fish but where possible concentrates on non-natives adjacent to or within Rio Grande cutthroat trout populations (Colorado Division of Wildlife 2004). Utilization of a population as a source of fish for translocations or as a source of gametes to develop a hatchery broodstock could have a deleterious effect on that population as total population size and/or annual reproductive output are reduced by such manipulations. No study has yet examined the impact of such manipulations on Rio Grande cutthroat trout populations.

Disease

Whirling disease has been present in both New Mexico and Colorado for approximately two decades (U.S. Fish and Wildlife Service 2002), and it occurs in several drainages containing Rio Grande cutthroat trout populations. At one time, 13 state hatcheries in New Mexico and Colorado tested positive for the disease, but intensive clean-up and hatchery modification programs have now eliminated *Myoxobulus cerebralis* from the majority of these facilities (Nehring 2006). Both states have regulations and policies in place intended to control the further spread of whirling disease. Transmission of whirling disease requires the secondary host *Tubifex*, which is rarely abundant in clear coldwater streams (U.S. Fish and Wildlife Service 2002). In addition, infection rates tend to be low at low water temperatures (<10 °C Thompson et al. 1999). It has been suggested that these factors may help to limit whirling disease impacts within the high-elevation habitats to which Rio Grande cutthroat trout are currently restricted (U.S. Fish and Wildlife Service 2002). De la Hoz Franco and Budy (2004) investigated the occurrence of whirling disease within cutthroat trout in Utah and found the lowest prevalence to be in low-discharge headwater streams with an average summer temperature <9.5 °C (49 °F). Conversely, high prevalence of *M. cerebralis* was associated with temperatures >12 °C (54 °F) and high stream discharges. The presence of migration barriers is also expected to help protect the Rio Grande cutthroat trout from the spread of the disease. Nevertheless, whirling disease, together with other exotic salmonid diseases that may accidentally be introduced into New Mexico and Colorado in the future, remains a potential threat to Rio Grande cutthroat trout populations.

Bacterial kidney disease was present in the Rio Grande cutthroat trout broodstock population in Haypress Lake in 1995 (Harig and Fausch 1996), but this has not recently been noted as a problem (Alves 2003, 2004), and disease control procedures are sufficient to prevent transmission to hatchery broodstock or wild populations. Currently no other pathogens are known to pose a significant threat to Rio Grande cutthroat trout populations in Region 2.

Conservation Status of Rio Grande Cutthroat Trout in Region 2

Since 1973, the Colorado Division of Wildlife has implemented an aggressive conservation plan for Rio Grande cutthroat trout in USFS Region 2. This has resulted in the successful introduction or re-introduction of pure, naturally reproducing Rio Grande cutthroat trout populations into numerous streams and lakes and improved protection and monitoring for most populations (**Table 1**; Colorado Division of Wildlife 2004). Additionally a Rio Grande cutthroat trout broodstock has been developed as a source of fish for the purposes of conservation and recreational angling

(Colorado Division of Wildlife 2004). During the same time period, however, at least 15 historic populations, of varying degrees of genetic purity, have been lost (Harig and Fausch 1996, Alves 1996 - 2004). The primary risk factor for Rio Grande cutthroat trout in USFS Region 2 appears to be invasion of populations by non-native trout, in particular brook trout. Currently, 33 populations co-exist with non native trout (**Table 2**; Colorado Division of Wildlife 2004), with at least 17 populations having been invaded or re-invaded by non-natives in the past three decades (Alves 1996 - 2004). Temporary habitat loss due to drought and/or water extraction in certain years also appears to be a major threat to Rio Grande cutthroat populations in Colorado (Alves 1996 – 2004, Colorado Division of Wildlife 2004). Drought conditions in 2002 were implicated in the loss or major decline of several Rio Grande cutthroat trout populations (**Table 2**; Alves 2003, 2004). Insufficient habitat, small population size, poor recruitment, habitat damage by livestock, fine sediment deposition associated with logging or road use, angling pressure, housing development, and the presence of whirling disease are additionally noted as potential problems for some populations (**Table 2**; Alves 1996 – 2004, Harig and Fausch 1996, Colorado Division of Wildlife 2004, Nehring 2004, 2005). Natural phenomena such as forest fires, anchor ice formation, and flash floods are other potential threats.

The CDOW Conservation Plan for Rio Grande cutthroat trout (Colorado Division of Wildlife 2004) documents 'self-sustaining' (i.e., naturally reproducing) populations of Rio Grande cutthroat trout in 78 water bodies in Colorado (**Table 1**, **Table 2**). Thirty-two of these populations were created in the past few decades by transplanting fish from existing populations or hatchery stock into suitable habitat (**Table 1**; Colorado Division of Wildlife 2004). An additional transplant population was created in 2003 - 2004 in Big Spring Creek within the Rio Grande National Forest. Water bodies receiving these transplants were either previously fishless or had been chemically treated in order to remove non-native trout. In certain cases, introgressed Rio Grande cutthroat trout may have been removed along with the non-native fish. The most frequent source of fish for transplants has been West Indian Creek in the Alamosa-Trinchera drainage, with fish also being transplanted from Torcido and Placer creeks in the Alamosa-Trinchera drainage and Osier Creek in the Conejos drainage. In many cases, streams received fish from multiple populations. More recently, transplant populations have been created using broodstock fish from Haypress Lake and Pitkin Hatchery, which also have at least part of their genetic origins in Osier, Placer, and West Indian creeks.

Surveys performed in 2003 and 2004 (Alves 2003, 2004) showed that four of the 46 historic populations documented in the Conservation Plan (Grayback Creek, South Fork Placer Creek, Deep Creek, Wannamaker Creek) and two of the 32 documented transplant populations (Unknown Creek, Little Medano Creek) appeared to have become extirpated, primarily as a result of drought conditions in 2002 and/or the impact of non-native trout. A further three water bodies, whose population status was previously unknown, were found to contain no Rio Grande cutthroat trout in recent surveys (Fish Creek, North Fork West Indian Creek, South Fork Jim Creek; **Table 2**; Alves 2003, 2004). Several of the above streams are tributaries to larger creeks containing Rio Grande cutthroat trout, and they may never have supported permanent trout populations. Natural re-colonization of these streams may be possible when conditions are favorable. Alves (2004) notes that little natural reproduction is expected in Glacier Lake, and periodic stocking of Rio Grande cutthroat trout will therefore be necessary to maintain this population. The population in Upper West San Francisco Lake also appears to be primarily maintained by stocking (Alves 2004). The Rio Grande cutthroat trout population in Pass Creek West Fork appears to consist only of non-reproducing stocked fish from an adjoining recreation population (Alves 2003). Surveys were performed for 26 Rio Grande cutthroat trout populations during 2005 (**Table 2**), but the data from these surveys were unavailable at the time of publication of this Assessment. Although no further population extirpations were noted, no Rio Grande cutthroat trout were found at survey sites in Tuttle Creek, and further investigation is planned (Alves personal communication 2006).

Populations of Rio Grande cutthroat trout in Region 2 vary in their genetic purity. Early studies assessed level of non-native introgression using meristic traits; populations were graded from A to D, with 'A' indicating fish whose morphology corresponded to pure Rio Grande cutthroat trout and 'D' indicating fish exhibiting distinct hybridization. More recent studies have utilized mitochondrial DNA and diagnostic nuclear genetic markers (i.e., PINEs, BiAms and allozymes; see later discussion; Colorado Division of Wildlife 2004, Douglas and Douglas 2005, Colorado Division of Wildlife unpublished data) to test for introgression from Yellowstone cutthroat trout, Snake River cutthroat trout, and rainbow trout. Thirty-one of the 40 historic Rio Grande cutthroat trout populations documented in the Conservation Plan have now been examined using these genetic markers (**Table 1**). Currently, 23 of these populations appear to conform to the 'Core Conservation Population' definition

Table 2. Results of population and habitat surveys for self-sustaining Rio Grande cutthroat trout populations within US Forest Service Region 2 that are documented in the Colorado Division of Wildlife Conservation Plan (2004). Populations wholly or partially occurring within the Rio Grande National Forest are shown in bold.

Sub-basin	Water body	km (mi) or *ha (ac)*	Barrier[1]	Last survey date[2]	Population estimate[3]	Current status[4]	Non-natives[5]	Habitat problems[6]: None noted	High gradient	Low temps	High temps	Bank damage	Fine sediments	Grazing/roads	Water diversion	Drought impact	Beaver activity	Other potential problems and further notes
Alamosa-Trinchera	**Cat Creek**	**10.0 (6.2)**	**Water diversion**	**1997**	**3224**	**Secure, Stable**	**none**					**x**	**x**	**g**			**x**	
Alamosa-Trinchera	**Cat Creek North Fork**	**5.8 (3.6)**	**see Cat Creek**	**none**	**unknown**	**unknown**	**none**	**x**										**Not surveyed**
Alamosa-Trinchera	**Cat Creek South Fork**	**6.4 (4.0)**	**see Cat Creek**	**none**	**unknown**	**unknown**	**none**	**x**										**Not surveyed**
Alamosa-Trinchera	Alamosito Creek	10.3 (6.4)	none	2002*	627	At Risk, Declining	none		x				x					Few pools
Alamosa-Trinchera	Cuates Creek	7.4 (4.6)	Water diversion	2000*	1168	Secure, Stable	none								x			
Alamosa-Trinchera	Culebra Creek North Fork	11.0 (17.7)	none	1992	462	unknown	BNT, BKT, RBT	x										
Alamosa-Trinchera	Jaroso Creek	9.3 (5.8)	Road culvert	2003*	3434	Secure, Stable	none	x										
Alamosa-Trinchera	Torcido Creek	9.5 (5.9)	Water diversion	2003*	3109	Secure, Stable	none									x		
Alamosa-Trinchera	Vallejos #2	4.8 (3.0)	none	1999*	960	At Risk, Stable	BNT	x										
Alamosa-Trinchera	Vallejos North Fork	11.3 (7.0)	none	1999*	1645	At Risk, Stable	BNT	x										Nearby mine
Alamosa-Trinchera	Willow Creek	12.9 (8.0)	Water diversion	2003	3000	Secure, Expanding	YSC							g				Few pools
Alamosa-Trinchera	**Jim Creek**	**9.8 (6.1)**	**Gabion, failed**	**2004**	**1550**	**At Risk, Stable**	**BKT**					**x**	**x**	**g**				
Alamosa-Trinchera	**Jim Creek South Fork**	**3.2 (2.0)**	**see Jim Creek**	**2004**	**0**	**No RGCT**	**none**											**Intermittent stream**
Alamosa-Trinchera	**Torsido Creek**	**6.6 (4.1)**	**Gabion, failed**	**1999***	**33**	**At Risk, Declining**	**BKT**					**x**	**x**	**g**				

Table 2 (cont.).

