
The effects of 1080 on invertebrate communities and fish in West Coast streams

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The effects of 1080 on invertebrate communities and fish in West Coast streams

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Executive Summary

1. The fate and ecological effects of 1080 baits that fall into streams during aerial operations were investigated by:
 - surveying streams flowing through areas where 1080 was applied aerially to quantify the number of baits in streams;
 - examining the fate of 1080 baits that fall into streams, and the rate that 1080 leaches from baits;
 - quantifying the toxic effect of 1080 baits on freshwater fish and invertebrate communities in small streams.
2. A total of 48 streams were surveyed in four areas where 1080 was applied aerially to quantify the number of baits in 100 m stream sections. The number of baits found varied widely, and was related only to bait size. There was no relationship between stream width and the number of baits within a stream. These findings suggest that the potential number of baits falling into a stream cannot be calculated from the bait application rate and the stream size.
3. The fate of baits in moving water was assessed by laboratory experiments, as was the rate that 1080 leached from baits. Weight loss of baits was relatively rapid, with most of the baits fragmenting after 3-4 days. 1080 also rapidly leached from the baits: almost 50% of the initial 1080 contained within a bait had leached after 2 h, and >90% after 24 h. Such rapid leaching is most likely due to the high solubility of 1080. This finding has implications for water quality monitoring programmes that commonly sample water 24 h after an operation. We suggest that sampling programmes collect water within 8-12 h of potential contamination to detect presence of 1080.
4. The toxic effect of 1080 baits on invertebrate and fish was examined in 4 streams in the Mawhera State Forest in the Grey Valley using a **Before-After, Control-Impact (BACI)** experimental design. All 4 streams were < 3 m wide and had discharges ranging from 6 – 238 l s⁻¹ during the course of the study. The BACI design involved examining fish and invertebrate communities at 4 sites up a 400 m section of each stream. Two sites were located in an impact area 10 m and 100 m below where 1080 baits were experimentally placed, and two sites were located in a control area 10 m and 100 m above this. These sites were monitored 4 days and 1 day before the baits were added to each stream, and then 1 day and 4 days after. Separate experiments were conducted for the fish and invertebrate experiments.

5. Three species of native fish (longfin eels (*Anguilla dieffenbachii*), koaro (*Galaxias brevipenis*) and upland bullies (*Gobiomorphus breviceps*) were chosen for this experiment, conducted in March 2004. Eight fish of each species were placed into separate fish cages at the 4 sites in each stream. Fish survival in the cages was high at all sites except one, where significant mortality of upland bullies occurred between the first and second sampling occasions. This mortality reflected the effects of high river flows between these time periods where high instream velocities overturned one cage. No fish mortality was observed in any of the fish species after the 1080 baits were added, suggesting that these species were tolerant to 1080 that leached from the baits.
6. Analysis of water samples collected during the fish experiment showed that 1080 was detected in the water for only a short duration. Concentrations were all low, and below Ministry of Health's guidelines of $2 \mu\text{g l}^{-1}$, despite the very large quantities of baits that were placed in a small area in each stream. 1080 concentrations 10 m below the baits were higher than those 100 m below, reflecting the greater dilution experienced at the lower sites.
7. The second experiment quantified the effect of 1080 on natural invertebrate communities. Invertebrates were collected from the 4 sampling sites by sampling ten replicate rocks at each site. A total of 72 taxa were collected during the study, which was conducted in May 2004. The caddisflies *Helicopsyche*, *Pycnocentroides* and *Pycnocentria*, orthoclad midges, and the leptophlebid mayfly *Deleatidium* dominated the fauna. These animals are indicative of streams with good water quality. Statistical analyses showed that there were no ecologically significant effects of the 1080 baits on the invertebrate communities in any of the streams.
8. Our results indicated that there was no apparent effect of submerged 1080 baits to either invertebrates or native fish. Concentrations of 1080 in stream water were very low, even when very large numbers of baits were experimentally placed in the small streams selected for the study. These streams were all < 3 m wide, and so would not normally have buffers placed around them. As we found no effect of adding 1080 baits to such small streams, the value of maintaining buffer zones around streams >3 m is questionable, except when the streams are used as water supply for domestic or stock uses. Such buffer zones may create areas where possums can seek refuge, which may negate the effectiveness of the poisoning operation.

1. Introduction

Large-scale control of introduced mammalian pests (especially the Australian brushtail possum (*Trichosurus vulpecular*) in New Zealand use baits containing sodium monofluoroacetate (compound 1080) which are often applied aerially over mountainous terrain. Such aerial applications are often contentious, with many public concerns over the fate of 1080 in the environment, its effect on non-target species, and potential contamination of surface and ground water.

As a result of these concerns, regional councils throughout New Zealand often impose specific conditions on aerial 1080 operations, and in particular whether buffer zones are placed around waterways to prevent accidental contamination. Information gleaned by the Animal Health Board (AHB) from regional councils regarding these conditions shows that 4 councils impose buffers around all waterways, 5 have buffers around all waterways >3 m in width, and 2 councils have no buffer zones around waterways that are not used for drinking water (Table 1). Although buffer zones may exist around streams >3 m wide, there are many streams smaller than this, especially in steep mountainous country where the aerial 1080 application occurs most often. 1080 bait is thus likely to fall into small streams during aerial operations.

The most common aerial application rate of 1080 baits is 3 kg ha⁻¹ (Table 1), or 500 baits per ha for the 6 g Wanganui No 7 baits. This is equivalent to 1 bait every 20 m². Thus 10 baits could by chance fall into a 100 m section of stream that is 2 m wide. Such unavoidable discharges of 1080 to streams may have potential ecological consequences, especially given the sensitivity of invertebrates to 1080 (Eason et al. 1993; Spurr and Drew 1999). The effects of 1080 on New Zealand aquatic invertebrates and fish are, however, unknown. This knowledge gap is of concern, especially considering the often contentious nature of aerial 1080 drops. Moreover, small streams may have only a limited ability to dilute any 1080 leaching from baits, and so any adverse effects on stream life could be higher in small streams than larger streams.

Aquatic invertebrates are a key component of freshwater ecosystems. These animals are mostly the immature stages of a wide range of insects such as caddisflies, stoneflies, mayflies and two-winged flies such as midges and sandflies. They are important in transferring plant-derived organic carbon into animal biomass, and are food for a variety of native and introduced fish, as well as a number of bird species (e.g., Blue Duck). 1080 is toxic to terrestrial invertebrates that come into contact with baits and eat them (Eason et al., 1993; Spurr and Drew 1999). 1080 baits may also affect aquatic invertebrates, although few of these animals are likely to consume baits

directly (see Suren and Bonnet 2004). However, freshwater invertebrates may be adversely affected by 1080 as it leaches from baits in streams. Toxicity tests performed by the US EPA have shown that the non-observable effect concentration (NOEC) of 1080 for the small freshwater invertebrate *Daphnia magna* was 130 $\mu\text{g l}^{-1}$ (Fagerstone et al., 1994). At this high concentration, the effect of 1080 was regarded as being “practically non-toxic” to *Daphnia*. This NOEC level was about 30 times higher than the highest concentration of 1080 detected in streams in New Zealand (4 $\mu\text{g l}^{-1}$: Eason 2002, Green 2003), so it would seem unlikely that 1080 baits falling into streams would have a toxic effect on New Zealand invertebrates. Despite this assertion, no studies have quantified the toxicity of 1080 to any New Zealand freshwater invertebrates, nor looked at what effects baits that fall into streams have on freshwater invertebrates.

Table 1: Summary of the size of waterway buffer, application rate and preferred bait type of selected regional councils throughout New Zealand. Data courtesy of AHB.

Region	Waterway buffer	Average application rate	Type of bait
Waikato	No buffers for RMA resource consents. Some TLAs require 50m buffers	5 kg/ha Range 2 to 15 kg	Carrot (7-9 g) at 0.08% to 0.15% w/w toxic loading Wanganui No 7 baits in 2-3 g; 5-7 g and 7-9 g bait sizes up to 0.15% w/w
Bay of Plenty	60m buffers irrespective of width	5-8 kg/ha	Carrot (7-9 g) at 0.08% w/w
Hawkes Bay	20m irrespective of width	Initial Operations: 10 kg/ha Other Operations: 3 kg/ha and 5 kg/ha	Wanganui No 7 baits (5-7 g) preferred. Toxic loadings of 0.08% and 0.15% w/w.
Manawatu/Wanganui	20m for waterways wider than 3m	3 kg/ha prefeed 5 kg/ha toxic	Wanganui No 7 (0.15% w/w)
Wellington	If not for drinking, no set distance. If water-ways used for drinking water a 20 m buffer required.	2 kg/ha	Wanganui No 7 (0.15% w/w)
Marlborough	20m for waterways wider than 3m	2.5kg/ha for 6-8 g and 12 g baits 3-4 kg/ha for 2-3g baits	RS 5 (0.15% w/w)
Tasman	20m but 100m over water supplies and intakes	3 kg/ha	RS5 (0.15% w/w)
Canterbury	50m for waterways wider than 3m	3 kg/ha	RS5 (0.15% w/w)
West Coast	20m for waterways wider than 3m	3 kg/ha	Wanganui No 7 (0.15% w/w)
Otago	20m but 100m over water supplies and intakes	3 kg/ha	RS5 (0.15% w/w)
Southland	50m for waterways wider than 3m	3 kg/ha	RS5 or Wanganui No 7 (0.15% w/w)

Native or introduced fish may also be exposed to toxic effects of 1080. Most fish are predators that consume either freshwater invertebrates or terrestrial invertebrates that have fallen into streams, so the risk of consumption of baits would be extremely low. However, fish may be exposed to potential toxic effects of 1080 that leaches from baits. Toxicity tests have shown that fish are generally resistant to 1080. For example, fingerling bream and bass survived in water containing $370 \mu\text{g l}^{-1}$ (King and Penfound 1946), and rainbow trout showed no observable effects to being exposed to $580 \mu\text{g l}^{-1}$ over a 24 h period (Batcheler 1978). 1080 is thus regarded as being either non-toxic, or slightly toxic to rainbow trout (Fagerstone et al. 1994). The effect of 1080 on New Zealand native fish is, however, unknown. Absence of such specific information is of concern, especially given the contentious nature of the use of 1080 within New Zealand. Moreover, many native fish such as eels (tuna) and whitebait (inanga) are culturally important to Maori, who may be concerned at the effects of 1080 on these animals.

Despite the widespread use of 1080, and the potential for adverse impacts to aquatic organisms, only 3 studies to date have attempted to quantify the impacts of the aerial application of 1080 on stream ecosystems (Taranaki Regional Council 1993, 1994b; Suren and Lambert 2002). These studies used qualitative kick samples to characterise the invertebrate communities in streams that flowed through areas where 1080 had been aerially applied. Invertebrate samples were collected from streams before and after the aerial application of 1080. A number of indices were calculated that described the invertebrate communities; for example the number of sensitive mayfly, stonefly and caddisfly taxa, or the % abundance of specific insect groups.

No demonstrable effects of the aerial 1080 application were observed on invertebrate communities in any of these studies (Taranaki Regional Council 1993, 1994b; Suren and Lambert 2002), suggesting that stream invertebrates were not sensitive to 1080. However, the sampling strategy of these studies made the implicit assumption that they adequately sampled areas where 1080 contamination had occurred. This was, however, not controlled for, as the aerial application of 1080 would have resulted in an uneven and unknown quantity of bait entering the streams. As such, it could be argued that while both studies reported no adverse impacts of the 1080 operations at a catchment scale, neither was able to adequately quantify the effects of 1080 bait in individual streams, as no attempt was made to quantify the number of baits in streams. Moreover, neither study specifically examined the effects of potential 1080 contamination on the fish communities in the streams.

