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ORIGINAL PAPER

Multiple electrofishing as a mitigate tool for removing nonnative Atlantic brown trout (*Salmo trutta* L.) threatening a native Mediterranean brown trout population

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Abstract In the native range of the brown trout (*Salmo trutta* L.) in Europe, the hybridization of native populations by nonnative domesticated strains introduced by stocking is one of the most serious threats to the long-term conservation of diversity within this species. With the objective of conserving and restoring the native gene pool, fishery managers are beginning to implement various management strategies at the local scale. Nevertheless, few case studies have been published that investigate the effectiveness of the various different conservation strategies for native brown trout populations. In the Chevenne Creek, a small French mountain stream, we tested the strategy of removing nonnative individuals by multiple electrofishing carried out by fishery managers in order to evaluate its feasibility and effectiveness for eliminating a nonnative population threatening a native population. Electrofishing produced major reductions in the nonnative population between 2006 and 2009, with 82–100% of nonnative individuals being removed over a period of 4 years. Nevertheless, despite multiple-electrofishing campaigns, this nonnative population was not entirely eradicated, and some natural recruitment persisted. The young of the year and subadults were less effectively removed than the adults. The results suggest that repeated electrofishing campaigns can be used by managers to reduce the nonnative brown trout population with the objective of conserving the native gene pool, but the removal operation must be continued for at least 4 consecutive years. This strategy, which is feasible in small streams, has to be followed by complementary operations to allow the restoration of a new, native, self-sustainable brown trout population.

Keywords Brown trout · Conservation · Electrofishing · Nonnative trout removal

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Introduction

The brown trout (*Salmo trutta* L.) displays a high level of genetic diversity in its native range in Europe (Bernatchez 2001; Cortey et al. 2004). In France, both Mediterranean (MED) and Atlantic (ATL) brown trout populations show a clear genetic distinction at mitochondrial and nuclear genes (Guyomard 1989; Bernatchez et al. 1992; Launey et al. 2003). In the French north alpine hydrographic catchment area, which is part of the Mediterranean basin, MED populations of brown trout are considered to be native (Guyomard 1989; Launey et al. 2003). For more than a century, stocking practices implemented by fishery managers have led to massive introductions of domesticated ATL trout (Krieg and Guyomard 1985; Launey et al. 2003) into rivers inhabited by native MED populations and consequently to hybridization and to the decline of native MED populations in this area (Guyomard 1989; Largiadèr et al. 1996).

Similar negative genetic effects of stocking with a domesticated ATL strain on native MED populations has been reported in several streams in the Mediterranean basin in France (Barbat-Leterrier et al. 1989; Beaudou et al. 1994; Poteaux et al. 1998; Berrebi et al. 2000), in Spain (Aparicio et al. 2005; Almodóvar et al. 2006), and in Italy (Caputo et al. 2004; Lucentini et al. 2006).

Throughout the native range of the brown trout in Europe, the hybridization of the native populations by domesticated ATL strains is considered to be one of the most serious threats to the long-term conservation of diversity within this species (Laikre et al. 1999; Ferguson 2006).

Over the last 10 years, in relation to the multiplication of genetic case studies, fishery administrators have become aware of the importance of conserving native populations, and fishery managers have begun to take intraspecies genetic diversity into account in their management programs. Gradually, alternative management strategies are being implemented with the aim of conserving and restoring the native gene pool at the population level. The tools that can be used for the conservation and restoration of stream salmonids are (i) genetic refuges (Guyomard 1989; Allendorf et al. 1997; Araguas et al. 2009), (ii) the elimination of nonnative fish by chemical methods (Rinne et al. 1981; Harig et al. 2000) or their removal by electrofishing (Moore et al. 1986; Kulp and Moore 2000; Peterson et al. 2004), (iii) the deliberate isolation of threatened native individuals (Novinger and Rahel 2003; Van Houdt et al. 2005), (iv) stocking with local native breeding stock (Cowx 1994; Waples 1999; Crivelli et al. 2000; Young and Harig 2001; Caudron et al. 2006), (v) translocation of wild individuals (Hilderbrand 2002; Schmetterling 2003; Caudron et al. 2009), and (vi) selective angling (Mezzera and Largiadèr 2001).