Sub-basin	Water body	km (mi) or *ha (ac)*	Barrier[1]	Last survey date[2]	Population estimate[3]	Current status[4]	Non-natives[5]	Habitat problems[6]										Other potential problems and further notes
								None noted	High gradient	Low temps	High temps	Bank damage	Fine sediments	Grazing/roads	Water diversion	Drought impact	Beaver activity	
Alamosa-Trinchera	**Rhodes Gulch**	**5.5 (3.4)**	**Waterfall**	**2001**	**690**	**Secure, Stable**	**none**					**x**						
Alamosa-Trinchera	**Rough Canyon**	**1.6 (1.0)**	**Mineral pollution**	**2001**	**219**	**Secure, Stable**	**none**	**x**										
Alamosa-Trinchera	Placer Creek	8.4 (5.2)	Gabion, failed	2003*	166	At Risk, Declining	BKT					x	x	g		x		
Alamosa-Trinchera	Grayback Canyon	5.6 (3.5)	see Placer Creek	2003	0	Extirpated	BKT		x							x		Few pools
Alamosa-Trinchera	Little Ute Creek	4.2 (2.6)	Waterfall	2004	686	Secure, Expanding	none	x										
Alamosa-Trinchera	Placer Creek Middle Fork	7.7 (4.8)	see Placer Creek	2003	173	At Risk, Declining	BKT						x	g				Timber harvest
Alamosa-Trinchera	Placer Creek South Fork	7.6 (4.7)	see Placer Creek	2003	0	Extirpated	BKT			x		x	x			x		
Alamosa-Trinchera	Sangre de Cristo Creek	27.4 (17.0)	Water diversion	2001	3621	At Risk, Stable	BKT				x		x				x	Whirling disease positive
Alamosa-Trinchera	Ute Creek	20.3 (12.6)	see Sangre de Cristo	2000	983	At Risk, Declining	BKT, RBT							g	x			Flash floods
Alamosa-Trinchera	Ute Lake Lower	*1.2 (3.0)*	Waterfall	none	unknown	unknown	none	x										
Alamosa-Trinchera	Ute Lake Upper	*0.8 (2.0)*	Waterfall	none	unknown	unknown	none	x										
Alamosa-Trinchera	Wagon Creek	18.5 (11.5)	see Sangre de Cristo	2001*	5796	At Risk, Stable	BKT				x		x				x	
Alamosa-Trinchera	West Indian Creek	10.3 (6.4)	Gabion	2003	1690	At Risk, Declining	BKT	x										
Alamosa-Trinchera	West Indian Creek North Fork	4.8 (3.0)	see W I Creek	2004	0	No RGCT	BKT											Intermittent stream

Table 2 (cont.).

Sub-basin	Water body	km (mi) or *ha (ac)*	Barrier[1]	Last survey date[2]	Population estimate[3]	Current status[4]	Non-natives[5]	Habitat problems[6]: None noted	High gradient	Low temps	High temps	Bank damage	Fine sediments	Grazing/roads	Water diversion	Drought impact	Beaver activity	Other potential problems and further notes
Alamosa-Trinchera	West Indian Creek South Fork	9.7 (6.0)	see W I Creek	2004	522	unknown	BKT	x										
Alamosa-Trinchera	San Francisco Creek	14.5 (9.0)	Water diversion	2004*	3312	Secure, Stable	none						x				x	Housing development
Alamosa-Trinchera	**San Francisco Creek Middle Fork**	**9.3 (5.8)**	**Gabion**	**2004**	**275**	**Secure, Stable**	**none**	**x**									**x**	
Alamosa-Trinchera	**San Francisco Lake Upper West**	*1.7 (4.2)*	**see South Francisco Middle Fork**	**2004**	**unknown**	**Stocked**	**none**											**Winter kill**
Alamosa-Trinchera	Deep Canyon	5.3 (3.3)	see Trinchera #2	1999*	429	At Risk, Stable	BKT		x	x			x					
Alamosa-Trinchera	Trinchera #2	10.1 (6.3)	Gabion, failed	1983	1827	unknown	BNT, BKT, RBT	x										
Alamosa-Trinchera	Trinchera Creek North Fork	12.4 (7.7)	Gabion	1998	1324	At Risk, Stable	BKT	x										
Alamosa-Trinchera	Trinchera Creek South Fork	15.0 (9.3)	see Trinchera #2	1999	800	At Risk, Declining	BKT	x									x	
Chama	Nabor Creek	3.7 (2.3)	Gabion, failed (NM)	1995*	2323	Secure, Stable	none	x										Extends into NM
Chama	Native Lake	*2.0 (5.0)*	unknown	1996	100	Secure, Stable	none											Winter kill
Chama	Rio Chamita	6.4 (4.0)	none	1985	424	unknown	BNT	x										Not surveyed
Chama	Sexto Creek	5.6 (3.5)	none	1988	unknown	unknown	BNT, RBT, CRC	x										Not surveyed
Chama	**WoLake Fork Creek**	**1.4 (0.9)**	**Rail culvert**	**2001***	**2381**	**Secure, Stable**	**none**						**x**	**r**				**Timber harvest**

Table 2 (cont.).

Sub-basin	Water body	km (mi) or *ha (ac)*	Barrier[1]	Last survey date[2]	Population estimate[3]	Current status[4]	Non-natives[5]	Habitat problems[6] None noted	High gradient	Low temps	High temps	Bank damage	Fine sediments	Grazing/roads	Water diversion	Drought impact	Beaver activity	Other potential problems and further notes
Conejos	**Big Lake**	*4.8 (11.9)*	**See Conejos Lake Fork**	**na**	**na**	**New renovation**	**none**	**x**										
Conejos	**Canyon Verde**	**6.4 (4.0)**	**Cascades**	**1982**	**unknown**	**unknown**	**RBT, YSC**	**x**										
Conejos	**Conejos Lake Fork**	**6.6 (4.1)**	**Boulder barrier**	**na**	**na**	**New renovation**	**none**					**x**	**x**	**r**				
Conejos	**Rock Lake**	*2.0 (5.0)*	**Rockslide**	**unknown**	**unknown**	**unknown**	**BKT, BNT**											**Algal blooms**
Conejos	**Cascade Creek**	**4.0 (2.5)**	**Waterfall**	**2000**	**2875**	**Secure, Expanding**	**none**					**x**	**x**	**g**				
Conejos	**Osier Creek**	**4.2 (2.6)**	**Rail culvert**	**2004**	**3460**	**Secure, Stable**	**none**					**x**	**x**	**g**				
Conejos	**Rio de los Pinos #2**	**0.8 (0.5)**	**Waterfall**	**2003***	**178**	**Secure, Stable**	**none**			**x**								
Rio Grande Headwaters	**Pass Creek**	**14.6 (9.1)**	**none**	**2003**	**237**	**At Risk, Declining**	**BNT, BKT, RBT**					**x**		**g**				**Popular fishing stream**
Rio Grande Headwaters	**Pass Creek West Fork**	**1.6 (1.0)**	**Reservoir**	**2003**	**0**	**No RGCT**	**BKT**		**x**	**x**								**Proposed housing**
Rio Grande Headwaters	**West Alder Creek**	**13.2 (8.2)**	**none**	**2000***	**1246**	**At Risk, Declining**	**BKT**					**x**						
Rio Grande Headwaters	**West Bellows Creek**	**11.7 (7.3)**	**none**	**2004**	**4500**	**At Risk, Stable**	**BNT, BKT**	**x**										
San Luis	Medano Creek	20.9 (13.0)	Great Sand Dunes	2004*	14235	Secure, Stable	none								x	x		Some ORV erosion
San Luis	Medano Lake	*1.0 (2.7)*	see Medano Creek	none*	unknown	unknown	none	x										
San Luis	Little Medano Creek	8.8 (5.5)	see Medano Creek	2004	0	Extirpated	none									x		Sandy substrate

Table 2 (cont.).