Although there have only been a few studies examining the effect of 1080 on stream ecosystems, extensive programmes have tested water quality for signs of 1080 contamination following aerial applications. Between 1990–2002, 1556 water samples

have been collected after large-scale possum or rabbit control operations (Eason 2002; Green 2003). Of these samples, only 3.5% (i.e., 58) showed any residues of 1080. Of these, the highest concentration was $4 \mu\text{g l}^{-1}$ (Eason 2002) found in a stream in the Te Kopia Scenic Reserve. Two more water samples contained $\leq 3.5 \mu\text{g l}^{-1}$ of 1080. In most of the other cases where 1080 has been detected, the amounts were $< 1 \mu\text{g l}^{-1}$. The Medical Officer of Health has stipulated a Provisional Maximum Acceptable Value (PMAV) for 1080 in water as $3.5 \mu\text{g l}^{-1}$, although a lower value of $2 \mu\text{g l}^{-1}$ is used when water is used for human consumption. Thus concentrations hitherto found by extensive water monitoring programmes have all been below this PMAV.

The few instances where 1080 has been detected in streamwater most likely reflects the presence of bait in streams (Parfitt et al. 1994; Eason et al 1999). Laboratory tests show that 1080 breaks down relatively quickly in water, mostly by bacteria naturally present in stream water or associated with aquatic plants, and that the rate of 1080 breakdown is faster in warmer water (Ogilvie et al. 1996). Given the fact that the aerial application of 1080 is generally done during winter months, it is likely that 1080 could persist in cold-water streams for some time. Eason et al (1999) suggested that dilution of 1080 in streams would be more important in reducing it to toxicologically insignificant concentrations than its breakdown by bacteria. However, despite this assertion, no studies have specifically looked at the effect of 1080 on aquatic invertebrate and fish communities found in streams. This lack of information is especially pertinent, as

- water quality monitoring studies have occasionally shown traces of 1080 in stream water, and as such may affect instream communities on a local patch-scale;
- information on the fate and effects of 1080 baits that fall into streams may provide important information for the ERMA reclassification process for 1080;
- considerable public concern currently exists about the fate of 1080 in water, but this concern has not been matched by adequate research.

The present study was designed to increase our understanding of the fate and effects of 1080 baits that fall into streams during aerial operations. For this we investigated 3 key questions:

- What was the degree of accidental contamination of streams by 1080 baits flowing through areas where 1080 was applied aerially;

- What was the fate of 1080 baits in streams, how long do they take to break down, and how quickly does the 1080 leach from baits;
- What was the effect of 1080 baits on freshwater fish and invertebrate communities in small streams?

2. Methods

2.1. Quantifying the degree of accidental contamination

2.1.1. Field methods

Streams flowing through areas where aerial 1080 operations were conducted were surveyed. Four areas were surveyed: the Lewis Pass and Mt Grey regions in North Canterbury, the Awatere Valley in Marlborough, and the Moana – Ruru region on the West Coast. Different regulatory agencies were responsible for the operations in each of these areas, and each used different sized baits and had different application rates (Table 2).

Table 2: Summary data of the 4 1080 operations where streams were surveyed to quantify the degree of accidental 1080 contamination by baits.

Operation	Regulatory authority	Bait (all 0.15% w/w 1080)	Bait weight	Application rate (kg ha ⁻¹)	Streams surveyed
Lewis Pass	DoC	12 g baits	12 g	2.5	20
Mt Grey	Target Past (ECan)	RS5	7.6 g	2.6	8
Awatere	Marlborough District Council	2 g baits	2 g	2.5	9
Moana - Ruru	West Coast Regional Council	Wanganui No 7	6.4 g	3.0	11

Surveys were made of randomly selected streams in each area by walking up 100 m transects located randomly within each stream. The location of baits along each transect was recorded to the nearest meter. A small underwater viewing tube was used to look for baits in deep pools, or in areas of turbulent water where the streambed was not visible. All baits that were either submerged in the stream or in the active bed area were recorded. The average stream width and depth was also recorded, and velocity estimated by timing small twigs to travel a specific distance. Details about overhead canopy cover (0-25%, 25-50%, 50 – 75% and > 75%) were also recorded.

2.1.2. Statistical analysis

We assumed that aerial application of 1080 baits would result in a uniform distribution along streams flowing beneath areas where baits were applied. This assumption was tested using the Kolmogorov-Smirnov test (Zar 1984) to see whether baits were uniformly distributed up the 100 m stream segments. Following this test, the average number of baits found in a 2 m, 5 m or 10 m block up each stream was calculated. Thus, for the 10 m block analysis, we calculated 91 moving averages (i.e., the average number of baits from 1-10 m, 2-11 m, 3-12 m etc), 96 moving averages for the 5 m block analysis (i.e., the average number of baits from 1-5 m, 2-6 m, 3-7 m etc.), and 99 moving averages for the 2 m block analysis (i.e., the average number of baits from 1-2 m, 2-3 m, 3-4 m etc.). The maximum of these moving averages was then used to represent the “worst-case” assessment of the number of 1080 baits that could be found in different block sizes in each stream.

The expected loading of 1080 bait contamination in the streams was calculated based on the weight of baits, the application rates used in each operation, and the stream area that was surveyed. The χ^2 -test was used to assess whether the observed number of baits in a stream was significantly different to the expected number of baits in each stream.

Finally, regression analyses was used to see whether there was any relationship between the number of baits in each stream and stream width: the assumption being that the number of baits would be in proportion to the stream area.

2.2. The fate of 1080 baits in water

Laboratory experiments were performed to examine the fate of baits that fall into moving water and to quantify the rate that 1080 leaches from baits. Samples of the large baits (mean weight = 11.1 g) as used by the Department of Conservation (hereafter referred to as DoC baits) and Wanganui No 7 baits (mean weight = 6.4 g), both containing 0.15% w/w of 1080, were obtained from Animal Control Products for this experiment. An oval recirculating flow-tank consisting of curved ends (inside diameter = 60 cm) separated by two parallel-sided chambers (40 cm wide x 30 cm high, 265 cm long) was used for these experiments. A mixture of cobbles (mean size = 30 mm) was placed in this flume to a depth of 7 cm. Eight replicate cobble-filled plastic containers were buried into the cobbles so that their surfaces were flush with the bed of the flume. These containers were placed 15 – 20 cm apart along each straight channel (Fig. 1). Preweighed samples of both bait types were placed on the top of the cobbles in the plastic containers, which could be removed from the flow

tank when necessary. In this way, any material that had fragmented from individual baits and had fallen into interstitial spaces between cobbles were still recovered.

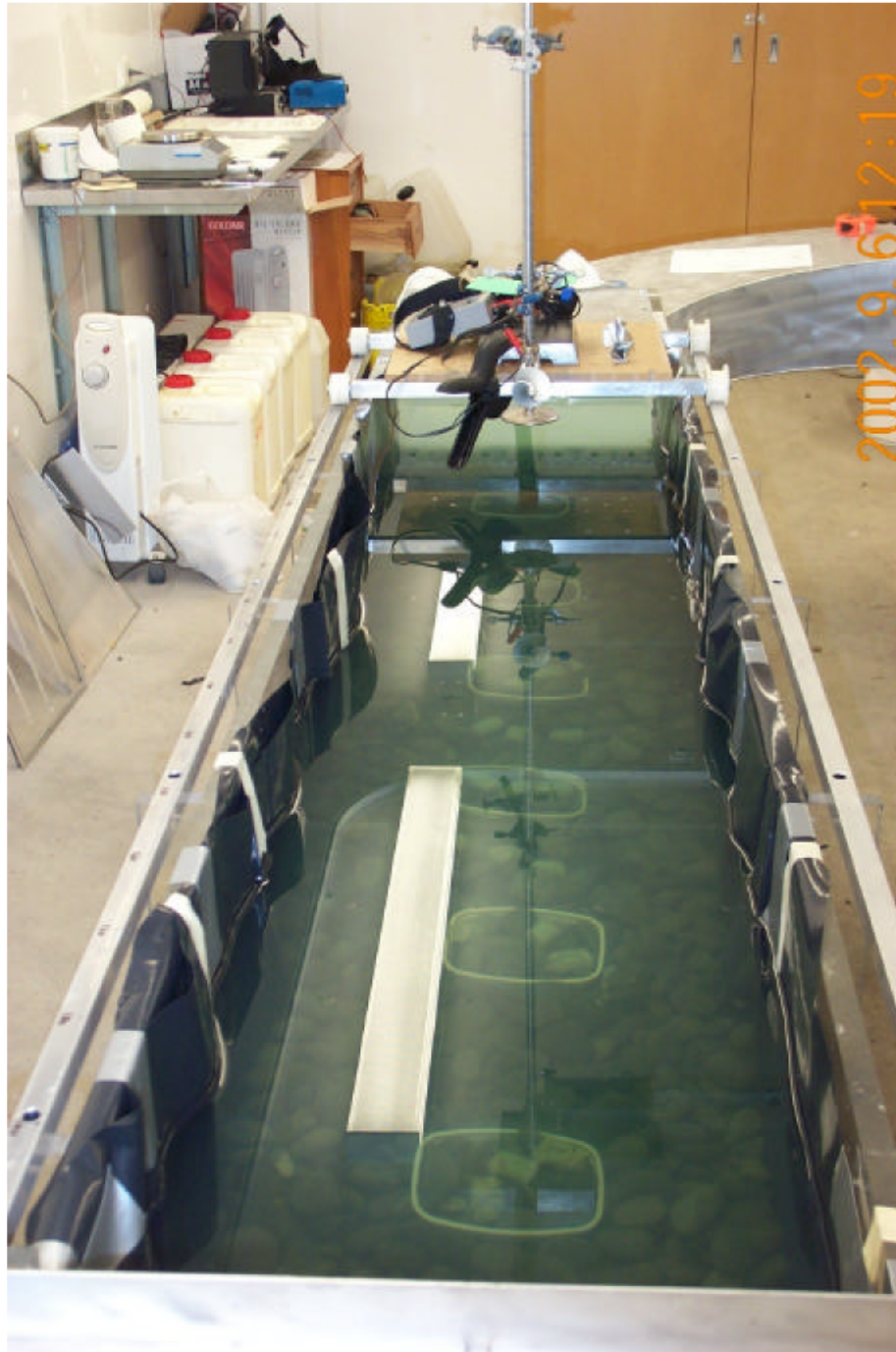


Figure 1: One side of the recirculating flume where 1080 baits were placed in small plastic containers to measure the degree to which they fragment and decompose in water.

Water velocity was maintained in the flume by the use of a propeller, and velocities checked in both straight channels using an Ott meter at 0.4 x the depth. Velocities were set at 15 cm s⁻¹, typical of many steep headwater streams where aerial operations commonly occur.

Replicate samples of bait were placed into each plastic container, and left for increasing lengths of time. They were removed at regular intervals (8, 24, 48, 72 and 84 h), dried (60°C) and reweighed to calculate weight loss due to fragmentation. Separate trials were carried out for the DoC baits and Wanganui No 7 baits.

An additional experiment was conducted using Wanganui No 7 bait to measure the rate that 1080 leached from the baits. Baits were placed on the surface of cobbles in the flume, and collected after 2, 4, 6, 12, 24 and 36 h. These were placed in individual plastic bags and frozen (-18°C) pending analysis. They were sent to Animal Control Products (Wanganui) where they were analysed for residual 1080 concentration.

2.3. The effect of 1080 on native fish

2.3.1. Study sites

Four streams in the Mawhera State Forest in the Grey Valley were selected for the field study (Fig. 2). These sites were all headwater tributaries of Deadhorse and Wallaby Creeks, and were selected based on the following criteria:

- small enough (< 5 m wide) not to have buffers around them;
- relatively remote from human habitation and intensive farming;
- in areas where previous 1080 operations had not recently occurred;
- had little recreational use;
- were relatively accessible for logistical reasons.

Two of these streams (Deadhorse trib and Washout Creek) flowed through catchments dominated by plantation pines, while the upper catchments of the other 2 streams flowed from native bush into pine. The immediate riparian vegetation of all streams was a mixture of pines, native scrub and grasses. Boulders, cobbles and gravels dominated the streambed material at all sites. Discharge in the 4 streams varied, and was highest in Wheelbarrow Creek and lowest in Washout Creek (Table 3). Stream pH was relatively low at all sites, and was lowest at Wheelbarrow Creek and highest at Washout Creek (Table 3). Conductivity was low at all sites.