Nevertheless, fishery managers who are also conservationists are often confronted by problems related to the spatial distribution of both native and non-native individuals. For instance, they can be confronted with an allopatric spatial distribution, in which nonnative individuals are located downstream and/or upstream of a river stretch inhabited by a native population, or by a sympatric distribution, in which nonnative and native individuals are found together in the same stream. It is not easy for wildlife managers and fishery managers to select the most appropriate conservation method adapted at each situation. Moreover, fishery managers need to balance two different objectives, sometimes antagonist, the angling activities and the conservation of native populations.

In the northern French Alps area, joint studies involving both scientists and fishery managers have made it possible to monitor several restoration strategies for native MED brown trout populations carried out by managers and to evaluate their effectiveness. In the Chevenne Creek, a small mountain fragmented stream, both ATL nonnative and MED native populations show an allopatric spatial repartition with an ATL population in the isolated upper part of the river and a MED population located immediately downstream (Barnetta 2005). In this situation, where the native gene pool of the MED population is threatened by ATL gene flow from upstream, managers decided to carry out several restoration strategies. Since 1993, the genetic refuge banning stocking with ATL trout was established and the fishing activity has been prohibited since 1998. In 2006, because the ATL population persisted despite 13 years without stocking, managers decided to carry out the strategy of removing nonnative individuals by repeated electrofishing. This choice was motivated by the allopatric distribution of both ATL and MED populations and the isolated location of the ATL trout in a short upstream section which offered an opportunity to evaluate in situ this practice. The goal of this action, in addition of the genetic refuge strategy, was to stop the source of alien gene flow from upstream, which led the introgression of the native population located downstream.

The present publication reports data on the still poorly investigated use of electrofishing removal of brown trout. The aims of our study were (i) to monitor a case study of this strategy implemented by fishery managers, (ii) to evaluate its feasibility and effectiveness, and (iii) on the basis of these results obtained for brown trout, together with published results concerning other trout species, to discuss possible in situ applications of this tool for the brown trout in its native range in Europe.

Study area

The Chevenne Creek, a typical northern French Alpine mountain stream, is a first-order tributary of the Dranse d'Abondance river, a tributary of the Dranses system, which is the second largest affluent of Lake Geneva and belongs to the Mediterranean catchment (Fig. 1). The Chevenne stream is 2.5 km in length and 1–4 m in width; it ranges in altitude from 1,250 to 1,000 m and has a mean slope of 10%.

It has the typical geomorphologic characteristics of a middle-altitude mountain stream in the Alpine zone, with a complex and fragmented habitat dominated by riffles, cascades, and pools, containing many boulders and small woody debris. The mean conductivity is 360 μS/cm, and the average pH value is 8.2.

Several genetic studies (Bernatchez et al. 1992; Largiadèr and Scholl 1996; Largiadèr et al. 1996; Launey et al. 2003) have shown that the main stream of the Dranse d'Abondance and the downstream and median parts of the Chevenne Creek harbor a nearly pure MED populations despite intensive stocking with domesticated ATL trout over several decades (Fig. 1). To preserve the native MED gene pool, fishery managers set up a genetic refuge where stocking was stopped since 1993, and angling was forbidden since 1998.

Genetic analysis of samples between 1995 and 2003 showed that, after a period of 10 years without stocking