Sub-basin	Water body	km (mi) or *ha (ac)*	Barrier[1]	Last survey date[2]	Population estimate[3]	Current status[4]	Non-natives[5]	Habitat problems[6]										Other potential problems and further notes
								None noted	High gradient	Low temps	High temps	Bank damage	Fine sediments	Grazing/roads	Water diversion	Drought impact	Beaver activity	
San Luis	Medano Hudson Branch	17.7 (11.0)	Culvert	2004	2403	Secure, Stable	none		x						x	x		Few pools
Sanguache	Carnero Creek	13.4 (8.3)	Water diversion	1998	465	At Risk, Declining	BNT				x	x	x		x		x	Low flow
Sanguache	**Carnero Creek Middle Fork**	**15.8 (9.8)**	**Gabion**	**2003***	**357**	**At Risk, Declining**	**none**						**x**	**g,r**		**x**	**x**	**Possible whirling disease**
Sanguache	**Carnero Creek North Fork**	**13.5 (8.4)**	**Gabion**	**2003***	**177**	**At Risk, Declining**	**BNT**					**x**	**x**	**g**		**x**		**Low flow**
Sanguache	**Carnero Creek South Fork**	**22.2 (13.8)**	**see Carnero Creek**	**2000**	**26123**	**At Risk, Stable**	**BNT**					**x**	**x**				**x**	**Culvert problems**
Sanguache	**Cave Creek**	**9.8 (6.1)**	**see Carnero Creek**	**2000**	**397**	**At Risk, Stable**	**BNT, BKT**	**x**										
Sanguache	**Miners Creek**	**6.9 (4.9)**	**see Carnero Creek**	**2004**	**1655**	**At Risk, Stable**	**BKT**	**x**									**x**	
Sanguache	**Prong Creek**	**5.8 (3.6)**	**see Carnero Creek**	**1998**	**475**	**At Risk, Declining**	**BKT**						**x**				**x**	
Sanguache	**Cross Creek**	**8.0 (5.0)**	**see Jacks Creek**	**2000***	**3310**	**Secure, Stable**	**none**					**x**	**x**	**g**				**Flood damage**
Sanguache	**Jack's Creek**	**14.3 (8.9)**	**Water diversion**	**2000***	**1104**	**At Risk, Stable**	**BKT**						**x**	**g**	**x**			
Sanguache	**Big Spring Creek**	**unknown**	**Gabion**	**na***	**na**	**New introduction**	**none**	**x**										
Sanguache	**Middle Creek East**	**12.4 (7.7)**	**Intermittent drying**	**1997***	**1656**	**Secure, Stable**	**none**						**x**	**g,r**				**Intermittent at terminus**
Sanguache	**Middle Creek East**	**5.1 (3.2)**	**Waterfall**	**1994***	**646**	**Secure, Stable**	**none**					**x**						
Sanguache	**Tuttle Creek**	**11.1 (6.9)**	**Gabion, failed**	**1996***	**1546**	**At Risk, Declining**	**BKT**					**x**		**g**				
Sanguache	**Deep Creek**	**5.2 (8.4)**	**none**	**2003**	**0**	**Extirpated**	**BKT**		**x**							**x**		**Flash floods**

Table 2 (concluded).

Sub-basin	Water body	km (mi) or *ha (ac)*	Barrier[1]	Last survey date[2]	Population estimate[3]	Current status[4]	Non-natives[5]	Habitat problems[6]										Other potential problems and further notes
								None noted	High gradient	Low temps	High temps	Bank damage	Fine sediments	Grazing/roads	Water diversion	Drought impact	Beaver activity	
Sanguache	**Unknown Creek**	**5.0 (3.1)**	**Cascades**	**2003**	**0**	**Extirpated**	**none**			x								**Few pools**
Sanguache	**Wannamaker Creek**	**10.9 (6.8)**	**none**	**2003**	**0**	**Extirpated**	**BKT**					x		g			x	**Nearby mine**
Sanguache	**Whale Creek**	**4.8 (3.0)**	**Waterfall**	**2003**	**400**	**At Risk, Declining**	**none**									x	x	**Few pools**
Upper Rio Grande	Costilla Creek East Fork	3.7 (2.3)	Gabion (NM)	2004	1532	unknown	none	x										
Upper Rio Grande	Costilla Creek West Fork	2.3 (1.4)	Gabion (NM)	2004	739	unknown	none	x										
Upper Rio Grande	Glacier Lake	*2.6 (6.5)*	Isolated Lake	2004	unknown	Stocked	none	x										Little spawning habitat
Canadian	Little Vermejo Creek	9.7 (6.0)	Gabion (NM)	2003	few	At Risk, Declining	BKT						x					Few pools
Canadian	Fish Creek	5.6 (3.5)	See Little Vermejo	2003	0	No RGCT	BKT	x										
Canadian	Ricardo Creek	9.7 (6.0)	Gabion (NM)	1996	180	At Risk, Declining	BKT	x										Extends into NM

[1] Barrier = presence and type of barrier protecting population from non-native trout

[2] Last survey date = most recent year (prior to 2005) that the population was surveyed, * = the population was also surveyed in 2005, but data are not yet available

[3] Population estimate = the estimated number of individuals >120 mm (>4.7 inches) present in the population in the survey year indicated

[4] Current status = status of population as recorded by Colorado Division of Wildlife in 2005

[5] Non-natives = the presence of non-native trout within the population, BRN = brown trout, BRK = brook charr, CRC = Colorado River cutthroat trout, RBT = rainbow trout, YSC = Yellowstone cutthroat trout

[6] Habitat problems noted during Colorado Division of Wildlife surveys (Harig and Fausch 1996, Alves 1997-2004) and USDA Forest Service surveys (unpublished data) are indicated by 'x'. Under grazing/roads, 'g' indicates that some habitat problems are attributed to grazing pressure, 'r' indicates that some habitat problems are attributed to the presence of roads

of the Utah Position Paper ('<1 percent non-native introgression'). A further four populations conform to the 'Conservation' definition ('<10 percent non-native introgression) and four to the 'Sportfish' definition (containing up to 60 percent non-native introgression, primarily from Yellowstone cutthroat trout; Douglas and Douglas 2005). While there is generally good agreement between the results of studies using meristics and studies using various different genetic markers, several populations that previously tested pure using allozymes exhibit evidence of relatively recent hybridization using PINEs (Douglas and Douglas 2005, Colorado Division of Wildlife unpublished data). Of the nine extant historic populations which have not be tested using genetic markers, three are graded A or A+ and three graded B or B+ based on meristic studies, and two are in tributaries to Core populations and are therefore also considered Core (**Table 1**). Transplant populations are generally assumed to correspond to the Core Conservation Population definition, and genetic studies have confirmed this for 10 of these populations. However, two contiguous transplant populations, Rough Canyon and Rhodes Gulch, have recently been found to contain some genetic material from Yellowstone cutthroat trout, the source of which is unknown. Taking into account the above information, we conclude that a maximum of 65 self-sustaining Rio Grande cutthroat trout populations corresponding to the 'Core' or 'Conservation' definitions currently exist within USFS Region 2.

A number of these self-sustaining Rio Grande cutthroat trout populations occur in interconnected tributaries with no documented migration barriers between them and hence might be more accurately considered as single continuous populations (or 'metapopulations'; Young 1996, Colorado Division of Wildlife 2004). These include populations in the Carnero Creek system (Carnero Creek, Carnero Creek South Fork, Cave Creek, Miner's Creek, Prong Creek, total estimated stream length = 58 km [36 mi.], total estimated adult population >29,000); populations in the Sangre de Cristo and Ute Creek systems (Placer Creek, Middle Fork Placer Creek, Sangre de Cristo Creek, Ute Creek, Wagon Creek, total estimated stream length = 82 km [51 mi.], total estimated adult population >10,000); populations in the West Indian Creek system (West Indian Creek, South Fork West Indian Creek, total estimated stream length = 20 km [12 mi.], total estimated adult population >2,000); populations in the Trinchera Creek system (Deep Canyon, South Fork Trinchera Creek, Trinchera #2, total estimated stream length = 30.4 km [19 mi.], total estimated adult population >3,000); populations in the Cat Creek system (Cat Creek, North Fork Cat Creek, South Fork Cat Creek, total estimated stream length = 22.2 km [14 mi.], total estimated adult population >3,000); populations in the Jack's Creek system (Cross Creek, Jack's Creek, total estimated stream length = 22.3 km [14 mi.], total estimated adult population >4,000), populations in the Vallejos Creek system (North Fork Vallejos Creek, Vallejos #2, total estimated stream length = 16 km [10 mi.], total estimated adult population >4,000) and populations in the Costilla Creek system, extending into New Mexico (Costilla Creek, East Fork Costilla Creek, West Fork Costilla Creek, total estimated stream length = 7.5 km [4.6 mi.], total estimated adult population >3,000). Following the terminology of May et al. (2003), therefore, the 65 self-sustaining Rio Grande cutthroat trout populations documented in Colorado are distributed over 41 isolates and eight metapopulations. Unfortunately, the majority of metapopulation systems have been invaded by brook trout or brown trout or contain some hybridized fish. Additionally, whirling disease is present in at least one system (see later discussion). Artificial fragmentation may therefore be necessary to protect individual streams from further introgression, the incursion of non-native trout or the spread of disease. For example, construction of a barrier to prevent brook trout invasion into Trinchera Creek North Fork has recently isolated this creek from the rest of the Trinchera system (Alves 2000). Similarly, constructions of barriers on the Middle and North Forks of Carnero Creek in 2002 and 2003 in order to protect native fish populations isolated these populations from the rest of the Carnero system (Alves 2002, 2003). In addition, other natural or artificial migration barriers may be present, such as beaver ponds, cascades, and road culverts, which limit the movement of individuals within these systems. Nevertheless, the existence of such interconnected populations of Rio Grande cutthroat trout bodes well for the creation of secure 'metapopulations' in the future.

As previously discussed, Rio Grande cutthroat trout populations are potentially at risk from habitat degradation, the incursion of non-native salmonids, and the demographic processes associated with small, fragmented populations. Taking account of these factors, Colorado Division of Wildlife classifies the stability of populations of Rio Grande cutthroat trout using a suite of population parameters. A 'secure population' has a minimum of 500 fish >120 mm in length, successful reproduction in four years out of 10, a minimum biomass of 20 lb. per acre generated through natural reproduction, a physical, chemical, or biological barrier separating the population from other salmonids, and is not considered to be impacted by insufficient habitat size or quality or the presence of non-native salmonids (Colorado Division of Wildlife 2004). In practice, CDOW classifies several populations as 'secure' that conform to the other criteria but have estimated population sizes <500 (**Table 2**). Colorado Division of Wildlife also classifies populations as 'stable', 'declining', or 'expanding' based on changes in estimated population size between repeated surveys. Currently, 20 out of the 65 documented self-sustaining Rio Grande cutthroat trout populations in Region 2 are classified by CDOW as 'secure and stable' or 'secure and expanding', 12 as 'at risk and stable', 16 as 'at risk and declining', and the remaining 17 as 'unknown' (**Table 2**; Alves 2003, 2004, Colorado Division of Wildlife 2004). All of the six populations that have been re-classified as 'extirpated' since the publication of the Conservation Plan were previously classified as 'at risk', and four were estimated to contain <200 adult fish in previous surveys.