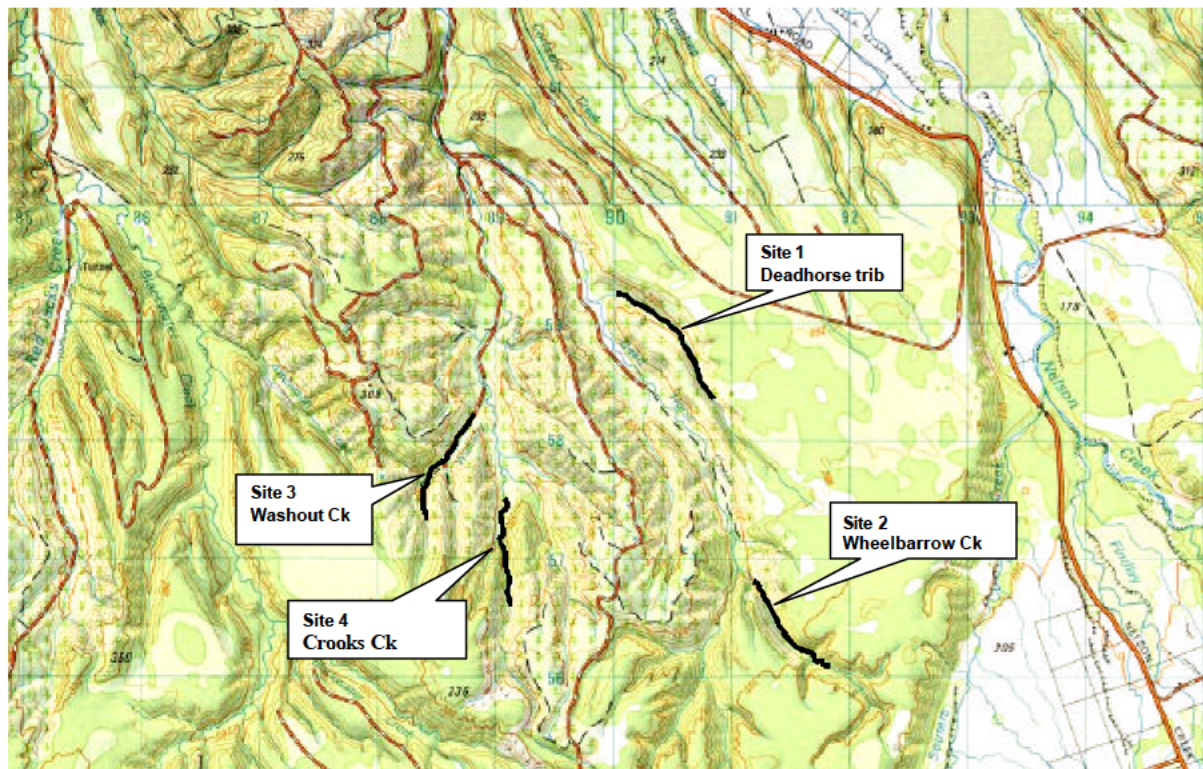


Figure 2: Location of the four study sites that were small headwater tributary streams of Deadhorse and Wallaby Creeks in the Mawhera State Forest. Names given to these streams are unofficial names only.

Table 3: Summary of the physical and water-quality conditions of the 4 study streams. Stream discharge was measured at the beginning of the experiment (13 March 2004) where it was relatively high, and at the end (23 March 2004), when it had decreased. Other parameters were measured on all sampling occasions, and are the averages ($n = 4$).

Stream	Discharge (l s ⁻¹)	Width (m)	pH	Conductivity (mS cm ⁻¹)	Dissolved O ₂ (mg l ⁻¹)	Temperature (°C)
Deadhorse Trib	128 - 45	2.6 - 2.9	4.73	0.025	7.94	12.5
Wheelbarrow Creek	238 - 35	1.5 - 2.8	4.53	0.025	7.88	12.5
Washout Creek	76 - 26	1.8 - 2.3	6.21	0.038	8.05	14.7
Crooks Creek	84 - 28	2.0 - 2.1	5.61	0.031	8.13	12.5

A 400 m section of stream was chosen for each study, with 2 stations above the area where the 1080 baits would be introduced, and 2 stations at increasing distances below.

2.3.2. Field methods

The impact of 1080 leaching from baits that fall into streams was examined on three selected fish species that are common in streams on the west coast: longfin eels (*Anguilla dieffenbachii*), koaro (*Galaxias brevipennis*) and upland bullies (*Gobiomorphus breviceps*). These fish were obtained for the experiment by electric fishing small streams flowing in the Grey Valley on the west coast. All fish were transported to the experimental streams within 12 h of collection in plastic buckets equipped with battery-operated air pumps to ensure constant aeration.

Eight fish of each species were placed in individual plastic fish cages, constructed from 250 l plastic drink containers that were cut in half longitudinally (Fig. 3). These cages had 2 mm mesh netting on their ends, as well as on their tops. One of the ends was hinged to enable fish to be transferred to a larger bucket for counting. Up to 10 large cobbles (mean diameter 140 mm) were added to each container to provide shelter for the fish in each cage. Four PVC tubes (24 mm diameter) were also added to cages holding the eels, to provide additional shelter for these normally cryptic animals. A velcro-sealed mesh lid enabled the cobbles and PVC pipes to be removed prior to examining the fish, and allowed the fish to be replaced into the cages while partially submerged. This lid was then resealed and the cage submerged again until the next observation.



Figure 3: Photo of the experimental fish cages that housed 8 individual fish at sites above and below areas where 1080 baits were added to the streams.

With the exception of Crooks Creek, cages were deployed at 4 locations in each stream: two stations above where 1080 baits were to be added (Sites 1 and 2), and at two stations at increasing distances below (Sites 3 and 4). Unfortunately, all 3 of the uppermost cages were stolen from Crooks Creek before the experiments started, so this site only had the one upper site located 10 m above the baits. All cages were deployed in relatively deep runs or pools. Small depressions were dug in the substrate and cages positioned in these depressions so that they were either completely or mostly submerged. Each cage was anchored to the streambed. Large cobbles were placed on top to ensure the velcro seal on the mesh lid remained closed, and to help weigh the cage down. Cages were placed at a site approximately 100 m and 10 m below the site where the 1080 baits were placed, while the upstream cages were placed approximately 10 m and 100 m above the 1080 baits. Fish were added to the cages on 13 March and the first observation made 2 days later (15 March) for signs of mortality. For each observation, all cobbles and PVC pipes (in the eel cages) were removed from each cage, which was then emptied into a large collecting bucket (Fig. 4). All fish were counted and any dead fish recorded. The cobbles and PVC pipes were replaced in each cage, which was then partially submerged under the water. The fish were then poured from the collecting bucket back into each cage, the mesh lid resealed, and the cage redeployed (Fig. 4). Aquatic insects that naturally drifted into the cages provided the fish with food. Extra invertebrates (obtained from the streams) were added to the cages every 4 days to ensure no food limitation occurred.

A second observation of fish survival was made 4 days later, on 18 March, after which 1080 baits (Wanganui No 7) were added to each stream. Observations of fish mortality at all 4 stations were made again 1 and 4 days later. All fish were anaesthetized after the last sampling day and their lengths measured. This data was assessed by 2-WAY ANOVA to see whether there was any significant difference in fish size between locations above or below the 1080 baits (location effect), or between sites in each location.

The number of baits added to each stream was based on the maximum number found within a 10 m block of the stream surveys (see Section 3.1). A total of 80 baits were added to the largest of the study streams (Wheelbarrow Creek), representing a 10 fold increase of the maximum number of baits that were found in the Mt Grey operation, using RS5 baits. The number of baits added to the other streams was in proportion to their discharge on the day, to keep the estimated 1080 concentration in each stream relatively similar. Thus 50 baits were added to Deadhorse Tributary, and 30 baits to both Washout and Crooks creeks. Baits were counted into three sets of nylon mesh bags (mesh size 10 mm) which were anchored to the streambed at even positions across the stream channel at each site. Care was taken to ensure that all bags were submerged (Fig. 5).

Water samples were collected at Sites 3 and 4 concurrently with observations of the fish. Samples were also collected 2, 4 and 8 h after the addition of baits, reflecting the rapid dissolution rate of 1080 (see Section 3.2 below). After the completion of the



Figure 4.: Fish mortality was assessed by carefully pouring the fish in each cage into a large tub where they could be examined. All healthy fish were netted and placed into a small bucket, and then carefully poured back into the cages which were sealed and repositioned in each location.



Figure 5: Photo of the 1080 baits placed in Deadhorse Trib. Equal numbers of baits were placed in coarse mesh bags, which were placed at even intervals across the stream to ensure complete mixing of any plume of 1080 compound that would leach from the baits.

trial, an additional set of 1080 baits were added to the streams and 5 replicate baits collected after 1, 2, 4 and 8 h to measure residual 1080 concentration. It was not possible to collect any of the baits used in the trial as this would have altered the potential contaminant load into the streams.

2.4. The effect of 1080 on invertebrate communities

2.4.1. Study sites

It was planned to conduct the fish and invertebrate sampling programmes simultaneously in the 4 streams. However, invertebrate densities in the study streams were all very low, and not sufficient to be able to detect any changes in density that may have been caused by the presence of 1080 leaching from the baits. This low density reflected the unusually high rainfall and river flows that occurred in the region during February 2004. River flows in the Grey River at Waipuna (approximately 20 km NE of the study sites) showed that the mean daily flow in the Grey River was $117 \text{ m}^3 \text{ s}^{-1}$ during this month. The average mean daily flow for February here was only $37 \text{ m}^3 \text{ s}^{-1}$, and the previous highest figure ($82 \text{ m}^3 \text{ s}^{-1}$) was in 1980. A large flood (with a peak hourly flow of c. $720 \text{ m}^3 \text{ s}^{-1}$) occurred on 22 February (Fig. 6), and this would have reduced invertebrate densities in the rivers and streams in the area (e.g., Scrimgeour and Winterbourn 1989; Gjerlov et al. 2003).

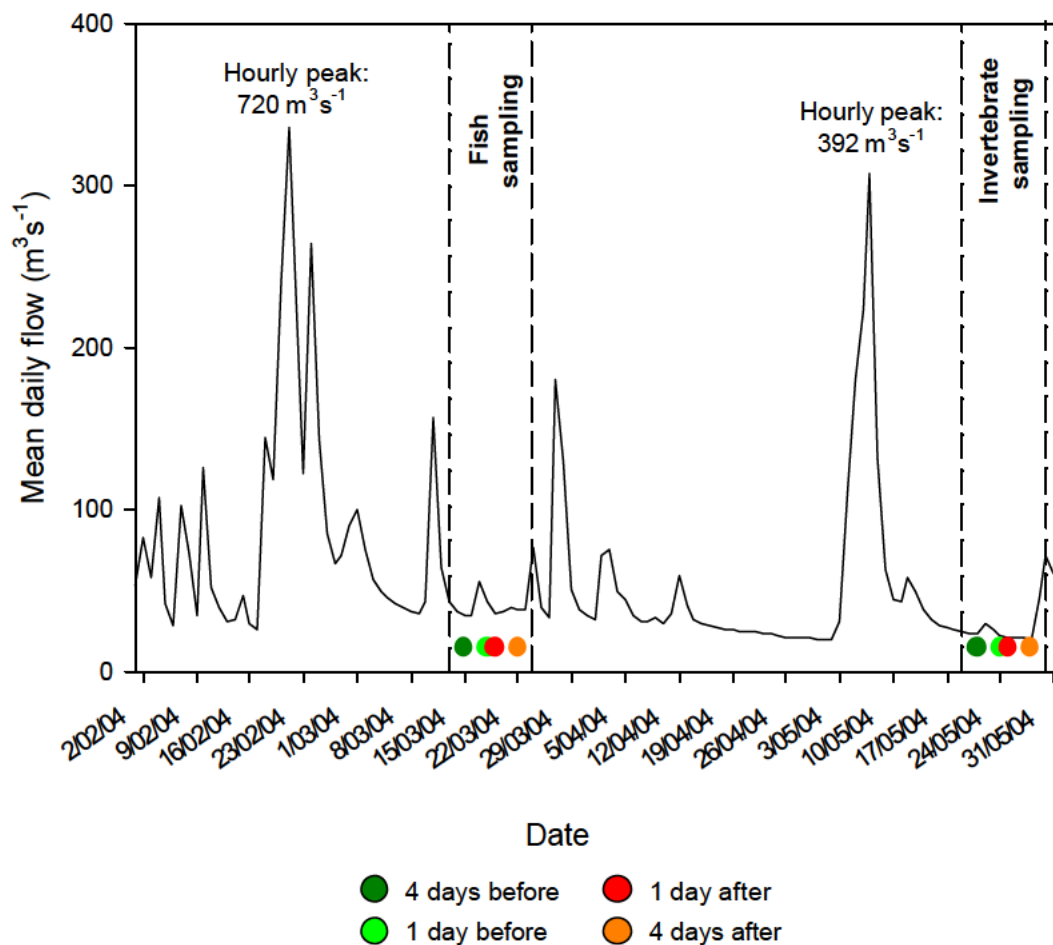


Figure 6: Hydrograph of the Grey River at Waipuna, approximately 20 km NE of the study sites showing mean daily flows during the course of the study, and the different sampling times for the fish and invertebrate studies. Note the large floods in mid February and early May. Hourly flows were much higher than daily means, with a maximum flow of $720 \text{ m}^3 \text{s}^{-1}$ on 21 February, and $392 \text{ m}^3 \text{s}^{-1}$ on 5 May.