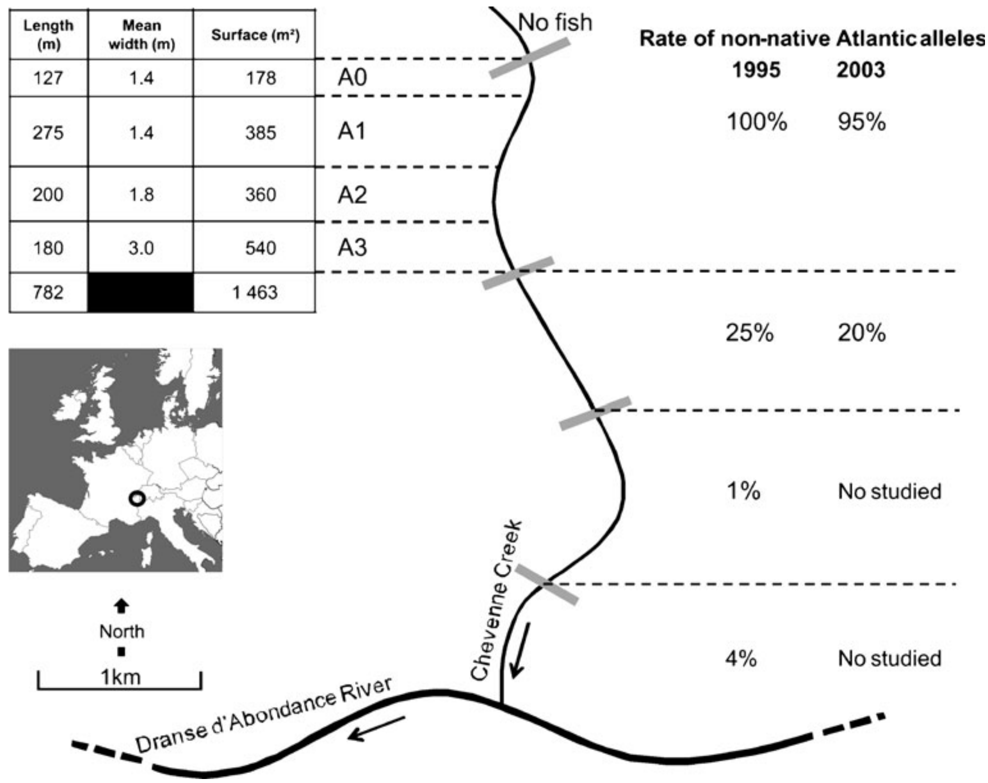

Fig. 1 Location and characteristics of the study area. Longitudinal gradient of admixture rate between nonnative Atlantic and native Mediterranean brown trout along the Chevenne Creek (according to Barnetta 2005). *Grey bar*: Waterfall

and 5 years without angling pressure, the pattern of admixture between ATL and MED trout had not changed significantly (from 25 to 20%), and that recent admixtures were still occurring in the lower section of the upstream part of the Chevenne stream (Barnetta 2005; Largiadèr and Champigneulle unpublished data). This mixing can be explained by the downstream migration of ATL trout from the isolated upper section of the upstream part of the Chevenne, which harbors a self-sustained ATL population, and where natural recruitment occurs. There is some evidence that this upstream ATL population was originally introduced by stocking: i) the stocking history indicates that intensive stocking had taken place in the upstream part of the Chevenne and ii) a genetic analysis indicated that there was no detectible genetic difference between this upstream ATL population, and the hatchery strain that was used for stocking the Chevenne (Estoup et al. 2000). In this situation, managers decided to remove ATL trout by electrofishing to avoid any possibility of introgressive hybridization for the native population located downstream.

Materials and methods

In the autumns of 2006, 2007, 2008, and 2009, repeated electrofishing campaigns were conducted by fishery managers in the upper section of the upstream part of the Chevenne with the aim of removing any non-native ATL individuals present. This upper section, which is 780 m in length, was divided in four sectors designated A3, A2, A1 and A0 from downstream to upstream (Fig. 1).

Each year, the electrofishing operations were carried out using the same battery-powered, portable, backpack electroshocker (Martin Pecheur II, Dream electronique, Saint-Germain-du-Puch, France; power source: cadmium–nickel battery 12 V, 10 A; currents produced: rectangular pulsed DC 400 Hz, voltage 300 V, power about 200 W; anode diameter 30 cm).

The electrofishing was conducted by a four-person crew; with one person doing the actual electrofishing, and two others netting fish followed by the fourth person with a bucket. After each electrofishing pass, a fifth person recorded the length (to the nearest mm) and the weight (±0.1 g) of each trout collected, while the electrofishing crew continued with their task.

Each year, two electrofishing passes were realized in each section. When the electrofishing passes were finished, all the trout that had been caught were removed from the study area.