Young and Harig (2001) provide a critique of the use of the above population parameters to assess population security in cutthroat trout. They point out that a closed population containing 500 individuals >120 mm in length is expected to have an N_e much smaller than 500, the minimum N_e recommended to avoid loss of genetic diversity. This is both because many individuals of this size may still be reproductively immature and because, in any taxon, N_e is generally much lower than the number of reproductively mature adults in the population. If Rio Grande cutthroat trout exhibit an N_e/N_{adult} ratio similar to that calculated for stream-dwelling brown trout (Palm et al. 2003, Jensen et al. 2005), then more than 2500 adults may be required to meet the target N_e of 500. We note, however, that the estimated number of fish >120 mm greatly exceeds 500 in many self-sustaining Rio Grande cutthroat trout populations in Colorado (Colorado Division of Wildlife 2004). Thirty-seven of the documented populations are currently estimated to contain at least 500 individuals >120 mm in length; 28 of these have an estimated population size >1000, and 13 have an estimated population size >2500. Additionally, out of 28 populations with a estimated population size <1000, 11 form portions of the interconnected stream systems described above and are therefore expected to be able to receive immigrants from other populations, and the census population size of one (Middle Fork San Francisco Creek) is thought to be an underestimate due the presence of numerous Rio Grande cutthroat trout in beaver ponds that could not be sampled (Colorado Division of Wildlife, unpublished data). Very few populations have sufficiently few individuals >120 mm that they could be considered at risk for inbreeding depression. For all populations, actual population sizes may be substantially larger than estimates due to the bias in the depletion method used by CDOW to enumerate fish (see later discussion). Young and Harig (2001) also note that experiencing year-class failures in six years out of 10 would put a population at high risk of extinction, a conclusion supported by the modeling efforts of Cowley (unpublished). However year-class failure generally appears to occur at a much lower rate than this within Rio Grande cutthroat trout populations in Colorado.

Colorado Division of Wildlife is currently undertaking a study to test cutthroat trout habitat in Colorado for the presence of whirling disease. By 2005, trout within at least eleven self-sustaining Rio Grande cutthroat trout populations (Big Lake, Cascade Creek, Carnero Creek Middle Fork and South Fork, Conejos Lake Fork , Jim Creek, Osier Creek, Rio de los Pinos, Sangre de Cristo Creek, Torsido Creek, Tuttle Creek, Alves 1999 - 2004; Nehring 2004, 2005) had been screened for *Myoxobulus cerebralis*. Nine of these populations tested negative, however whirling disease was found to be present in Sangre de Cristo Creek. The disease may also be present in Carnero Creek Middle Fork, although further testing is required for this population. The presence of whirling disease in Sangre de Cristo Creek is clearly a concern. As previously noted, there do not appear to be any significant barriers to fish movement between this population and the historic Rio Grande cutthroat trout populations in Placer Creek, Middle Fork Placer Creek, Ute Creek and Wagon Creek. Hence Rio Grande cutthroat trout in 82 km (51 mi) of habitat are potentially threatened by the disease.

In general, habitat problems for Rio Grande cutthroat trout populations on USFS land appear to be minor or localized. The two problems most commonly noted are bank damage as a result of grazing by cattle or elk and deposition of fine sediments from roads, primarily as a result of poorly designed stream crossings (**Table 2**; Alves 1996 - 2004, Harig and Fausch 1996, USDA Forest Service unpublished data). More serious problems have been noted outside the Rio Grande National Forest, in particular severe livestock impacts to Jim Creek and Torsido Creek on State Trust Land Board property.

In addition to the naturally-reproducing populations, CDOW maintains a further 83 Rio Grande cutthroat trout 'recreation populations' by periodic stocking of pure fish from hatchery stocks. The majority of these are located within the Rio Grande National

Forest. They are primarily managed for their sportfish benefit for the public and are frequently located in high-elevation lakes where cold water temperatures and lack of spawning habitat is expected to prevent natural trout reproduction. Some of these recreation populations have the potential to act as 'genetic refugia' for pure historic populations, but many also contain other *Oncorhynchus* taxa that are expected to hybridize with Rio Grande cutthroat trout where opportunities for natural reproduction occur (Colorado Division of Wildlife 2004). Captive Rio Grande cutthroat trout hatchery broodstocks are primarily maintained at Pitkin Hatchery. Colorado Division of Wildlife also maintains a large, naturally-reproducing, Rio Grande cutthroat trout population in Haypress Lake, containing genetic material from multiple populations, which is used as a 'feral broodstock'. The Haypress Lake population was first established in 1990 with fish from West Indian Creek, Placer Creek, and Osier Creek. The lake is stocked annually with approximately 6,000 fingerlings originating either from spawn taken at the lake or from Rio Grande cutthroat trout hatchery broodstocks. In turn, eggs from approximately 20 females are taken from Haypress Lake every year to augment the hatchery broodstocks. In 2002, 795 Rio Grande cutthroat trout from Placer Creek Middle Fork, West Indian Creek, and North Carnero Creek were marked and transplanted to Haypress Lake in response to drought conditions threatening these populations. Offspring of fish removed from Placer Creek Middle Fork were stocked into Placer Creek in 2003.

A number of streams within the Rio Grande National Forest are potentially suitable sites for the creation of new Rio Grande cutthroat trout populations. These include several streams from which Rio Grande cutthroat trout are believed to have been extirpated in the past few decades (e.g., Bennett Creek, La Garita Creek), habitat currently containing populations of non-native or introgressed cutthroat (e.g., Little Squaw Creek, Iron Creek, John's Creek), and stream reaches adjoining current Rio Grande cutthroat trout populations (e.g., Pass Creek).

Potential Management of Rio Grande Cutthroat Trout in Region 2

Implications and potential conservation elements

Rio Grande cutthroat trout were previously distributed throughout the Rio Grande drainage in Region 2, probably occurring in a variety of different fluvial habitats from headwater streams to the Rio Grande mainstem and possibly exhibiting a range of life-history strategies. Decline of the subspecies is believed to have commenced in the mid-1800's as a result of overfishing and habitat degradation associated with grazing, logging, mining, and water extraction for irrigation purposes (Cowley 1993). Stocking of non-native trout, commencing in the late 1800's further impacted Rio Grande cutthroat populations via hybridization and predation and/or competitive displacement. As a result, the Rio Grande cutthroat trout today is estimated to occupy less than 11 percent of its former range and is primarily restricted to small, high elevation streams and lakes, which in many cases may represent marginal trout habitat. Populations remain vulnerable to invasion by non-native trout, introduction of salmonid diseases, anthropogenic and natural habitat disturbance, and in certain cases over-exploitation by anglers. The small size and isolation of many of these populations theoretically also renders them at increased risk of extinction as a result of demographic and population genetic processes and stochastic environmental events. Most populations are unlikely to be re-colonized naturally once lost, and Rio Grande cutthroat trout are currently unable to re-expand into most suitable habitat due to the presence of naturalized populations of non-native trout or migration barriers.

Conservation of Rio Grande cutthroat trout necessitates measures to protect populations from anthropogenic habitat degradation, invasion by non-native trout, and disease transmission. The vulnerability of a population to extirpation as a result of demographic, genetic, and environmental factors will be reduced by increasing population size and enabling migration between populations. This can be achieved by improving the quantity and quality of habitat available to individual Rio Grande cutthroat trout populations, eliminating non-native trout from this habitat, and re-connecting isolated populations so that gene flow can occur between them and habitat can naturally be re-colonized following local extinctions. Where such goals cannot be achieved, artificial translocations of fish may be an alternative way to buffer populations against demographic stochasticity and loss of genetic diversity (see later discussion). The overall vulnerability of the subspecies will further be reduced by re-establishing new populations in currently unoccupied habitat. Ultimately, the continued persistence of Rio Grande cutthroat trout could most readily be guaranteed by halting the stocking of fertile non-native trout and eliminating naturalized populations of non-natives throughout a major portion of the subspecies' native range. However, this option is currently not politically, socially, practically, or economically feasible. Conservation of Rio Grande cutthroat trout in Region 2 will additionally be facilitated by sharing of data (e.g., habitat surveys) and co-operation between the relevant government agencies.

Tools and practices

Species inventory and monitoring

Management agencies most commonly use electrofishing apparatus to estimate fish species distribution or population size in small water bodies. Any survey or treatment utilizing electrofishing needs to take into account the potential harm this capture technique can cause to the fish population. Snyder (2003) provides a comprehensive overview of electrofishing theory and practice, with particular emphasis on the deleterious effects of electrofishing and approaches that can be used to minimize these.

Several stream networks in Colorado and New Mexico potentially contain remnant Rio Grande cutthroat trout populations but have not yet been inventoried for the presence of the subspecies (New Mexico Department of Game and Fish 2002, Colorado Division of Wildlife 2004). The most commonly used approach to quantifying the distribution of stream fish is presence-absence sampling using electrofishing equipment. Presence of trout in a stream is generally confirmed visually, and then electrofishing capture is used to determine species identity. Since electrofishing may only capture a portion of individuals within a water body (potentially less than 50 percent of the salmonids in streams; Peterson et al. 2004b, Rosenberger and Dunham 2005), a species may mistakenly be recorded as 'absent' from a stream where it is rare. This is unlikely to be a problem in the species-poor environment of headwater streams where most Rio Grande cutthroat trout remain, but there is still a possibility that the presence of Rio Grande cutthroat trout may be overlooked where the subspecies occurs at extremely low abundance in combination with high abundances of non-native trout. There have been several incidences in Colorado where population surveys failed to document Rio Grande cutthroat trout in streams where they were later found to be present (Alves 1996 - 2004).