As a result of low invertebrate densities in the streams in March, the invertebrate experiment was delayed until mid-late May. Three of the 4 streams (Deadhorse trib, Wheelbarrow Creek, and Crooks Creek) were selected for the study: Washout Creek was not suitable at this time as large amounts of filamentous green algae covered the streambed, which would have reduced habitat conditions for invertebrates, and hindered the processing of invertebrate samples. We subsequently chose a fourth stream ('Small Creek'), located near Crooks Creek (Fig. 7). This was much smaller than the other three streams (Table 4). Like the other sites, it too flowed from a catchment dominated by pines.

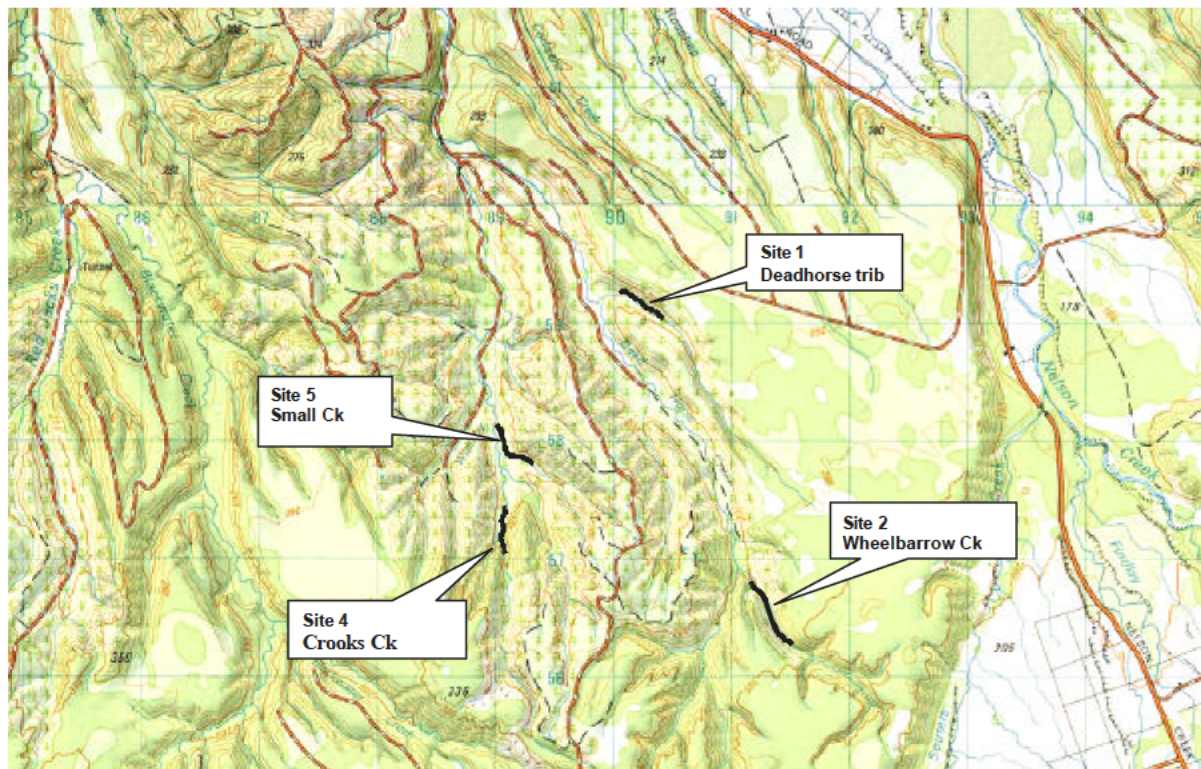


Figure 7: Location of the four study sites used in the invertebrate study. Names given to these streams are unofficial names only.

Although there was a relatively high flow event on 5 May (Figure 5), the peak hourly flow in the Grey River was only $392 \text{ m}^3 \text{ s}^{-1}$, and much less than the peak flow in February ($720 \text{ m}^3 \text{ s}^{-1}$). This suggests that rainfall intensity in the region was not as high as in February, and so there would have been less impact on the invertebrate communities. This was confirmed by examination of invertebrate densities on individual rocks on 21 May, which were high enough for the invertebrate programme to commence.

Discharge in all streams during May was low, and much lower than during the February fish study (Fig. 6). Discharge was lowest in Small Creek, and highest in Wheelbarrow and Crooks Creeks (Table 4). Stream pH was relatively low at all sites, as was conductivity, although this was higher in Small Creek than the other streams (Table 4). Stream temperature was much lower than during the February sampling period, and dissolved oxygen was higher, most likely reflecting the colder water temperatures.

As with the fish study, a 400 m section of stream was chosen for each study, with 2 stations above the area where the 1080 baits would be introduced, and 2 stations at increasing distances below

Table 4: Summary of the physical and water-quality conditions of the 4 study streams selected for the invertebrate study. Stream discharge was measured at the beginning and end of the experiment. Other parameters were measured on all sampling occasions, and are the averages ($n = 4$).

Stream	Discharge (l s ⁻¹)	Width (m)	Median pH	Conductivity (mS cm ⁻¹)	Dissolved O ₂ (mg l ⁻¹)	Temp (°C)
Deadhorse Tributary	14.6 – 11.6	2.6 - 2.9	5.9	0.032	11.95	6.5
Wheelbarrow Creek	17.5 – 15.6	1.5 – 2.8	5.7	0.033	10.84	7.4
Crooks Creek	16.6 – 15.2	1.8 – 2.3	6.1	0.033	11.23	7.6
Small Creek	6.1 – 5.9	2.0 – 2.1	6.5	0.053	10.53	8.7

2.4.2. Field methods

Because 1080 is a toxin, we were specifically interested in examining invertebrate densities at the selected sites, as these were expected to decrease at sites below where the baits were placed. We thus needed to collect data on actual invertebrate densities at all the sites. Such quantitative information is normally collected using a Surber or Hess sampler (Stark et al. 2001). These sampling devices are basically quadrants or cylinders of a known surface area that have their upstream ends open to the water flow, and their downstream ends occupied by a collecting net. They are placed flush to the streambed, and all material within them is stirred up by hand, washing invertebrates and leaf litter into the net. Unfortunately, the substrate in each experimental stream consisted of relatively large cobbles and small boulders - conditions that precluded the use of the conventional quantitative samplers, as their bases would not be flush with the streambed. This problem was overcome by sampling individual rocks within the streams, and removing all invertebrates found on each rock (e.g., Death and Winterbourn 1994). A triangular net (mesh size 300 μ m) was placed immediately below each rock (mean size 16.2 cm $\times$ 11.9 cm $\times$ 7 cm) that was then lifted into the net. Any invertebrates living on each rock were removed by scrubbing, and collected in the net. All samples were preserved in the field using iso-propanol. Rock sizes (length $\times$ width $\times$ height) were also measured so that invertebrate densities could be calculated.

Invertebrates were collected at all sites 4 days and 1 day prior to the introduction of the 1080 baits. Samples were then collected from all sites 1 day and 4 days after. A power analysis of invertebrate data collected previously from individual rocks in Deadhorse Creek showed that 10 replicate rocks from each sample location would be sufficient to detect a 20% reduction in invertebrate density with an 80% certainty. Examination of invertebrate densities in small headwater streams in Arthurs Pass National Park has shown that monthly densities can vary by up to 30% of the long-term (18 month) average (Suren 1991, unpublished data). Given this natural

variability, the ability of the present analysis to detect a 20% reduction in density over a shorter time period (8 days) was deemed to be sufficiently powerful to detect even small changes to the invertebrate communities as a result of 1080 leaching from the baits.

The number of baits added to each stream was based on the number used in the fish survey (see Section 2.2), and corrected for the reduced flows in the streams during May. We therefore added 25 baits to Wheelbarrow Creek 20 baits to Washout and Crooks creeks, and 15 baits to Small Creek (Table 5). Note that the predicted concentrations in the streams as a result of these additions were almost three times higher than the predicted concentrations as used for the fish trials (Table 5). In this regards, the experiment represented a worst-case scenario that could be expected to happen following aerial application of 1080.

Table 5: Summary of stream flows, the number of 1080 baits added to each stream, the total weight of 1080 and the estimated concentration of 1080 in the water (assuming complete dissolution in 8h).

Site	Trip	Flow (l/s)	Volume (l / 8h)	Baits	1080 mg	Estimated concentration ($\mu\text{g l}^{-1}$)
Deadhorse	1	86	1238400	50	480	0.39
Wheelbarrow	1	105	1512000	80	768	0.51
Washout	1	51	734400	30	288	0.39
Crooks	1	56	806400	30	288	0.36
Deadhorse	2	14	201600	20	192	0.95
Wheelbarrow	2	17	244800	25	240	0.98
Crooks	2	16	230400	20	192	0.83
Small	2	6	86400	15	144	1.67

All invertebrate samples were returned to the NIWA Greymouth laboratory for processing using a modification of Protocol P3 methods as outlined by Stark et al (2001). This modification consisted of all samples being scanned on small Bogorov trays (see Winterbourn and Gregson 1989) under a stereo-microscope. This method allowed small animals such as small chironomids and nematodes, copepods and ostracods to be counted, and gave us a greater ability to observe changes in the invertebrate communities.

2.4.3. Statistical analysis

A number of metrics were calculated from the invertebrate data, including taxonomic richness, the number of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, and the % of EPT taxa. The EPT taxa are regarded as being particularly sensitive to instream conditions, and are affected by even relatively small changes to water quality. Although no data exists on the lethal effects of 1080 to different New Zealand aquatic invertebrates, we assumed that aquatic insects such as mayflies, stoneflies and caddisflies may be affected by 1080 leaching from baits that had fallen into the streams. Consequently, we expected to observe a reduction in either the number of EPT taxa, or the % of EPT taxa if 1080 leaching from the baits drop had affected instream conditions. Even if the “sensitive” EPT groups were unaffected by 1080 and other invertebrate groups were, there would still be differences % of EPT taxa.

The Macroinvertebrate Community Index (MCI) score was also calculated, based on invertebrate presence – absence data. This index, developed by Stark (1985; 1993), has been widely adopted for use as a biotic index by many biologists throughout the country (e.g. Quinn and Hickey 1990; Donald 1995; Quinn et al. 1992; Main 1995; Norton and Valentine 1995; Taranaki Regional Council 1994a; Quinn and Cooper 1997). This index, and its quantitative variant the QMCI, was originally developed to assess organic pollution in 27 cobble streams in Taranaki (Stark 1985). It was derived by examining the abundance of a number of invertebrate taxa in otherwise similar streams that were assigned *a priori* to one of three water quality classes based on their degree of organic pollution. In principle, streams with an MCI >120 are considered pristine, while streams with an MCI <80 are considered grossly enriched. Although the MCI was derived to assess organic pollution, *any* stress placed on the invertebrate communities that causes a loss of some taxa will most likely result in a change in MCI score. Thus, although we know little about the individual susceptibility of aquatic invertebrates to 1080, any effects that change the relative abundance of different invertebrate taxa will result in changes to the MCI and QMCI scores.