Population density (individuals/100 m²) and biomass (kg/ha) estimates and 95% confidence intervals were calculated for each sector and for the entire stream using the maximum likelihood population estimate of Carle and Strub (1978), as recommended by Gerdeaux (1987). The following condition stipulated by Seber and Lecren (1967) has been checked prior to these estimations: $[C1^2*(C1-C2)^2]/[C2^2*(C1+C2)]>16$, where C1 and C2 represent the number of trout collected during the first pass and the second pass, respectively.

The estimates were calculated separately for three different length classes that corresponded to young of the year (YOY, 40–89 mm), subadults (90–159 mm), and the adult spawners (≥160 mm), respectively. The correspondences between size classes and sexual maturation used in this study had been validated by a previous study in the Chevenne population (Champigneulle et al. 2003).

The effectiveness of the removal by electrofishing was determined by the ratio of the number of trout caught, NC (sum of the two passes), divided by the estimated total number of trout present, NE, given by the Carle and Strub (1978) formula. Confidence limits of 95% were also given using the Carle and Strub (1978) formula. The time taken to perform these removal operations each year was also documented.

Results

Removal of nonnative brown trout

The population data of 2006 showed that a large and viable population of nonnative ATL brown tout was still present in the upstream section of the Chevenne Creek after 13 years without stocking. This population had a density of 33.2 (±2.2) individuals/100 m² and a biomass of 148 (±18) kg/ha (Fig. 2).

In all, 893 ATL brown trout were removed from the upper part of the Chevenne by the four removal campaigns. This operation reduced the total density by a factor of 4.2, and the total biomass by a factor of 8.5, with 7.9 (±0.8) individuals/100 m² and 17.4 (±2.5) kg/ha in 2009 after four removal campaigns. During the first three removal campaigns, the total density fell each year by a factor of about 2, but for the third and the fourth years, the densities were similar. The total biomass decreased markedly after the first and fourth removals, but did not change between the second and the third years. The spatial decrease patterns of both density and biomass were showed some differences between the four sections (Fig. 2).

Of the 893 trout removed, 395 (44%) were age 0 (<90 mm), 228 (26%) were adults (≥160 mm), and the rest, 270 (30%), were in the intermediate size class (90–159 mm).

The removal efficiencies, based on the population depletion data obtained each year, were lower for the first year than for the next 3 years for all three categories: YOY, subadults, and adults (Table 1).

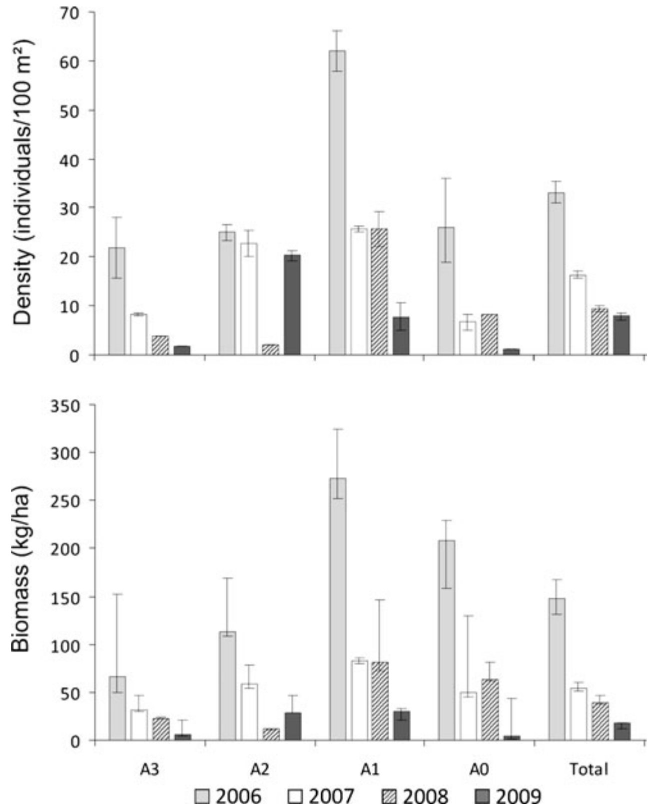

Fig. 2 Density and biomass of nonnative brown trout obtained by maximum likelihood population estimation in the four sections and in the entire stretch for 2006–2009. Bars show 95% confidence intervals

Overall, the effectiveness of the removal of adults was high (from 96% to 100%), which was higher than that of YOY (from 71% to 100%) or subadults (from 50% to 100%). The estimated removals of the YOY and subadult classes were similar (Table 1).