Several methods are available to estimate the size of fish populations in streams. The two most widely used are depletion (removal) estimates and mark-recapture estimates (Lockwood and Schneider 2000). Of these, the depletion method is less labor-intensive, but the mark-recapture method appears to give more accurate results in habitat typical for Rio Grande cutthroat trout (Rogers et al. 1992, Rosenberger and Dunham 2005).

Using the depletion method, fish are captured using two or more subsequent electrofishing passes of a chosen stream section. Fish captured on each subsequent pass are either removed from the stream until the survey is completed, or they are marked and returned. Passes are generally repeated until a pre-specified number has been completed. Depending upon the number of electrofishing passes used, population size for the surveyed stream section is then calculated using relevant equations provided in, for example, Zippin (1956), Seber and Le Cren (1967), or Carle and Strub (1978), or alternatively using maximum likelihood methods such as those implemented in the programs MARK (White and Burnham 1999) or Microfish 3.0 (Van Deventer and Platts 1989). A habitat-wide population estimate is then calculated by extrapolating from the population estimate for the surveyed section. The depletion method relies on several assumptions for an accurate abundance estimate for the sampled section:

- ❖ there is negligible immigration or emigration from the sample site
- ❖ there is no variation in sampling effort between each pass
- ❖ there is no variation in capture probability between individuals or habitats or between successive samples.

Movement of fish into and out of a sample site can largely be prevented by fixing block nets at either end of the stream section being surveyed (Peterson et al. 2005). Variation in sampling effort can be minimized by the use of standardized electrofishing protocols. The expected variation in catchability between size classes and species when using electrofishing gear can to some extent be accounted for by calculating results separately for different sample groups (e.g., different age or size classes). Changes in fish catchability over successive passes may be reduced by allowing a sufficient recovery interval (e.g., 1 hour) between passes. If three or more sampling passes are performed, then the assumption of equal catchability between passes can be tested using the chi-squared test (White et al. 1982, although Rosenberger and Dunham (2005) question the efficacy of this test), and alternative methods of calculating fish abundance can be used if this assumption is violated. However, even when these precautions are taken, multi-pass electrofishing surveys are generally expected to over-estimate capture efficiencies and therefore underestimate population size (Nordwall 1999). Studies where the true number of fish is known suggest that this technique may underestimate abundances in mountain streams by 13 to 116 percent (Thompson 2003, Peterson et al. 2004). The extent of this under-estimation can depend upon size and species of fish being sampled and habitat characteristics (e.g., stream width and habitat complexity) (Kennedy and Strange 1981, Heggnes et al. 1990, Peterson et al. 2004b, Rosenberger and Dunham 2005). Peterson et al. (2004b) showed that multipass depletion surveys in small, high-elevation streams underestimated the abundance of westslope cutthroat trout by an average of 60 percent. Peterson et al. (2004b) and Rosenberger and Dunham (2005) recommend that biologists perform studies, for example using a known number of marked fish, to quantify the expected bias of their abundance estimation for the habitat and species that they expect to sample.

In the mark-recapture method, a random sample of fish is collected for a chosen stream section, marked, and returned to the stream. These fish are given the opportunity to re-disperse through the stream section (at least one day; Lockwood and Schneider 2000). A second random sample is then collected, and total population size is estimated from the proportion of marked fish in the new sample using relevant equations (Ricker 1975) or procedures implemented in programs such as MARK (White and Burnham 1999). This method assumes equal mortality and catchability of marked and unmarked fish, random mixing of the marked fish back into the unmarked population, and negligible movement of fish into and out of the study area. Again, emigration and immigration can be minimized using block nets, and variation in catchability between species and age classes can be controlled by calculating separate abundance estimates for different groups. Rogers et al. (1992) and Rosenberger and Dunham (2005) showed that this approach gave more accurate results than the depletion method when enumerating salmonids in small mountain streams.

A number of other approaches have been utilized to estimate trout populations in small habitats. Bankside, snorkeling, and single-pass electrofishing counts have the advantage of being less labor intensive than multi-pass electrofishing surveys. In some cases, they may also be less disruptive to fish populations. However, Peterson et al. (2005) found that trout in streams were more disturbed by snorkelers than by electrofishing. Bozek and Rahel (1991) found that streamside visual counts were useful for estimating abundance of cutthroat trout fry, but not for enumerating older fish. Mullner et al. (1998) found that day time snorkeling estimates of trout abundances and size length distributions in small streams with little instream cover were highly correlated with results from depletion surveys. In contrast, Roni and Fayram (2000) and Grost and Prendergast (1999) found that snorkeling day counts did not correlate with depletion survey results while night time counts did. Several studies have shown that single pass electrofishing surveys can be used to predict the abundances of stream-dwelling salmonids estimated using multi-pass surveys (Jones and Stockwell 1995, Kruse et al. 1998, Decker et al. 1999, Mitro and Zale 2000).

Using any enumeration technique, the accuracy of the total population estimate will depend upon the number of stream sections surveyed and the habitat composition of these sections. Some workers stratify streams into different habitat types and reaches, performing a count for each stratum and then extrapolating total population size according to the frequency of different habitat in the stream. This approach, however, can be very labor intensive and may not be feasible in small streams where habitat type changes over a few meters or in remote streams with difficult access. In an attempt to improve accuracy of population estimations while minimizing sampling effort and disruption to the fish, Hankin and Reeves (1988) propose a method that involves both snorkel surveys and electrofishing to estimate fish abundance in small streams. Snorkel surveys are first performed in a randomly selected unit within each habitat stratum; multipass electrofishing estimates are then performed for a subset of these strata and used to adjust for incomplete detection of fish by snorkelers.

Although, as previously discussed, none of the available survey techniques may provide an exact estimate of the number of trout in a stream, consistent use of the same methodology over repeated years can provide valuable information on population levels and trends. The usefulness of the data will be greatly improved if studies are performed to estimate the degree of bias expected from the enumeration protocol used. However, because of the large interannual variability in abundance known to occur in cutthroat trout populations, even if enumeration is accurate, multiple abundance estimates over an extended time period (e.g., 10 years; Roni et al. 2002) may be required in order to conclude with confidence that a population is expanding or declining, for example in response to a habitat manipulation. Colorado Division of Wildlife is currently undertaking a study to evaluate the sampling approaches being used to enumerate cutthroat trout and to determine how they might be improved (Colorado Division of Wildlife unpublished data).

Habitat monitoring

Assessing habitat quality for trout and monitoring habitat changes over time require a standardized protocol that will assess habitat at both the local and basin-wide scale. Various protocols have been developed to assess habitat quality for stream-dwelling salmonids (e.g., Binns 1979, Milner et al. 1998, Bain and Stephenson 1999). Elements of habitat that have been shown to be important for cutthroat trout, and that should therefore be included in such an assessment protocol, include stream length, stream width, stream gradient, number of deep pools, availability of cover such as that provided by large woody debris or undercut stream banks, availability of spawning gravels, availability of fry rearing habitat, summer water temperatures, and composition of riparian vegetation. Herger et al. (1996) note that variation in stream flow changes the physical features of stream habitats and therefore recommend that all habitat inventories be conducted at similar discharge levels. Impairments to cutthroat trout habitat that should be quantified in habitat quality assessments include bank damage by livestock, deposition of fine sediments, and the presence of artificial barriers to movement such as culverts.

Habitat quality assessments are of use only if the data are used to direct management activities, for example to identify and mitigate damage to streams containing extant Rio Grande cutthroat trout populations, or to identify suitable sites for re-introduction of the subspecies. In this context, sharing of data between the various agencies responsible for cutthroat trout management is important.

Habitat management approaches

The Regional Watershed Conservation Practices Handbook (FSH 2509.25), the revised land and resource management plan for the Rio Grande National Forest (USDA Forest Service 1996), and the management indicator species amendment for Rio Grande cutthroat trout (USDA Forest Service 2003) provide detailed guidance on land management considerations and practices that will prevent and mitigate anthropogenic impacts on salmonid habitat. We therefore deal with these only briefly in this section.

Habitat degradation as a result of excessive grazing pressure can most easily be reversed by excluding livestock from the riparian area; riparian vegetation generally recovers quickly with cessation of grazing (Platts 1991, Binns and Remmick 1994). Where this is not feasible, the impact of grazing can be reduced by decreasing the number of livestock or restricting grazing in the riparian area to limited time periods (Platts 1991). Providing alternative water supplies away from the riparian area may also help to reduce the impact from livestock. Binns and Remmick (1994) showed that collapsed banks armored with machine-placed rocks stabilized more rapidly following livestock exclusion than banks that were left to heal naturally. However, such construction activities may also negatively impact trout populations in the short term (Knudsen and Dilley 1987) and may have unintended effects on stream channel morphology. Implementation of natural channel design (Rosgen 1996) should improve both the efficacy of stream restorations and the success of habitat enhancements for salmonids.

Options to reduce the impact of timber harvest on trout habitat include retaining a streamside buffer zone, limiting the percentage of the watershed that can be cut, implementing measures to minimize the transport of surface sediment down slope, and applying stability modeling when planning clear cuts (Chamberlin et al. 1991). Where timber harvest has previously occurred adjacent to a stream, coniferous trees may be replaced by small hardwoods that are unable to provide the large woody debris important in structuring stream morphology. In this case, planting suitable tree species may be warranted. Romero et al. (2005) suggest that a mixture of conifers, deciduous trees, and shrubs in the riparian corridor will both provide large woody debris and maximize the inputs of nutrients and terrestrial vertebrates into the stream.

Correct construction and maintenance can greatly reduce the impact of roads and trails on trout habitat. Options to reduce road impacts include routing roads away from stream areas, installing suitable culverts, and gravelling road surfaces. Furniss et al. (1991) and the texts noted above provide advice. Where Rio Grande cutthroat trout habitat is threatened by water diversion, several options are available to ensure that sufficient stream flow is retained. These include applying for minimum stream flow rights through Colorado Water Conservation Board, implementing and enforcing Forest Reserve Water Rights, and purchasing water rights (Colorado Division of Wildlife 2004).