The effect of the 1080 baits on the overall invertebrate communities in the 4 streams was first assessed using the PERMANOVA computer programme (Anderson 2004). This programme first calculates the Bray-Curtis similarity of the invertebrate communities collected on the 4 sampling occasions (4 days and 1 day before the 1080 was added, and 1 day and 4 days after) at the 4 sample sites (2 sites above and 2 sites below). The Bray-Curtis similarity measure is a statistical measure of how similar the species assemblages are in two or more samples: samples with a similarity score of 1 are considered identical, whereas samples with a similarity score of 0 share no similar species. The PERMANOVA programme compares the actual Bray-Curtis similarity measures of the samples collected at the different times and locations with the Bray-

Curtis scores of samples that were randomly assigned to different sampling occasions and locations. If the original sampling protocol (i.e., 2 samples collected before and 2 samples collected after the 1080, and 2 sites above and 2 sites below the 1080 baits) was indeed responsible for significant differences in invertebrate community structure, then the chances of finding a similar structure in the data by random permutations would be remote. If, however, differences in the invertebrate communities caused by the original sampling protocol were weak, then the random allocation of samples to a sampling occasion or location would result in similar differences (Bray-Curtis similarity measures) as those obtained by the actual protocol. The PERMANOVA procedure used 5000 different permutations to test whether observed differences could have arisen due to chance, or whether they were indicative of an actual effect.

Changes to the invertebrate communities in the 4 streams during the experiment were next analysed by Non-metric Multidimensional Scaling (NMDS). This statistical technique graphically represents how the invertebrate communities change over time by plotting samples on a 2-dimensional x - y axis plot on the basis of how similar the invertebrate communities are. Community similarity between all samples is first calculated by the Bray-Curtis similarity measure, and the NMDS then plots the samples according to the rank order of the pairwise similarities among sites (Digby and Kempton 1987). Thus samples with very similar communities (and high Bray-Curtis similarities measures) will be plotted close together on a graph, while samples with very different communities (and lower Bray-Curtis similarity measures) will be plotted further apart. If 1080 baits were affecting the invertebrate communities, then those samples exposed to the baits would be grouped separately from samples not exposed to the baits.

Finally, the invertebrate data was analysed by ANOVA based on the classical Before-After, Control-Impact (BACI) design that is often used to detect changes in biological communities as a result of human activities (Underwood 1991). This design uses 2 main experimental effects: *site* and *time*. It examines the status of the community over 2 *time* periods **B**efore and **A**fter a particular impact (in this case adding 1080 baits to the stream), and at 2 specific *sites* (a **C**ontrol site above the baits, and an **I**mpact site below the baits). The test detects whether there is any change in communities as a result of the impact over time, and whether this change differs between sites above and below the impact. Of particular interest in the standard BACI design is the *site* $\times$ *time* *interaction* term. If this term is statistically significant, it suggests that the behaviour of the biological community in the 2 sites over time was not constant. If, however, the interaction term is not statistically significant, then the pattern of any changes occurring at the control and impact sites is similar between the 2 sampling periods. Under this circumstance, the biological communities are not being affected by

the particular activity in question, but instead are responding to large-scale factors other than the one in question.

However, this was not a straight-forward BACI design, as:

1. the 2 downstream sites were located along a dilution gradient, so the lowermost site (100 m below the baits) would receive lower concentrations of 1080 than the upper site (10 m below the baits);
2. samples collected 4 days after the baits had been added to the streams may have been exposed to animals recolonising the area from downstream drift, so some recovery in invertebrate densities may have occurred.

As such, a significant site $\times$ time interaction term would not explicitly show whether there was an effect of 1080 on invertebrate densities. We therefore included extra terms in the ANOVA model:

1. the ‘site’ term was divided into a general “Above” and “Below” comparison as well as a “100m” and “10m” comparison. The interaction between “Above vs Below” $\times$ “100m vs 10m” represented a *spatial interaction* term;
2. the ‘time’ term was divided into a general “Before” and “After” comparison as well as a “4 day” and “1 day” comparison. The interaction between “Before vs After” $\times$ “4 days vs 1 day” represented a *temporal interaction* term.

The ANOVA model was calculated for all sites over all sampling times, as well as for the specific streams. We were particularly interested in the interaction terms that suggested that sites below the 1080 baits were behaving differently to those above. The relevant interaction terms that indicated this were:

- “Above vs Below” $\times$ “Before vs After” effect;
- the *spatial interaction* term $\times$ “Before vs After”;
- the *temporal interaction* term $\times$ “Above vs Below”.

If we detected significant differences in the invertebrate data for one of these interactions, then we could examine the invertebrate data in the individual streams to see where these differences occurred.

Because 1080 is a toxin, we expected it to reduce densities of at least some taxa at the sites below the 1080 baits. We also assumed that the magnitude of this reduction decreased with increasing distance downstream, and with increasing time. As such, we specifically looked for evidence that some aspects of the invertebrate communities were reduced at sites below the 1080 baits. Any results that showed an increase in invertebrate density below the 1080 baits would have most likely represented random effects that were unrelated to the presence of 1080 in the streams.

Differences in total invertebrate density, taxonomic richness, density of selected taxa, and the MCI and QMCI scores, and the EPT and % EPT were assessed by the ANOVA model using the GENSTAT package (GENSTAT 2001). All data was examined for normality and log transformed where necessary, adding the smallest non-zero number to measurements where zeros were present.

3. Results

3.1. Quantifying the extent of accidental contamination

A total of 48 streams were surveyed in the 4 aerial 1080 operations (Table 6). No baits were found in 10 of these streams: 5 each in the Lewis Pass and West Coast operations. All streams surveyed in the Awatere and Mt Grey operations had baits in them. The number of baits found in streams flowing through each operational area varied widely (Table 6), most likely reflecting the different bait sizes and subsequent application rates. The highest number of baits in streams occurred in the Awatere operation, which used the smallest bait size (2g), while the fewest number of baits were found in the Lewis Pass operation, which used 12g baits.

Table 6: Summary table showing the mean, minimum and maximum number of 1080 baits found in 100 m sections of streams exposed to four aerial 1080 operations.

Operation	Mean number of baits per section	Minimum number of baits per section	Maximum number of baits per section
Lewis Pass	4.8	0	15
Mt Grey	12.7	2	21
Awatere	23	7	38
Moana – Ruru	7	0	19

The data describing the location of baits in the 100 m sections of the 48 streams was examined to determine whether bait distribution was uniform up these streams. The Kilmogorov-Sminov test showed that the distribution of baits in 38 streams was not uniform, suggesting that the expected number of baits falling into a stream cannot be calculated from the bait application rate and the stream size. This contention was supported by comparing the theoretical number of 1080 baits that could fall in a stream based on application rate and stream area with the actual number found. Of the 48 streams examined, only 19 had a similar number of baits in them as would be expected based on their application rate. Sixteen streams had significantly less ($P < 0.05$) than the expected number, while 13 streams had significantly ($P < 0.05$) more.

There was also no significant relationship between the number of baits that were found in a stream and the stream width (Fig. 8), again emphasizing that the number of baits that fall into a stream during an operation is random.

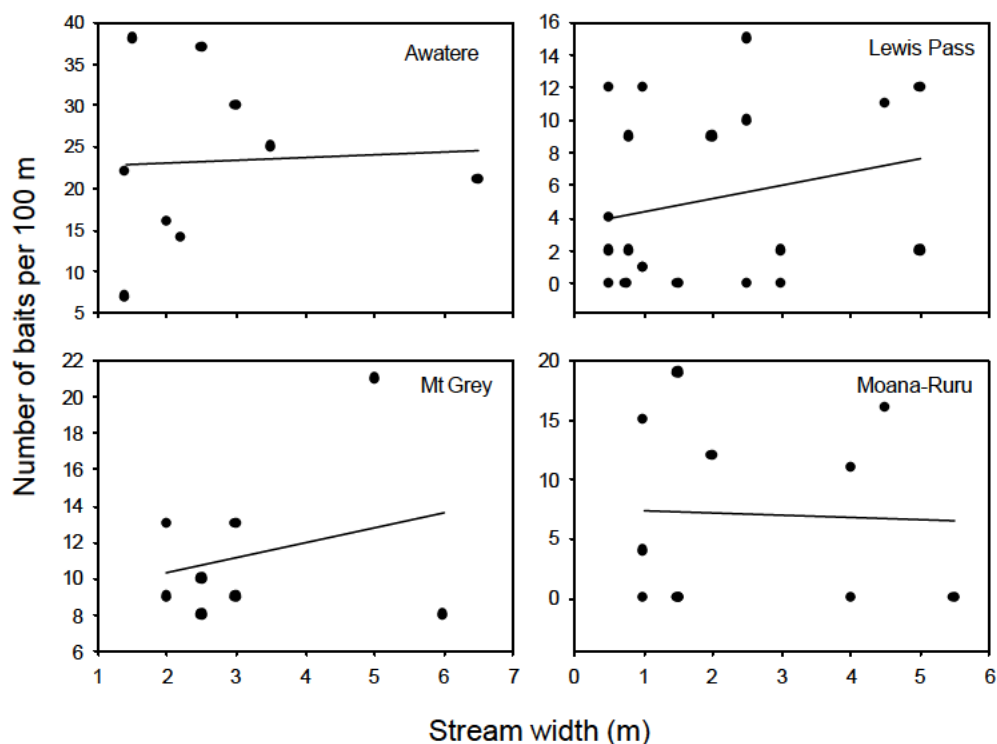


Figure 8: Relationships between the number of baits found in a 100 m transect and stream width as found in surveys of streams during four 1080 aerial operations. The regression lines for each graph are all not significant ($P > 0.05$).

Because we wanted to assess the effect of 1080 baits on fish and invertebrate communities, we needed to know what a realistic number of baits would be to add to

the experimental streams. The number of baits found in the 48 streams where 1080 was aerially applied was unrelated to either stream width or bait application rate. As such, there were no specific guidelines to use to determine the number of baits to be used in the experimental streams. To help set a realistic number, we calculated the maximum and average number of baits up a 2, 5 and 10 m section of the 48 surveyed streams (Table 7), with a view of using these figures as a guide to what was a realistic number of baits to experimentally add to streams.

Table 7: The maximum and average number of baits as found in 10m 5m, and 2 m sections up 100 m of streams in each of 4 operational 1080 drops.

Operation	Maximum number of baits in a section			Average number of baits in a section		
	10 m	5 m	2 m	10 m	5 m	2 m
Lewis Pass	5	4	2	2.5	1.7	1.5
Mt Grey	8	5	4	5	3	2
Awatere	14	11	5	7.3	5.3	3.4
Moana – Ruru	5	4	4	3.7	2.8	2.1

The maximum number of baits found in a 10 m stream section was 14 (Table 7). The Mt Grey operation, which used RS5 baits had a maximum of 8 baits per 10 m section, while both the Lewis Pass and Moano-Ruru drops had a maximum of 5 baits per 10 m stream section (Table 7). This information was useful in setting the limits for the number of baits that were added to the experimental streams (see Sections 3.3 and 3.4 below).

3.2. The fate of 1080 baits in water

There was a significant decrease in weight over time for both the DoC baits as well as the Wanganui No 7 baits ($F = 51.8$, $P < 0.001$), and the pattern of weight loss was similar in both baits ($F = 0.655$, $P > 0.05$). Most of the weight loss occurred over the first 72 h, with little extra after this (Fig. 9). For the first 48 h, the baits remained relatively intact, but slowly lost their original green coloration (Fig. 10). After 72 h, the baits had swollen to such an extent that they began to fragment, so that after 84 h, most of the structural integrity of the baits had been lost and they began to disintegrate (Fig. 11). It is expected that during this time in natural streams, these small fragments would be washed downstream, or become trapped in the spaces between the streambed. Any baits that fall into streams would thus be expected to disappear after 3-4 days as a result of this physical fragmentation.