The estimated removal of YOY was greater in 2007 and 2009 than in 2006 and 2008 (Table 1). In the latter years, the size class distributions showed that the fish age-0 were smaller than in 2007 and 2009, and this may therefore be why they were less effectively caught by electrofishing (Fig. 3). The densities of YOY decreased greatly after the first removal, falling from 17.6 to 4.9 individuals/100 m², but after the two next removals, they remained nearly unchanged, indicating the difficulty of collecting age 0 trout.

For the subadult class, the densities were similar for the two first removals (7.7 and 8.7 trout/100 m², respectively) and decreased only in the third year, with 2.5 trout/100 m². The densities of adults fell from 9.6 to 2.7 individuals/100 m² as a result of the first removal and from 2.7 to 0.8 trout/100 m² as a result of the third removal (Fig. 3).

All these findings indicate that the nonnative ATL individuals and also their natural recruitment had not been entirely caught and removed after 4 years of electrofishing.

Table 1 Removal efficiency of nonnative brown trout obtained each year for the three size classes: young of the year (YOY), subadults, and adult spawners

	YOY (<90 mm)			Intermediate class (90–159 mm)			Adults spawners (≥160 mm)		
	Nc	Ne (95%CI)	Depletion estimates (%)	Nc	Ne (95%CI)	Depletion estimates (%)	Nc	Ne (95%CI)	Depletion estimates (%)
2006	206	258 (226–290)	71–91	85	113 (100–126)	67–85	139	141 (139–144)	96–100
2007	67	71 (67–77)	87–100	122	127 (122–134)	91–100	39	39 (39–36)	100
2008	55	63 (55–74)	74–100	36	36 (36–36)	100	39	39 (39–39)	100
2009	67	69 (67–73)	92–100	27	37 (27–54)	50–100	11	11 (11–11)	100

Nc Number caught, *Ne* number estimated with 95% confidence intervals in parentheses

According to the depletion estimates obtained in 2009, the maximum number of trout still present after the last electrofishing campaign can be estimated to be 33, with 6 YOY and 27 subadults.

Field time

The total time required for a two-pass electrofishing campaign per year was 212 h (26.5 man days), spread over 4 years. The number of hours per 100 m² of stream, ranged from 5.5 to 2.2, decreasing from 2006 to 2009. If we assume a mean rate of 200€ per people per day, the total cost of the operation carried out in the Chevenne over a period of 4 years can be estimated at about 28,000€ per km or 1,500€ per 100 m² of stream treated.

Discussion

Failure of the genetic refuge strategy implemented alone

The initial population data of 2006 revealed the presence of a self-sustaining nonnative ATL brown trout population with high density and biomass despite a stocking-free genetic refuge that had been established for 13 years. Our results indicate that this "no stocking" strategy failed to reduce the nonnative ATL populations in the upper part of this stream, which threatened a native MED brown trout population downstream (Barnetta 2005). These results are consistent with those obtained for MED brown trout in several other rivers in the Mediterranean catchment area. Indeed, Poteaux et al. (1998) showed in a southern French river that this strategy was not efficient to collapse the introgression of the MED native gene pool by domesticated ATL strain. Araguas et al. (2008, 2009) in MED populations in the Eastern Pyrenees did not detect genetic differences in the samples before and after the establishment of genetic refuge. Caudron et al. (2010) showed similar results in a mountain stream of the northern French Alps.

All those results suggest that although setting up a stocking-free genetic refuge area is indeed a first essential strategy for preserving the native gene pool, it is not sufficient by itself, and other additional measures are necessary to restore the native gene pools, and avoid nonnative introgression from strain populations established in some river stretches.