Several techniques are available to restore and improve habitat quality for stream-dwelling salmonids (Seehorn 1985, Reeves et al. 1991, Hunt

1993). Although many studies have shown increased salmonid abundance when these techniques have been implemented, not all are successful (Rinne 1981, Reeves et al. 1991, Binns 2004). The success of these approaches will depend upon the durability and suitability of habitat manipulations and upon characteristics of the watershed, stream, species, and population being addressed. For example, adding instream structures to create pools will not significantly improve trout numbers if these structures are washed away within a few years. Similarly, increasing the number of large pools available to adult fish may not increase carrying capacity of a stream if fry rearing habitat or availability of spawning gravels are the factors limiting population growth (Rosenfeld and Hatfield 2006). In practice, cost and accessibility are further considerations.

The most commonly used technique to improve habitat quality for salmonids is the introduction of large woody debris or other instream structures in order to increase the number of deep pools. Riley and Fausch (1995) and Gowan and Fausch (1996) showed that experimental installation of log weirs in high elevation Rocky Mountain streams resulted in increased pool volume, decreased current velocity, and increased depth and cover. Abundance of subadult and adult trout increased in treated sections compared to untreated sections, but abundance of juveniles was not affected. However, the observed increases in abundance appeared to be due to trout immigration into the study area rather than increased survivorship; hence installation of such structures in Rio Grande cutthroat streams may not increase population-level abundance if immigration into these streams is precluded by the presence of migration barriers.

Population isolation

Non-native trout are ubiquitous throughout the native range of Rio Grande cutthroat trout. Protection of Rio Grande cutthroat trout populations, therefore, requires measures to prevent invasion of non-native trout and monitoring of populations to ensure that such measures are effective. In some cases, pure Rio Grande cutthroat trout populations appear to have persisted because they are protected by a natural migration barrier, such as a waterfall. Other historic Rio Grande cutthroat trout populations in Region 2 appear to have been protected from the incursions of non-native trout by anthropogenic activity unrelated to fish conservation. For example, many populations appear to be protected by seasonal de-watering of the lower reaches of the stream as a result of water diversion for irrigation; others are protected by road or rail culverts or mine pollution.

Where a sufficient barrier is not present, exclusion of non-native trout generally requires construction of an artificial barrier. These barriers must be designed so that fish cannot jump upstream over them or swim around them during high water flows. Since the ability of salmonids to leap over obstacles depends upon having pools that provide a launching site (Bjorn and Reisner 1991), an important component of a fish barrier is the inclusion of a splash pad immediately downstream to prevent downcutting and pool formation. Barriers need to be located with care in order to ensure that they serve their intended purpose of protecting cutthroat trout populations. Location, design, and construction of a barrier will be influenced by its intended lifespan, local hydrology, landscape, stream features, ease of access, availability of materials, and cost. In some cases, a natural barrier may be artificially modified to improve its ability to block fish passage.

Artificial waterfalls are the type of barrier most often constructed to protect cutthroat trout habitat. Materials that may be used to construct falls barriers include gabion (rocks contained within wire mesh cages), wood and concrete. Thompson and Rahel (1998) showed that brook trout were able to pass upstream through crevices in gabion barriers. They recommended that an appropriate rock size (150-200 mm) be selected so that silt and gravel are able to accumulate in the interstitial spaces and note that 2-3 years of sediment accumulation may be required before these spaces are filled. Alternatively, installation of a hydrostatic material such as Mirifi 140N fabric on the upstream side of the barrier will prevent fish movement through the spaces (Colorado Division of Wildlife personal communication).

There appear to be no formal recommendations published regarding the height of migration barriers, but most agencies tend to use a minimum height of 1 m (3.3 ft.). Reiser and Peacock (1985) reported a maximum jumping height of 80 cm (2.6 ft.) for brown trout while Schrank and Rahel (2004) found that some Bonneville cutthroat trout were able to pass upstream over a barrier 1.1 m (3.6 ft.) in height. Kondratieff and Myrick (2006) found that brook trout up to 30 cm (11.8 inches) in length were unable to jump a barrier over 43.5 cm (1.4 ft.) in height, provided the depth of the plunge pool below the barrier was less than 10 cm (3.9 inches). With a deeper plunge pool, a higher barrier was required; where plunge pool depth was 40 cm (15.6

inches), for example, individuals >20 cm (7.8 inches) in length could jump as high as 73.5 cm (2.4 ft.).

Monitoring and maintaining all types of barriers are important to ensure that they continue to exclude non-native trout from Rio Grande cutthroat trout populations, as is monitoring of populations upstream of barriers to ensure that re-invasion has not occurred (Avenetti et al. 2006). Re-invasion of cutthroat trout populations by non-natives due to failure of artificial fish movement barriers has frequently been documented (e.g., Harig et al. 2000). The majority of gabion barriers constructed for the protection of Rio Grande cutthroat trout in Colorado have failed within five years (Alves 1996 - 2004).

Population isolation is currently necessary to protect extant pure cutthroat trout populations. However, as previously discussed, it may also have detrimental effects on the populations, making them more vulnerable to extinction due to demographic, population genetic, and environmental processes, and potentially selecting against mobile life-history strategies. In order to minimize these detrimental effects, Novinger and Rahel (2003) recommend a choice of barrier location that maximizes the area and quality of habitat isolated upstream. The models of Hilderbrand and Kershner (2000), Hilderbrand (2002), and Cowley (unpublished) and the work of Harig and Fausch (2002) provide guidance as to how habitat qualities and population carrying capacities might influence the chance of population persistence. Population genetic studies can also help guide barrier location, by providing information on whether construction of barriers will disrupt existing patterns of gene flow (Pritchard et al. submitted). As noted, reconnection of isolated populations by extending available habitat downstream to include the confluence of several streams containing Rio Grande cutthroat trout will enable expression of more mobile life history strategies, decrease the chance of individual population extinctions and allow natural re-colonization to occur. Shepard et al. (2005) note that control of disease and non-native trout is difficult in large, interconnected stream systems. For this reason, they recommend a management strategy for cutthroat trout that involves a combination of connected and isolated populations. They suggest that any increased extinction risk of isolated populations may be mitigated by replicating populations into new habitat, so that a population can be re-founded from the same genetic stock if lost. Clearly, this requires that the replicate population be founded from a sufficient number of individuals and be maintained at a sufficient size that it continues to represent the genetic diversity of the original population.

Population supplementation

Where population expansion and re-connection are not options, the process of migration between populations can be simulated by artificially moving fish between populations, or by introducing hatchery-reared individuals. Using the modeling approach detailed previously, Hilderbrand (2002) showed that such population supplementation was able to decrease the risk of isolated populations going extinct due to demographic processes. Again, supplementing with adults was more effective than supplementing with young fish; adding as few as 10 adults every 20 years greatly improved the probability of population persistence. Addition of fish from a different source may also act to increase population fitness in cases where morphological and genetic evidence suggests that local fish are suffering from inbreeding depression.

The role of supplementation in maintaining cutthroat trout populations is controversial. First, the degree to which introduced fish might contribute to an established population is unknown. Novinger and Rahel (2003), for example, monitored the abundance of juvenile hatchery-reared Colorado River cutthroat trout stocked into wild populations. They found that the supplemental fish failed to enhance population size; instead the majority (>99 percent) were lost from the population within three years, primarily as a result of movement downstream over the migration barrier. Even the small number of supplemental fish remaining after such emigration, however, may be sufficient to decrease the extinction risk (Hilderbrand 2002). Mesa (1991) suggests that maladaptive behavior of hatchery-reared fish may cause them to exhibit poor survival when stocked into streams. Miller (1954) found low survival of hatchery-reared cutthroat trout compared to wild cutthroat trout when both were stocked into the same stream. Second, although the genetic effect of a supplementation program can be positive, there is the potential for negative genetic effects on the population. Hatchery broodstock, for example, frequently become genetically adapted to the hatchery environment (e.g., McClean et al. 2005) and exhibit loss of genetic diversity over successive generations (Allendorf and Phelps 1980); incorporation of genetic material from such stock into the wild population may lower its fitness. The theoretical risk of outbreeding depression must also be a consideration when introducing either hatchery or wild-collected fish to a remnant cutthroat trout population. Such concerns can be minimized by managing a hatchery program to minimize adaptation to the artificial environment (e.g., Cowley 1993), or by selecting wild fish from populations that are expected historically to have exchanged migrants with the population to be supplemented, for example based on information regarding geographical proximity or measures of population genetic differentiation.

Population expansion and creation of new populations

Currently, most pure Rio Grande cutthroat trout are unable to re-expand into suitable habitat due to the presence of non-native trout and barriers to dispersal. However, the range of the subspecies can be increased artificially by introducing pure cutthroat trout into suitable habitat from which the non-native trout have been removed. Colorado Division of Wildlife and NMDGF have been successful in creating many new populations in this way. Water bodies that may be targeted for such population restorations include those that contain no cutthroat trout and those that contain populations of cutthroat trout that have hybridized with rainbow trout and other non-native cutthroat trout subspecies (e.g., 'sportfish populations' in the Utah Position Paper). In practice, selection of water bodies for restoration may also be influenced by logistical and social considerations such as ease of access and public support for re-introduction efforts.