Although the actual baits were expected to remain in a stream for up to 72 – 84 h before they disintegrated, examination of the 1080 concentration in the baits showed that over 50% of this chemical had leached from baits within the first 8 -12 h after being submerged (Fig. 12). By 24 h, the baits contained only 0.019% 1080: a loss of over 90% of the original 1080 in the baits (0.15%). This extremely rapid leaching of the 1080 from baits was surprising, especially as the 1080 was assumed to be evenly distributed throughout the baits, and that it would have taken some time for the baits to have become fully waterlogged. The short half-life of 1080 within submerged baits (c. 5 h) most likely reflects the high solubility of 1080 in water (Ogilvie et al. 1996). This finding has obvious implications for water quality monitoring programmes that are used to detect the presence of 1080 in water following aerial operations.

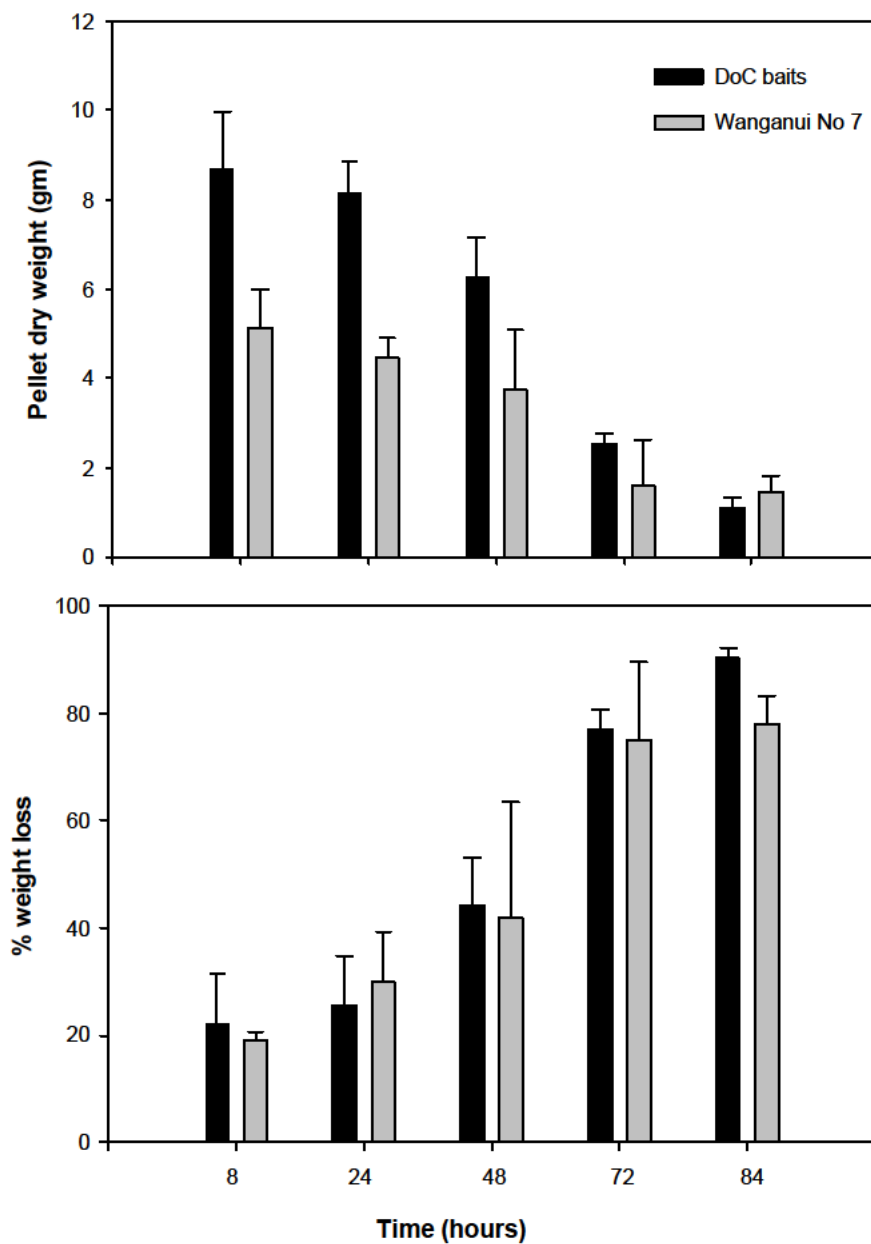


Figure 9: Graph of a) the dry weight of Doc baits and Wanganui No 7 baits and b) the % weight lost from these baits that were placed in the recirculating flume for increasing lengths of time ($\bar{x} \pm 1$ sd, $n = 5$).

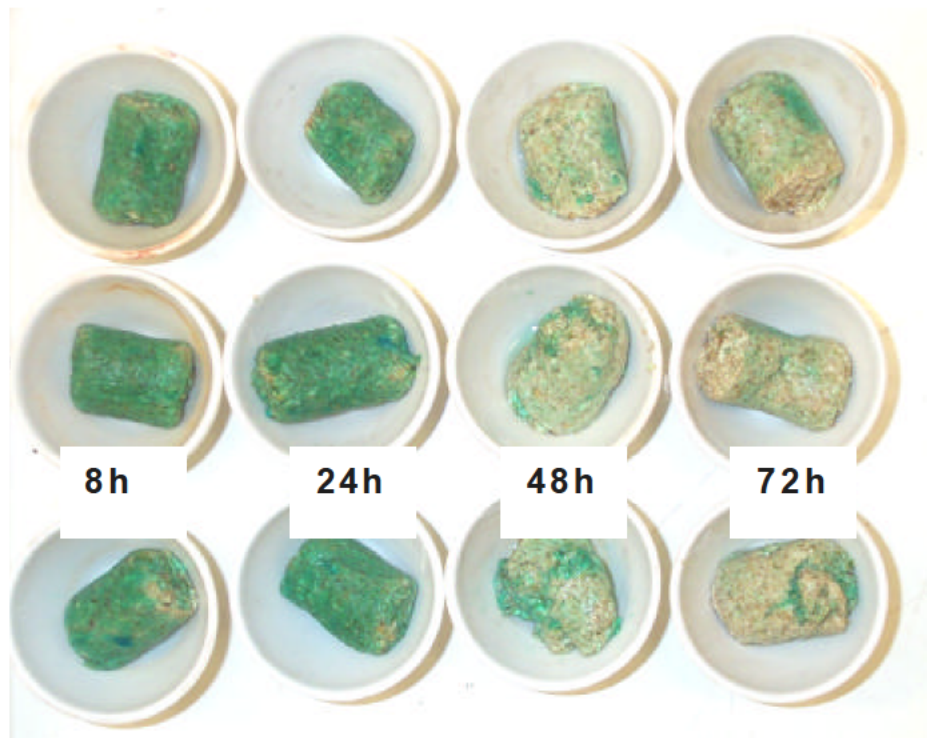


Figure 10: Photos of 1080 Wanganui No 7 bait collected from the flow tank after increasing lengths of time. Note the gradual discoloration of the baits over the 72 h period.



Figure 11: Photo of 1080 Wanganui No 7 bait collected from the flow tank after a 84 h period. Note that most of the baits have disintegrated, and that there was little remaining of the original coloration after this time.

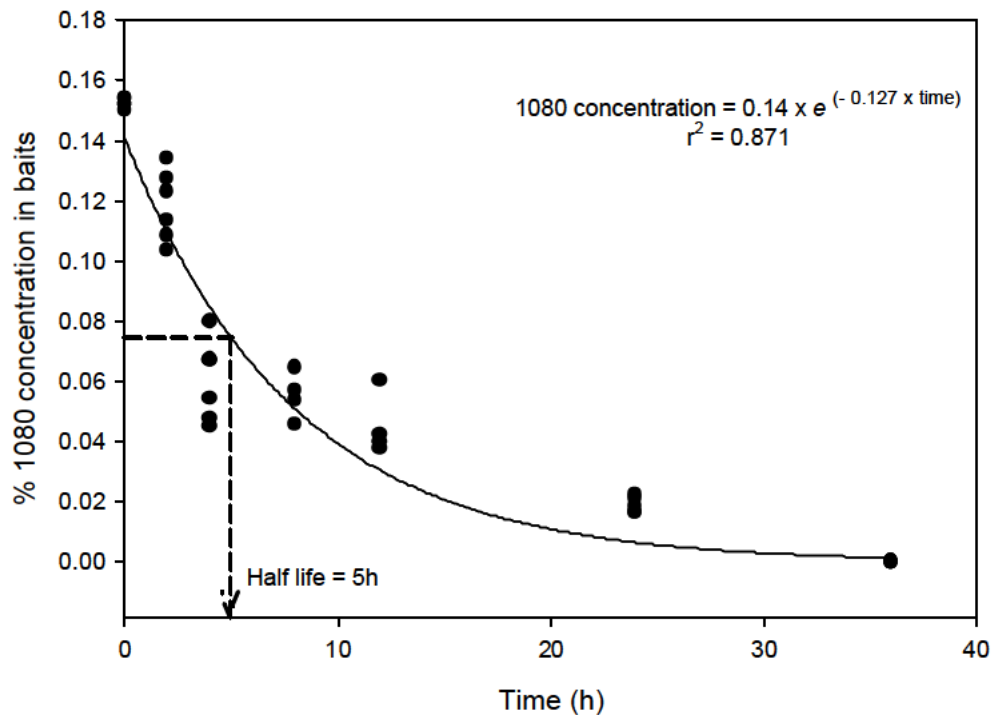


Figure 12: Graph showing the % of 1080 remaining in Wanganui No7 baits after increasing lengths of time under flowing water (15 cm s^{-1}). Also shown is the exponential decay relationship that displayed the best fit to the data, and the calculated half-life of 1080 under these experimental conditions.

If such monitoring studies are conducted $>24\text{h}$ from the time that a stream had been contaminated with 1080 baits, then there would be little chance of detecting any 1080 in the water, as most 1080 is leached within the first 12 h.

3.3. The biological effect of 1080 on native fish

There was no significant difference in fish length between any of the fish species placed in the cages above or below the 1080 baits ($P > 0.05$; Fig. 13). There was, however, a significant difference between the sizes of eels placed in the cages within each sampling location. Larger eels were inadvertently placed in cages 10 m upstream and 100 m downstream from the baits (i.e., at Sites 2 and 4), while smaller eels were placed at the site 100 m upstream from the baits and 10 m downstream (i.e., at Sites 1 and 3). Use of the smaller eels at Site 3 may have increased the likelihood of detecting an effect of 1080 leaching from baits, as smaller eels would be more sensitive to 1080 based on their lower weight.

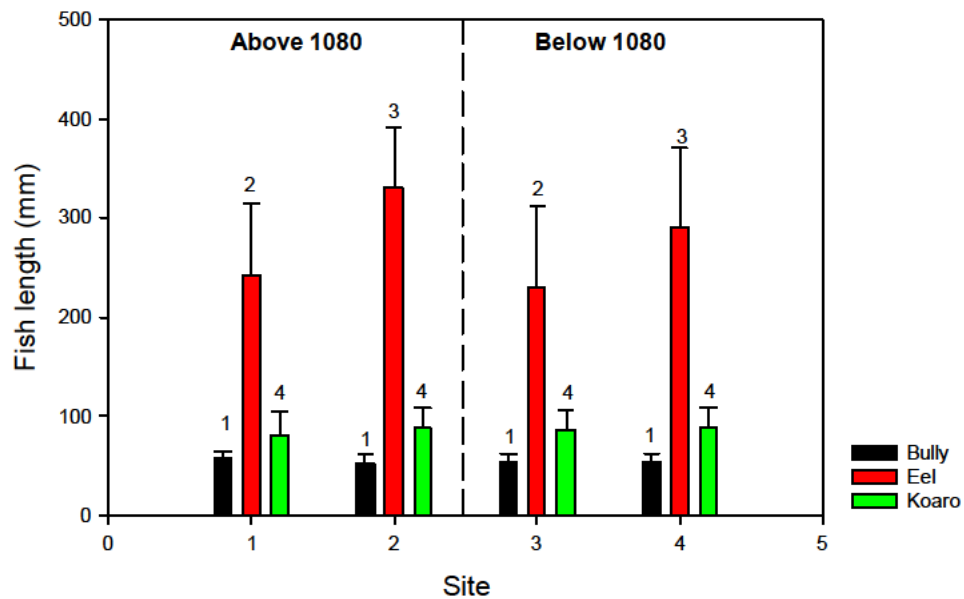


Figure 13: Mean lengths of the three fish species ($\bar{x} \pm 1\text{sd}$, $n = 8$) placed in the cages at sites above and below the 1080 baits. Numbers above each bar indicate means that were not statistically different from each other ($P > 0.05$). Note how eel length was higher at Sites 2 and 4 than at Sites 1 and 3.