In the Chevenne stream, angling has been prohibited since 1998. The absence of angling pressure may have allowed the ATL nonnative population derived from hatchery released trout to become better established. Indeed, several studies in Europe (Garcia-Marin et al. 1998; Mezzera and Largiadèr 2001; Champigneulle and Cachera 2003) have suggested that nonnative ATL brown trout are more susceptible to capture by anglers than the native MED brown trout. Similar results have also been obtained for the rainbow trout (*Oncorhynchus mykiss*) in USA (Dwyer 1990). Thus, selective angling pressure in the genetic refuge area could offer a complementary tool to limit the expansion of nonnative trout. However several studies have demonstrated that this method does not seem to be able to prevent the collapse of nonnative populations. Indeed, Larson et al. (1986) showed that angling pressure reduced the nonnative rainbow trout population in an Appalachian stream by only about 10%. Other studies, in the Rocky mountain creeks in Alberta (Paul et al. 2003; Stelfox et al. 2004), have demonstrated that nonnative brook trout (*Salvelinus fontinalis*) populations were highly resilient to overexploitation, and so selective angling exploitation was not an effective way to eliminate nonnative trout populations.

In the case of the Chevenne, fishery managers decided to assess an active method in order to stop the source of the ATL gene flow threatening the MED native population. The strategy of removing nonnative individuals by electrofishing has been chosen because the allopatric distribution of these two populations, and the isolated location of the nonnative ATL trout in a short upstream section offered a unique opportunity to evaluate this practice. Moreover, the chemical treatment with pesticides (rotenone or antimycin

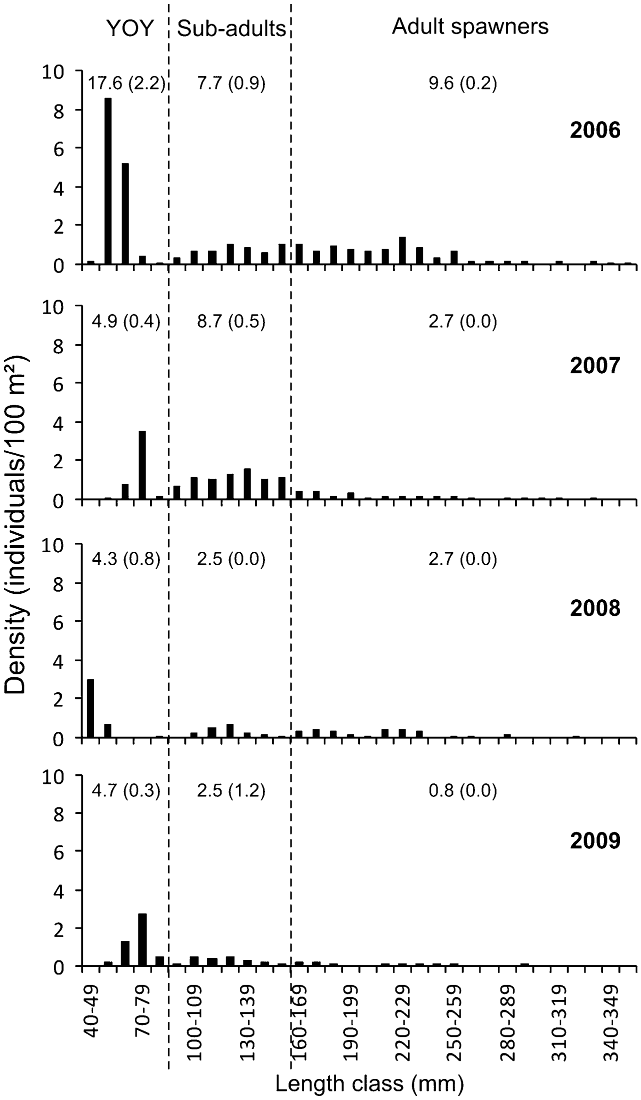

Fig. 3 Size distribution of nonnative brown trout captured in the stretch studied during 2006–2009. For each year, the density (±SD) of trout is shown for the three size classes, young of the year (*YOY*), subadults, and adult spawner

A), although it is more efficient, less time consuming and cheaper than multiple electrofishing (Moore et al. 2005), was not used because the eradication of aquatic organisms using poison is banned by the French legislation.