The success of a cutthroat trout restoration effort depends upon a number of factors, including size and habitat characteristics of the stream or lake area targeted, efficiency of removal of non-native trout, and number and age of fish subsequently stocked. Harig and Fausch (2002), for example, note that out of 65 recorded attempts to establish new greenback and Rio Grande cutthroat trout populations via translocation prior to 1999, only 27 were considered successful; the rest failed because of re-invasion by non-native trout or unsuitable habitat. Translocation success requires sufficient habitat to support a self-sustaining population, despite demographic and environmental stochasticity, and sufficient habitat quality to meet life history requirements. Harig and Fausch (2002) surveyed 27 streams in New Mexico and Colorado that had received cutthroat trout translocations in the previous 3 to 31 years, and they found that a model incorporating stream width, number of deep pools, and mean summer water temperature was most suited to explaining the success of translocated populations. Those streams found to contain few (<100) or no cutthroat trout tended to be narrower with fewer deep pools, to contain fewer pools with physical structures, and to have colder July water temperatures than those streams with high trout densities. Stream length did not appear to be an important contributory factor to translocation success. However, Harig et al. (2000) previously demonstrated that habitat area (>2 ha) was an important factor influencing the success of greenback cutthroat trout translocations. Harig and Fausch (2002) also examined basin-scale features influencing success of translocations; although these were not as important as stream scale features in predicting translocation success, they suggested that streams in larger watersheds (>1470 ha [3631 acres] drainage area) were more likely to support higher numbers of cutthroat trout than those in smaller watersheds. As detailed previously, Hilderbrand and Kersher (2000) and Cowley (unpublished) provide suggestions as to the area of habitat that may maximize the chance of achieving a self-sustaining population of cutthroat trout with a long-term N_e >500 and a low probability of going extinct within 100 years.

Several methods are available to remove non-native fish from water bodies. The most commonly used approach is to apply a piscicide such as rotenone or antimycin-A (Hepworth et al. 2002, Finlayson et al. 2005). Use of such piscicides is generally more efficient and effective than alternative methods, but it has the disadvantage of being indiscriminate, killing both target and non-target fish taxa and potentially affecting all gill-breathing organisms within the treatment area. Additionally, use of piscicides has occasionally caused fish kills outside of the target area (e.g., Stumpff 1999), and public opposition to this approach may be high (Quist and Hubert 2004). The effectiveness of piscicides will depend upon factors such as lake or stream morphology, temperature and pH (Tiffan and Bergersen 1996), and species targeted. Salmonid eggs are generally not killed by piscicide treatment, and complex habitats such as beaver ponds and bogs may provide refugia that allow non-natives to survive (Harig et al. 2000). Multiple piscicide applications may be required to eliminate all non-native trout (Rinne and Turner 1991). Antimycin-A is generally preferred to rotenone in the treatment of streams due to its shorter half-life, increased ease of neutralization, increased effectiveness at cold water temperatures, lower toxicity to non-target organisms, and the fact that it cannot be detected by fish. However, rotenone can be used across a wider range of water qualities. Studies have suggested that aquatic invertebrates can re-colonize areas treated with piscicide within several months; however, taxa differ in their response, with some unaffected by treatment, and others very slow to recover (Mangum and Madrigal 1999). Where a stream contains a taxon of conservation importance, such as a Conservation or Core Conservation population of Rio Grande cutthroat trout or other native fish species, individuals can be protected from the effects of piscicide by removing them from the stream and maintaining suitable facilities until they can be re-introduced. Dunham et al. (2002) and DeMarais et al. (1993) note that careful consideration should be given to the potential impact of a piscicide treatment on such taxa.

Alternatives to the use of piscicides include gill-netting, de-watering, and electrofishing. Although gill-netting is unlikely to be of use in most suitable cutthroat trout habitat, in some cases, for example where the use of piscicides is precluded by the presence of sensitive native species, it may be a viable option for the removal of non-native trout from small high-elevation lakes (Knapp and Matthews 1998). Electrofishing can be an effective tool for removal of non-native trout from short sections of small water bodies with low habitat complexity (Kulp and Moore 2000, Shepard et al. 2003), but trout elimination requires intensive removal efforts over several years, and electrofishing is unlikely to completely eliminate non-natives where areas are larger and habitat is more complex.

In some cases, complete removal of non-native trout prior to re-introduction of Rio Grande cutthroat trout may not be necessary. The presence of rainbow trout or non-native cutthroat trout within a population of pure Rio Grande cutthroat trout is unacceptable, as even a single individual can compromise the genetic integrity of the population via interbreeding (Allendorf et al. 2004). Populations of Rio Grande cutthroat trout can, however, co-exist with low numbers of brook trout or brown trout. Numbers of brook trout or brown trout within cutthroat trout populations can be managed by selective electrofishing removal over multiple years (e.g., Thompson and Rahel 1996, Peterson et al. 2004a). Selective angling has also been proposed as a method to control populations of non-native trout within cutthroat trout habitat. However, Paul et al. (2003) suggest that this approach may be ineffective. Even when complete removal of non-native trout is achieved, subsequent population monitoring is important to ensure that re-invasion has not occurred.

Hilderbrand (2002) used a stage-based model to evaluate re-introduction strategies for restoration of cutthroat trout populations into streams where sufficient habitat is available to support a viable population. The model incorporated density-dependent mortality and environmental stochasticity that caused reproductive failure in approximately 5 percent of years, and it was based on survivorship and fecundity data collected for westslope cutthroat trout. A viable and persistent population was considered to be one with <5 percent chance of extinction within 100 years. Results demonstrated that re-introduced populations had a greater chance of persisting where fish were stocked for multiple years, where large numbers of fish were stocked, and where mature fish were included in the stocking. Success of translocations may also depend upon the source of fish used for stocking. Some habitats chosen for restoration may contain specific environmental conditions (e.g., cold water temperatures) to which the original native trout population was adapted; cutthroat trout transplanted from other habitats may be unable to survive or reproduce successfully under these conditions. Greenback cutthroat trout from the headwaters of the Little South Poudre River in Colorado, for example, appear to have adapted to cold water conditions by producing eggs that develop more quickly at lower temperatures than do eggs produced by fish from other populations (Behnke 2002). Similar habitat-specific adaptations may have arisen amongst Rio Grande cutthroat trout, although none have yet been documented. Additionally, several authors have suggested that transplants of hatchery-reared trout may fail because these fish are behaviorally, morphologically, or genetically adapted to the hatchery environment (e.g., Mesa 2001). The likelihood of a successful transplantation may be increased by maximizing the genetic diversity of the population introduced into the new habitat and by using wild fish or those from a hatchery program managed to minimize the effects of the hatchery environment. Again, if wild Rio Grande cutthroat trout populations are used as sources of fish for translocations, care must be taken to limit the impact of fish removal from these populations (U.S. Fish and Wildlife Service 2002). Where a restored area of habitat is immediately downstream from an existing Rio Grande cutthroat trout population, natural colonization may in some cases preclude the need for fish translocation.

Genetic testing

Genetic markers can be used to assess a number of individual and population-level characteristics including levels of introgression from non-native trout, levels of genetic diversity within a population, degree of population sub-structuring, and levels of differentiation between populations and drainages. Different types of genetic markers have different properties and are suited to address different questions. Earlier studies of cutthroat trout purity utilized electrophoretic analysis of allozymes (e.g., Palma and Yates 1994), which required sacrifice of fish. However, most genetic work nowadays requires only a small tissue sample such as a fin clip.

Assessment of levels of introgression ('admixture') within a Rio Grande cutthroat trout population using the formula agreed upon in the Utah Position Paper (Utah Division of Wildlife Resources 2000) requires markers that are inherited through both sexes, are co-dominant, and exhibit diagnostic differences between Rio Grande cutthroat trout and the taxa that are the expected sources of introgression (i.e., rainbow trout, Yellowstone cutthroat trout and Snake River cutthroat trout). Currently, only allozymes satisfy all these conditions (e.g., Leary 2001), but their use has fallen out of favor, partly because of the requirement for lethal sampling. Restriction Fragment Length Polymorphism (RFLP) analysis of mitochondrial DNA is also diagnostic between Rio Grande cutthroat trout, Yellowstone cutthroat trout, Snake River cutthroat trout, and rainbow trout (Martin et al. 2005); however since mitochondrial DNA is inherited through the maternal line only, it is unlikely to provide an accurate assessment of the degree of introgression at the nuclear genetic level. Paired interspersed nuclear element PCR analysis (PINE; Spruell et al. 2001, Kanda et al. 2002, Douglas and Douglas 2005) also provides diagnostic markers for these taxa. PINE markers are not co-dominant but can be used to provide a conservative estimate of introgression levels using the proposed formula (Utah Division of Wildlife Resources 2000); for example a sample with a calculated introgression level of '10%' using PINEs will have a true introgression level of between 5 and 10 percent. Work is ongoing to develop alternative diagnostic genetic markers (e.g., bi-allelic markers BiAMs; Ostberg and Rodriguez 2002) that will satisfy all the conditions of the Utah formula. The formula cannot be reliably applied when using non-diagnostic polymorphic markers such as microsatellites. Alternative approaches can be used to estimate levels of introgression using these type of markers, but results vary depending upon the method and reference sample used and therefore cannot easily be reconciled with the strict '<1% introgression' and '<10% introgression' cut off points currently used to place cutthroat trout populations into different management categories (Pritchard et al. submitted).

Analysis of variation in mitochondrial or nuclear DNA sequences is extremely well suited to identifying large scale patterns such as isolation of populations and drainages over relatively long evolutionary time scales (Hallerman 2003). In contrast, highly polymorphic, co-dominant nuclear markers such as microsatellites are ideal for investigating small-scale population genetic variables such as levels of genetic diversity within populations or broodstock lines of Rio Grande cutthroat trout and levels of differentiation between populations. However, results must be interpreted with care. For example, many population genetic statistics are based on assumptions, such as migration-drift equilibrium and Hardy-Weinburg equilibrium, that Rio Grande cutthroat trout populations are unable to satisfy in their current, fragmented condition. Even with such caveats, however, results from population genetic analyses can be extremely useful in combination with other data (e.g., ecological studies) to advise Rio Grande cutthroat trout management decisions.