No fish mortality was observed in any of the cages except at Wheelbarrow Stream, where upland bullies died between the first and second sampling occasions (Table 8). This mortality reflected the effects of heavy rainfall that occurred between these time periods, resulting in high instream velocities and transport of large amounts of fine sandy sediment downstream. The uppermost cage had been overturned by the high flows, and was partially filled with fine sand. All fish in this cage died. The lower three cages experienced less mortality, with three bullies and a single koaro dying at Site 2, and 2 bullies dying at the lower sites (Table 8). No further mortality was observed for any fish after the 1080 baits were added to any of the 4 study streams (Table 8). Some fish (most notably eels) also escaped from the cages (Table 8), with as many as 4 and 5 eels escaping from the lowermost cages placed at Deadhorse trib and Washout Creek between the 1st and 2nd observation times. Despite the escape of these animals, enough fish remained in the cages placed 10 m below the 1080 baits to be able to monitor fish mortality arising as a result of being exposed to 1080.

Analysis of water samples collected during the experiment showed that 1080 was detected in the water for a short duration after the baits were added to the streams (Fig. 14). Concentrations of 1080 in the stream were all low, and below the Ministry of Health's upper value of $2 \mu\text{g l}^{-1}$, despite the very large quantities of baits that were placed in a small area in each stream. More 1080 was recorded in the site 10 m below

where the bait was placed than the site 100 m below this point, reflecting the lower dilution of the 1080 at the upper site.

Table 8: Number of fish observed during the 1080 exposure experiment at different times and locations. Eight fish of each species of bully (b), eel (e) and koaro (k) were originally placed in the experimental cages in each of the 4 streams. Cages exposed to 1080 are shown with brown shading. Times when fish mortality were observed are shown in bold, other reductions in density were due to fish escaping from cages.

Creek	Time	Above 1080 baits		Below 1080 baits	
		Site 1 – 100m	Site 2 – 10m	Site 3 – 10m	Site 4 – 100m
		(b, e, k)	(b, e, k)	(b, e, k)	(b, e, k)
Deadhorse trib	4 days before	8, 8, 8	8, 8, 8	8, 8, 8	8, 8, 8
	1 day before	8, 8, 8	8, 8, 8	8, 8, 7	8, 4, 8
	1 day after	8, 8, 8	7, 8, 8	8, 8, 7	8, 4, 8
	4 days after	8, 8, 8	7, 8, 8	7, 8, 7	8, 4, 8
Wheelbarrow Creek	4 days before	8, 8, 8	8, 8, 8	8, 8, 8	8, 8, 9
	1 day before	0, 6, 8	5, 8, 7	6, 8, 8	6, 8, 9
	1 day after	0, 6, 8	5, 8, 7	6, 8, 8	6, 8, 9
	4 days after	0, 6, 8	5, 8, 7	6, 8, 8	6, 8, 9
Washout Creek	4 days before	8, 8, 8	8, 8, 8	8, 8, 8	8, 8, 8
	1 day before	8, 8, 8	8, 8, 7	8, 8, 8	8, 3, 8
	1 day after	8, 8, 8	8, 8, 7	8, 8, 8	8, 3, 8
	4 days after	8, 8, 8	8, 8, 7	8, 8, 8	8, 3, 8
Crooks Creek	4 days before	cages stolen	8, 8, 8	8, 8, 8	8, 8, 8
	1 day before	cages stolen	8, 8, 8	8, 8, 8	7, 8, 7
	1 day after	cages stolen	8, 8, 8	8, 8, 8	7, 8, 7
	4 days after	cages stolen	8, 8, 8	8, 8, 8	7, 8, 7

Analysis of the 1080 baits that were submerged for increasing lengths of time showed a significant decline in 1080 concentration over time (Fig. 15), so that after about 8 h, just under ½ the original 1080 in the baits had been leached. Extrapolation of the data showed that by 18 h, all the 1080 would have leached from these baits.

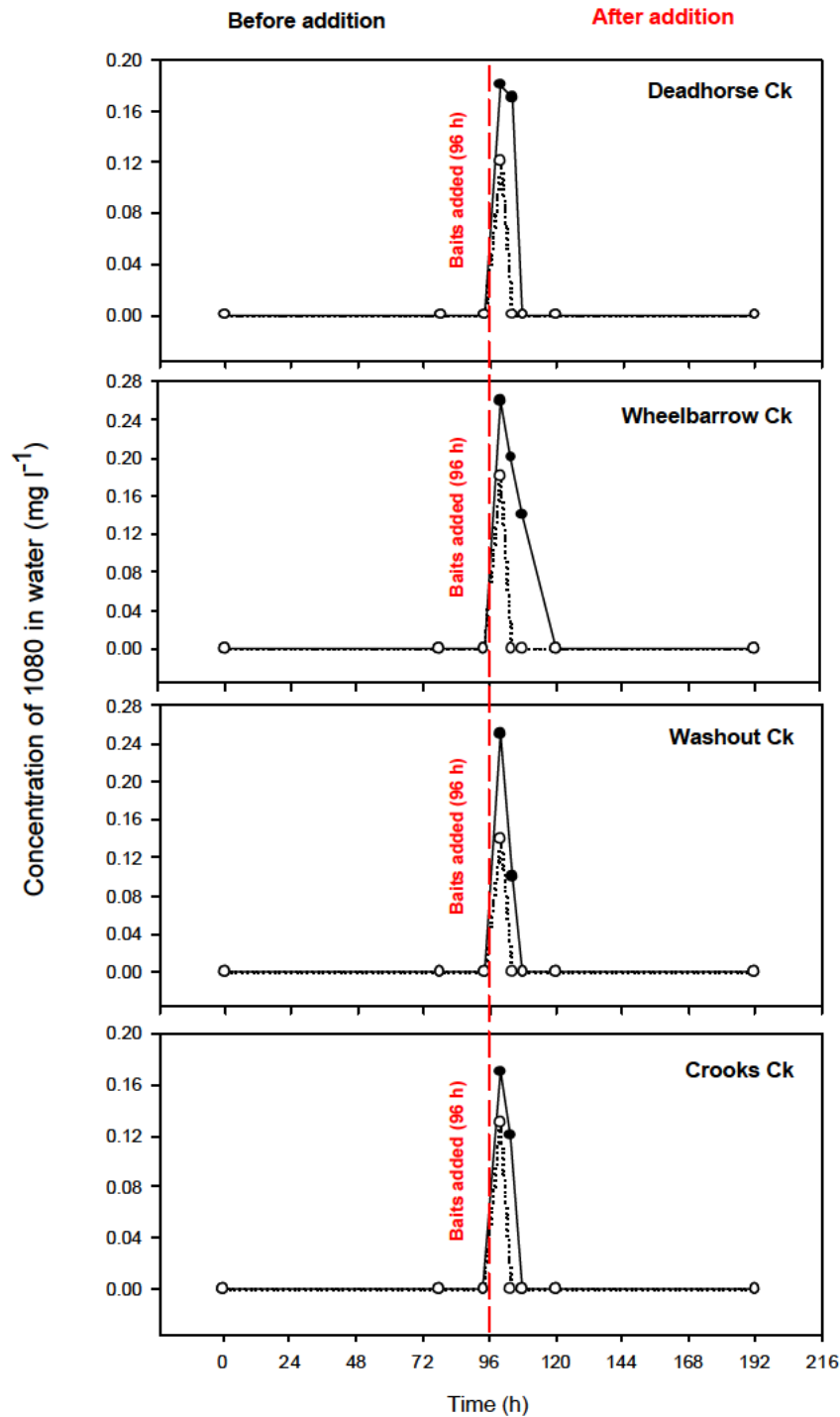


Figure 14: Concentrations of 1080 in water samples collected from the four streams during the field experiment. Samples were collected from site 3 (filled symbols) 10 m below the bait, and the site 4 (open symbols), 100 m below the baits on 2 occasions before the baits were added (at time = 0 and 78 h). Bait was placed in the streams at hours, and samples collected 2, 4, and 8 h after, as well as 1 day and 4 days later. Note that some symbols from site 3 are obscured by those from site 4.

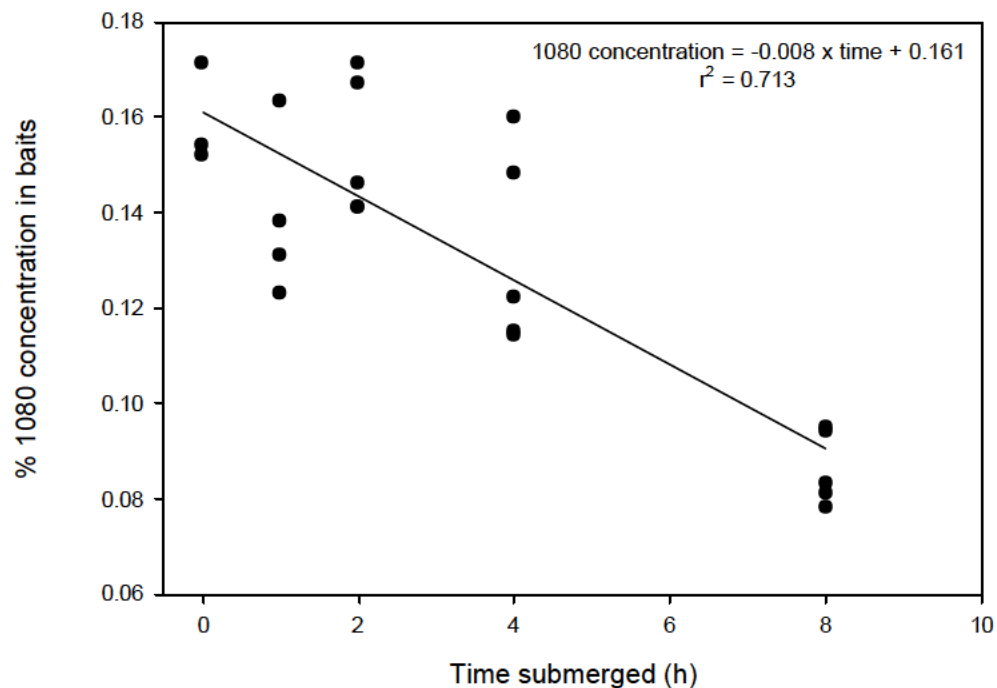


Figure 15: 1080 concentration in Wanganui No 7 baits submerged for increasing lengths of time in Deadhorse trib showing the regression line that best described the decline of 1080 in the baits over time.

3.4. The biological effect of 1080 on invertebrate communities

A total of 72 taxa were collected from the 4 streams during the study. Of the 87 889 individuals collected, the caddisfly *Helicopsyche* was the most common, contributing up to 35% to total density, followed by orthoclad midges (18%), the leptophlebid mayfly *Deleatidium* (10%), and the caddisflies *Pycnocentroides* and *Pycnocentria* (7% each). The stonefly *Zelandoperla*, and the crane fly *Aphrophilia* were also relatively common, contributing 3% to total density. Only 12 taxa were collected with densities >1%: 38 taxa were collected with densities <0.1% and could be considered rare. The most widespread taxa were *Helicopsyche*, *Deleatidium*, Orthoclad midges and *Zelandoperla*, which were collected in all sites.

Small Creek had the highest taxonomic richness (65 taxa) and invertebrate density (mean = 2755 individuals m⁻²), while Wheelbarrow had the lowest taxonomic richness (50 taxa) and lowest densities (758 individuals m⁻²).