Mitigate efficiency of the electrofishing removal to eliminate the introduced brown trout

No comparisons are possible for the brown trout as, as far as we are aware, no similar removal experiments have been published regarding the problem of two different origins (ATL/MED) of this species. Indeed, most studies concerning this approach have been conducted in the western United States in order to reduce or eliminate (i) nonnative rainbow trout in the native range of the brook trout (Moore and Larson 1983; Moore et al. 1986, Larson et al. 1986; Kulp and Moore 2000; Moore et al. 2005) or (ii) nonnative brook trout in the native range of the cutthroat trout, *Oncorhynchus clarkii* (Thompson and Rahel 1996; Shepard et al. 2002; Peterson et al. 2004).

In the Chevenne stream, electrofishing removal made it possible to reduce both the biomass and the density of the ATL nonnative population considerably between 2006 and 2009. On average 82–100% of nonnative ATL individuals were removed over a 4-year period. Nevertheless, despite the multiple-electrofishing campaigns, this nonnative population was not wholly eradicated and some natural recruitment persisted.

Our findings for brown trout species are similar to the results obtained in small streams for other nonnative trout species (rainbow trout and brook trout). Thompson and Rahel (1996) indicated a success rate of 59–100% for age 1 and older brook trout for single, three-pass electrofishing removal efforts. Shepard et al. (2002) found that 8 years were necessary for the total removal of brook trout. The effective removal of nonnative trout also required intensive electrofishing over some years (Moore and Larson 1983; Moore et al. 1986; Kulp and Moore 2000). All the studies published have reported lower removal efficiencies for age-0 than for older trout. In the present study, the efficiencies of YOY removal ranged from 71–91% to 91–100% (based on depletion estimates), and were lower than those for adults (≥160 mm). Furthermore, the removal efficiencies were lower in 2006 and 2008, when the YOY were smaller, than in 2007 and 2009. Size selectivity is a well-known phenomenon in electrofishing surveys in several species of fish (Junge and Libosvarsky 1965; Reynolds 1989; Dolan and Miranda 2003), and it must be allowed if we are to increase the success of restoration by means of electrofishing removal. In the present experiment, we did not find any difference for the removal efficiencies in YOY (<90 mm) and subadults (90–159 mm). The difficulty of catching this size class in the Chevenne could be explained by the presence of complex habitat with several deep pools (up to 1.5 m deep), unclogged substrates, boulders and woody debris. Indeed, the morphology and characteristics of the stream, in particular habitat complexity, stream cover and deep water all influence the efficiency of electrofishing for removal purposes (Moore et al. 1986; Riley and Fausch 1992; Thompson and Rahel 1996; Kulp and Moore 2000; Peterson et al. 2008).

According to all these studies, at least four consecutive years of repeated removals are necessary to greatly reduce or eliminate nonnative trout. Meyer et al. (2006) reported an unsuccessful operation of brook trout electrofishing removal over 2 years in a stream of more than 8 km of length. Peterson et al. (2008), by simulations, showed that even if total eradication was not possible, the maintenance control of brook trout by electrofishing can help maintain native cutthroat trout populations, but to be effective, multiple consecutive years of suppression over at least 3 years without interruption were necessary.

In the present study, single two-pass electrofishing removal campaigns per year for 4 consecutive years did effectively reduce the number of nonnative ATL brown trout but were not sufficient to eliminate them completely. Indeed, in the Chevenne, the latest population estimates indicate that 33 non-spawner trout (<160 mm) had been missed during the 2009 campaign. According to the age–maturity relationship and the growth in this area (Champigneulle et al. 2003; Caudron and Champigneulle 2006), we can assume that all adult female spawners had been removed from the study area, so that no reproduction can have occurred during the 2009–2010 spawning period. According to Kulp and Moore (2000), once natural recruitment has been eliminated, the complete elimination of rainbow trout followed within 1–2 years. This means that the elimination of nonnative ATL brown trout from the Chevenne is proceeding well, and that additional removal campaigns should be carried out in 2010 and 2011 in order to catch the last remaining nonnative ATL individuals.