When conducting any genetic study of a population, sampling methodology and sample size are extremely important considerations. For example, levels of non-native introgression within Rio Grande cutthroat trout populations may vary with distance from the migration barrier; in this case, a genetic sample taken from a single location may result in an estimated introgression level that does not accurately reflect the level in the population as a whole. Additionally, collection of genetic material from trout occurring in close proximity to one another may result in a genetic sample consisting mainly of closely related fish, which will produce highly misleading results (Hansen et al. 1997). Ideally, systematic sampling (e.g., every fifth fish) should be performed over a lengthy stream reach or multiple reaches. When tissue samples are collected for genetic analysis, recording data on fish size and position, even if this is approximate, can improve the interpretation of results. Sample size required for genetic analyses will depend upon the question of concern, the type and level of variability of the genetic marker, the level of confidence required (e.g., Ruzzante 1997), and in some cases the size of the population.

Information Needs

The current and historic distributions of Rio Grande cutthroat trout are understood in sufficient detail to formulate regional conservation strategies. Although some extant Rio Grande cutthroat trout populations may not yet have been documented, it appears that most have been identified, and the majority are the subject of previous, current, or planned assessment or protection activities. Information about the size, habitat condition, and genetic purity of some populations is lacking, but gathering this information is a priority for management agencies.

The subspecies' response to habitat changes is expected to reflect that of other stream-dwelling salmonids. The response of salmonids to habitat change has been relatively well studied. While different habitat factors may be limiting in different Rio Grande cutthroat trout populations, several factors that appear to be generally important include availability of deep pools and summer water temperatures. Although brook trout and brown trout appear to negatively impact Rio Grande cutthroat trout populations, the exact mechanisms remain unknown. Research is ongoing onto the impact of brook trout and brown trout on cutthroat trout populations and may suggest new management tools to control these impacts.

The annual, seasonal, and daily movement patterns of Rio Grande cutthroat trout are not well understood. Some portion of the subspecies may have previously expressed a migratory life-history strategy, as has been shown for other inland cutthroat trout subspecies; however the contemporary isolation of Rio Grande cutthroat trout to small headwater streams is expected to have eliminated expression of such a strategy. Collection of data on the frequency and extent of Rio Grande cutthroat trout movement within these small streams will provide a better understanding of the amount of habitat required to support a healthy population, the proportion of a population that is expected to be lost over a migration barrier, and the potential for natural re-colonization of available habitat.

The demography of Rio Grande cutthroat trout, in the habitat to which it is currently confined, is understood to some extent. Models have been developed to analyze probability of persistence at the local scale, both looking at habitat variables and demographic variables. However, much information is lacking, particularly regarding the breeding structure, sex ratio and social factors such as territoriality which may regulate population size. An improved understanding of such factors will improve, for example, our understanding of the relationship of N_e to census population size within this subspecies and therefore the general range of population size that may be required to minimize loss of genetic diversity and inbreeding.

Established methods are available to measure population abundance in stream dwelling fish. However, they exhibit some bias, and accurate whole-stream population estimates depend upon quantifying that bias and sampling a sufficient number of sites and habitats. In addition, cutthroat trout populations are expected to exhibit large inter-annual variations in abundance. Multiple years of monitoring may therefore be required to identify a true upward or downward trend in population abundance.

Reliable methods are available to restore degraded habitat and to artificially establish new populations. Models are available predicting habitat attributes and stocking strategies that are most likely to result in successful Rio Grande cutthroat trout translocations, and management agencies are experienced in creating new populations in this way. Rather than a lack of information, the main obstacles to the conservation and expansion of Rio Grande cutthroat trout are primarily social, political, logistical, and economic and include local opposition to piscicide use and angler demand for non-native trout species (Quist and Hubert 2004). Educational programs and provision of sportfish populations of Rio Grande cutthroat trout are two approaches currently being implemented by management agencies that may improve public support for preservation of the subspecies.

There are several pieces of information that would be useful for managers attempting to formulate effective conservation policies for Rio Grande cutthroat trout, but that current scientific knowledge is unable to address for any taxon. It is impossible to predict with any accuracy, for example, the likelihood that outbreeding depression will occur when fish from one population are able to breed with those from another population. Decisions such as when to supplement one Rio Grande cutthroat trout population with another, or whether to connect a pure populations with a slightly introgressed population will need to be made on a case-by-case basis taking into account all available knowledge, including geographical and genetic distance between populations, historical population connectivity, logistical and economic considerations, and the need to retain native genetic diversity. Finally, successful conservation of the Rio Grande cutthroat trout in USFS Region 2 requires information exchange and co-operation between the management agencies involved.

DEFINITIONS

Allele: One of the different forms of a **gene** or **genetic marker** that can exist at a single **locus**. Diploid organisms, such as most vertebrates, have two alleles at each locus.

Allopatric: having non-overlapping ranges

Allozyme: Form of an enzyme that differs in amino acid sequence from other forms of the same enzyme and is encoded by one allele at a single locus. Different forms of allozymes can be distinguished by **electrophoresis**, a process by which molecules can be separated according to size and electrical charge by applying an electric current to them.

Anchor ice: submerged ice attached to the stream bottom.

Anthropogenic: caused by humans.

Basibranchial teeth: the teeth on the basibranchial bone, behind the tongue and between the gills.

Benthic: occurring at the bottom of a body of water.

Carrying capacity: the maximum number of individuals that a habitat can support over a given time period.

Co-dominant genetic markers: genetic markers for which both **alleles** present at a **locus** can be identified.

Deleterious allele: an **allele** that determines a characteristic that reduces the fitness of an individual possessing it.

Demographic: pertaining to the study of population statistics, changes, and trends based on various measures of fertility, mortality and migration.

Demographic stochasticity: random variation in life-history characteristics such as sex ratio, birth rate, death rate and reproductive success.

Effective population size (N_e): number of breeding adults in an ideal population that would have the same observed temporal variation in gene frequencies as the population under study. Commonly N_e is much smaller than the actual number of adults observed in a population.

Electrofishing: capture of fish by passing an electric current through the water in order to immobilize them.

Enviromental stochasticity: random changes in environmental conditions.

Extinction: the loss of a taxon over its entire range.

Extirpation: the loss of a taxon from a portion of its range.

Fingerling: fish in its first or second year of life but older than the fry stage.

Fluvial: of, relating to, or inhabiting a river or stream.

F_{st}: a measure of the level of genetic differentiation between populations, which varies between 0 and 1.

Gametes: sperm and eggs.

Gene: a sequence of DNA that occupies a specific location on a chromosome, and determines a particular characteristic in an organism.

Generalist: able to exploit a variety of resources such as diverse prey items or habitats.

Genetic drift: random changes in the frequencies of **alleles** due to **demographic stochasticity**. Genetic drift is more pronounced in smaller populations.

Genetic marker: in the context of this document, a sequence of DNA occupying a specific location on a chromosome that can be used to address a population genetic question.

Genetic purity: in a cutthroat trout individual or population, the amount of genetic material that derives from the native **taxon**. An individual is considered 'genetically pure' if none of its genetic material is derived from introduced taxa.

Genetic variation: Genetic diversity in an individual, population or taxon. The greater the number of different **alleles**, the greater the genetic variation.

Gular fold: fold under the mandible that exhibits red coloration on the throat of cutthroat trout.

Hardy-Weinburg equilibrium: a population is in Hardy-Weinburg equilibrium when the observed frequency of heterozygote and homozygote individuals corresponds to that expected under the assumptions of infinite population size, random mating, no mutation, no migration and no selection.

Heterozygosity: the condition of having two different **alleles** at a **locus**. The more **heterozygous** an individual or population, the greater the number of different **alleles** it contains.

Homozygosity: the condition of having the same two **alleles** at a **locus**. The more **homozygous** an individual or population, the fewer the number of different **alleles** it contains.

Inbreeding depression: decrease in fitness as a result of increased **homozygosity**, which may occur due to matings between close relatives or to a reduction in the **genetic diversity** of a population.

Introgression: movement of genetic material from one **taxon** or population into another, generally via hybridization.

Lateral series: number of scales along the **lateral line**, which is a series of pores along the side of a fish.

Locus (s), **loci** (pl)**:** a portion of a chromosome containing a **gene** or **genetic marker**.

Macroinvertebrate: larger invertebrate.

Meristic: relating to the number or placement of body parts.

Metapopulation: in the context of this document, a set of populations between which individuals are able to migrate.

Mitochondrial: contained in the mitochondria, structures in the cell that are typically inherited from the mother only.

Nuclear: contained in the nucleus of a cell.

Obligate host: a host which a parasite requires in order to complete its life cycle.

Phenotype: the observable physical or biochemical characteristics of an organism, as determined by both genetic makeup and environmental influences.

Piscicide: a fish poison.

Piscivory: feeding on fish.

Pluvial lake: a lake that formed from rainwater falling into a landlocked basin during a glacial period.

Polymorphic: including many forms, for example many different **alleles**.

Polytypic: including many different types.

Population extinction: the complete loss of a population.

Population bottleneck: an event in which a significant portion of a population is lost or otherwise prevented from reproducing for a period of time.

Pyloric caecae: finger-like extensions from the gut at level where it contacts stomach.

Recessive allele: an allele of a **gene** that does not have an effect on the **phenotype** when a dominant allele is also present, or an allele of a **genetic marker** that cannot be identified when a dominant allele is present.

Riparian: relating to or living or located on the bank of a natural watercourse, a transitional zone between the aquatic and terrestrial habitats.

Salmonid: a member of the family Salmonidae, which includes salmon, trout and whitefish.

Scale annuli: growth rings on fish scales.

Sedentary: moving little.

Stream capture: a phenomenon which occurs when a stream from a neighboring drainage system erodes through the divide between two streams and "captures" another stream, which then is diverted from its former bed and now flows down the bed of the capturing stream.

Taxon (s.), **taxa** (pl.)**:** a taxonomic group of any rank, for example genus, species or subspecies.

Translocation: in the context of this document, **anthropogenic** movement of a fish to another location.

Vagility: amount of movement.

Young-of-the-year: fish hatched in a given calendar year (age 0).

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