Test of additional strategy to restore native brown trout populations in the Chevenne Creek

The present results showed that two coupled methods over a total period of 16 years, genetic refuge and electrofishing removals, were not sufficient to eliminate the nonnative population in the uppermost part of the Chevenne Creek. Thus, managers decided to test a third successive intervention: the translocation of native trout in the upper part of the river. The goal of this direct translocation was to complement the electrofishing removals in order to replace the upstream nonnative ATL population. This newly founded native population precludes the downstream alien gene flow and should mitigate the introgression rate in the transition zone. Only MED individuals from the downstream part of the Chevenne stream (section showing only 1% of ATL introgression) were involved in the translocation carried out in autumn 2009, once 4 consecutive years of removal seriously compromised the demography in the ATL population (less than 2.2 individuals/100 m^2). This approach was in agreement with results of two other field experiments (Caudron et al. 2010; Caudron unpublished data) showing that when the number of nonnative individuals was low in a stretch of river, the translocation of wild fish could be efficient to install new nearly pure native populations. It was primordial to transfer a sufficient number of individuals, first to reflect the genetic composition of the source population (Stockwell et al. 1996), and second to occupy at least 10% of the carrying capacity of the receiving site (Hilderbrand 2002). In the present case, 105 MED brown trout of different size classes were transferred, corresponding to about 20% of the initial carrying capacity of the stream studied (according to the population data in 2006). The transfer of at least 100 individuals with a wide range of age classes would increase the probability of establishment following an introduction operation (Minckley 1995; Fisher and Lindenmayer 2000). The dynamics of the colonization of the upper part and the change over time of the ATL introgression into the native MED population located further downstream will be monitored over time to evaluate the effectiveness of these combined strategies in restoring the native MED gene pool of brown trout in the whole Chevenne Creek. In addition, electrofishing operations will continue in 2010 to remove the remnant ATL trout. Indeed, it is possible to easily distinguish both ATL and MED trout by external characters for the conservation of native MED brown trout populations (Mezzera et al. 1997; Aparicio et al. 2005).

Guidelines to use electrofishing removals in recovery brown trout populations

Our study suggests that multiple-electrofishing removal campaigns can be used to eradicate nonnative brown trout populations established in the wild. Nevertheless, it is a costly and laborious method and some recommendations can be suggested.

First, due to the limited effectiveness of electrofishing, successive removal campaigns have to be conducted. We recommend to practice successive electrofishing for at least 4–5 years, with one or, if it is possible, two two-pass electrofishing operations, one in summer and one in autumn.

Secondly, this strategy is usable only for small streams, ideally in a length no more than 3 km and showing no complex habitat as deep pool or stream cover.

Thirdly, in contrast to the recommendation of Kulp and Moore (2000) for rainbow trout, we found that early autumn/fall (September/October) seems to be better for electrofishing to remove brown trout, especially in a mountain stream where growth is slow. Indeed, according to the brown trout lifecycle, adults generally spawn in November and December, and fry emerge between May and June. Thus, electrofishing should be conducted in early autumn/fall, when the YOY are large enough (>40 mm) to be immobilized by electrofishing gear so that their catchability is greater and when spawners have not yet spawned.

Finally, the electrofishing removal method can be used both when nonnative and native individuals are distributed in an allopatric manner, but it can also be used in a sympatric situation when it is easy to distinguish visually between fish of the different origins. For instance, this is the case for several native lineages of brown trout in Europe which display morphologic and phenotype differences such as those between the ATL and MED lineages (Mezzera et al. 1997; Aparicio et al. 2005), between the Danubian and Marmoratus lineages and those between the ATL and Marmoratus lineages (Delling et al. 2000). This removal method can also be used in a context of the invasion of native brown trout populations by introduced brook trout, as has recently been reported (Cucherousset et al. 2008; Korsu et al. 2009).

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