We placed between 15 and 25 baits in the study streams, depending on their discharge. However, no 1080 was detected in the water samples collected 1 day and 4 days after baits were deployed. Absence of detectable 1080 in the streams even 24 h after bait

placement again illustrates the rapid rate that 1080 leaches from submerged baits. If we assume a 4 h leaching time of the 1080 from the baits in these streams, then the estimated concentrations of 1080 in the study would have ranged from 0.83 – 1.67 $\mu\text{g l}^{-1}$. These concentrations are in the top 0.8% of 1080 concentrations ever recorded in streams (Eason 2002), and represent what we regard as being representative of a worse-case scenario.

Despite the high loading of 1080 baits in the streams, our analyses showed no demonstrable effect on the invertebrate communities. Analysis of community similarity by PERMANOVA showed that the Bray-Curtis similarity scores did not differ significantly between samples collected on the different sampling occasions ($F = 0.85$, $P > 0.05$), or in the samples collected at the different sites ($F = 0.58$, $P > 0.05$). There was also no significant time $\times$ site effect for the Bray-Curtis similarity scores ($F = 0.95$, $P > 0.05$). The NMDS ordination of the data showed that Small Creek had a very different invertebrate community than the other three streams, and that Crooks Creek had a community that was more similar to Deadhorse trib than Wheelbarrow Creek (Fig 16). Deadhorse trib and Wheelbarrow Creek appeared to have communities that were more similar to each other. Within each stream, there was no apparent difference in the invertebrate communities collected above or below the 1080 baits, or before or after their deployment (Fig. 16).

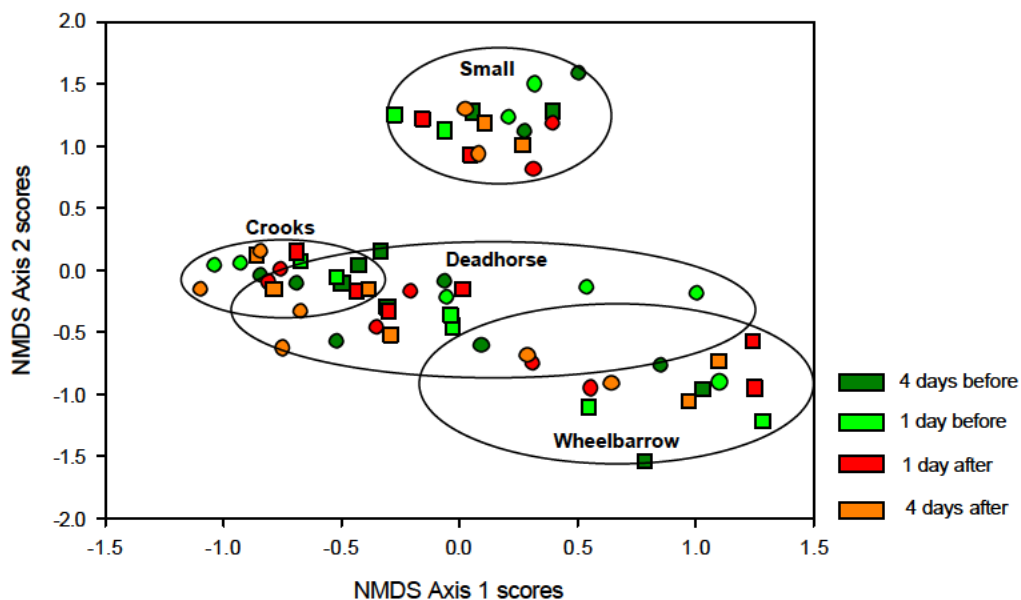


Figure 16: NMDS ordination of the invertebrate data collected in the 4 streams during the experiment showing how the communities differed more between streams than they did as a result of 1080 baits. Squares = samples collected above the 1080 baits; circles = samples collected below; timing as per legend.

The ANOVA results showed that there were significant *site* differences for 9 of the 10 metrics examined, with most metrics being higher in either the upper (control) sites, or differing between sites located 10 m and 100 m away from where the baits were deployed (Table 9). Significant *time* effects were also observed for 9 metrics: 4 metrics were higher in sites before the 1080 was applied and 1 metric (*Zelandoperla* density) was higher after the 1080 was deployed. Another 4 metrics were lower on Day 1 than Day 4 (Table 9), irrespective of where samples from were collected. Significant *site* $\times$ *time* interaction effects were observed for 6 metrics (Table 9), however examination of the other interaction terms showed little evidence of a direct effect of 1080 on the invertebrate communities. For example, the significant *site* $\times$ *time* interaction for total density reflected the fact that invertebrate densities declined after the 1080 baits were deployed (Fig. 17), but that the magnitude of this decrease was greater at the control sites (12 % of total densities here) than the impact sites (6% of densities). Both the MCI and QMCI displayed significant *site* $\times$ *time* interactions (Table 9), with the MCI increasing slightly at control sites, but decreasing slightly at impact sites (Fig. 17). However, MCI scores were very high at both control and impact sites, and the magnitude of the decrease at sites below the 1080 baits was negligible. A similar negligible change in QMCI scores was observed, with QMCI scores decreasing slightly at both control and impact sites (Fig. 17).

The % of EPT taxa was significantly lower at impact sites than control sites, and in samples collected after the deployment of the baits (Table 9). This metric displayed statistically significant *site* $\times$ *time* interactions, as the relative % of EPT taxa declined more at the impact sites (by 11.6%) than at the control sites (by 7.6%: Fig. 18). However, this difference in the reduction of %EPT taxa between the control and impact sites was very small (only 4%) and ecologically insignificant given the natural variability of invertebrate communities in streams.

Densities of the caddisfly *Helicopsyche* also exhibited a significant *site* $\times$ *time* interaction (Table 9), reflecting the fact that densities of this animal decreased more at control sites following the addition of the baits (by 47%) than they did at impact sites (a reduction of only 27%: Fig. 19). If 1080 were toxic to these animals, then densities should have declined more at the impact sites. Densities of the stonefly *Zelandoperla* increased at both sites over time (Fig. 19), although the magnitude of this increase at the impact sites (12 %) was less than increase at the control sites (60%). This lower increase at the impact sites could reflect a higher mortality of newly arriving individuals there, but given the fact that *Zelandoperla* densities still increased at these sites, any adverse impacts would be negligible at the population level. However, these results more likely reflect chance small-scale differences in invertebrate colonization dynamics between the different sites.

Table 9: Results of ANOVAs for the different metrics describing the invertebrate communities at the sites showing significant components of the ANOVA model ($P < 0.05$). Although there were some significant Site and Time effects in the model, there were only a few significant interaction effects that could be ascribed to effect of the 1080 on invertebrate communities (bold). Other effects represented chance differences between sampling sites or occasions. Most of the interactions were not statistically significant ($P > 0.05$; n/s).

Metric	Site effect (Above vs Below or 10 m vs 100 m)	Time effect (Before vs After or Day1 vs Day4)	Site $\times$ Time interaction	Above and Below $\times$ Before vs After	Spatial interaction $\times$ Before vs After	Temporal interaction $\times$ Above and Below	Stream $\times$ Above and Below $\times$ Before vs After
Density	Above > Below	Day 1 < Day 4	$P < 0.001$	n/s	n/s	n/s	n/s
Richness	10 m < 100 m	Day 1 < Day 4	n/s	n/s	n/s	n/s	n/s
MCI	Above > Below	n/s	$P < 0.001$	$P < 0.001$	n/s	n/s	$P = 0.039$
QMCI	Above > Below	Before > After	$P < 0.001$	n/s	n/s	n/s	$P = 0.003$
No of EPT	Above > Below	Day 1 < Day 4	n/s	n/s	n/s	n/s	n/s
% EPT	Above > Below	Before > After	$P < 0.001$	$P = 0.049$	n/s	n/s	$P = 0.033$
<i>Deleatidium</i>	Above > Below	Day 1 < Day 4	n/s	n/s	n/s	$P = 0.04$	$P < 0.001$
<i>Helicopsyche</i>	Above > Below	Before > After	$P < 0.001$	n/s	n/s	n/s	n/s
Orthocladinae	n/s	Before > After	n/s	n/s	n/s	n/s	n/s
<i>Zelandoperla</i>	10 m > 100 m	Before < After	$P < 0.001$	n/s	n/s	n/s	$P = 0.012$

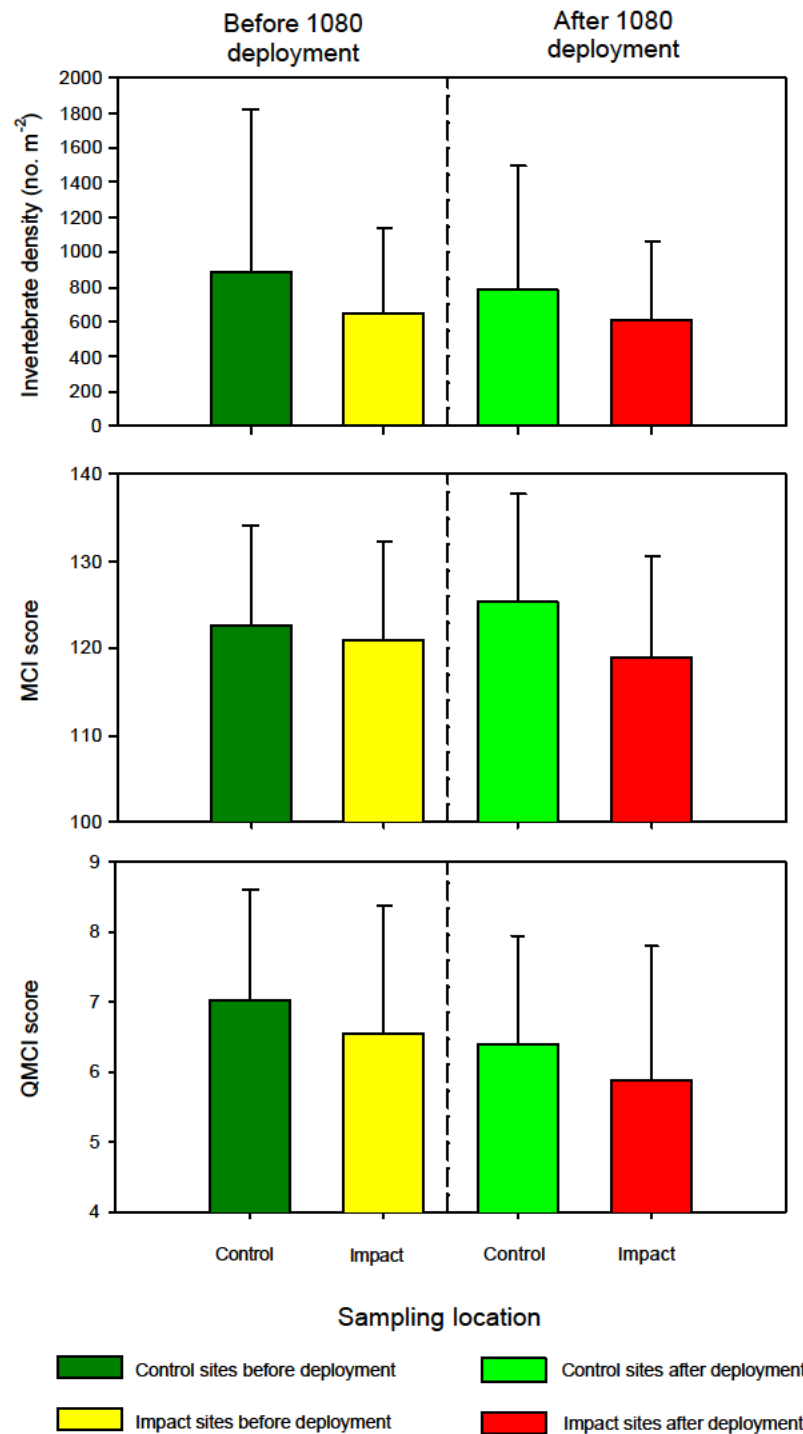


Figure 17: Graph of invertebrate density, MCI and QMCI in the 4 sampling locations ($\bar{x} \pm 1SD$, $n = 40$) during the study showing how these variables changed at the control and impact sites before and after the 1080 baits were placed in the streams. Note the lack of trends that could be attributable to the effects of 1080 on communities below the baits after they had been placed in the streams.

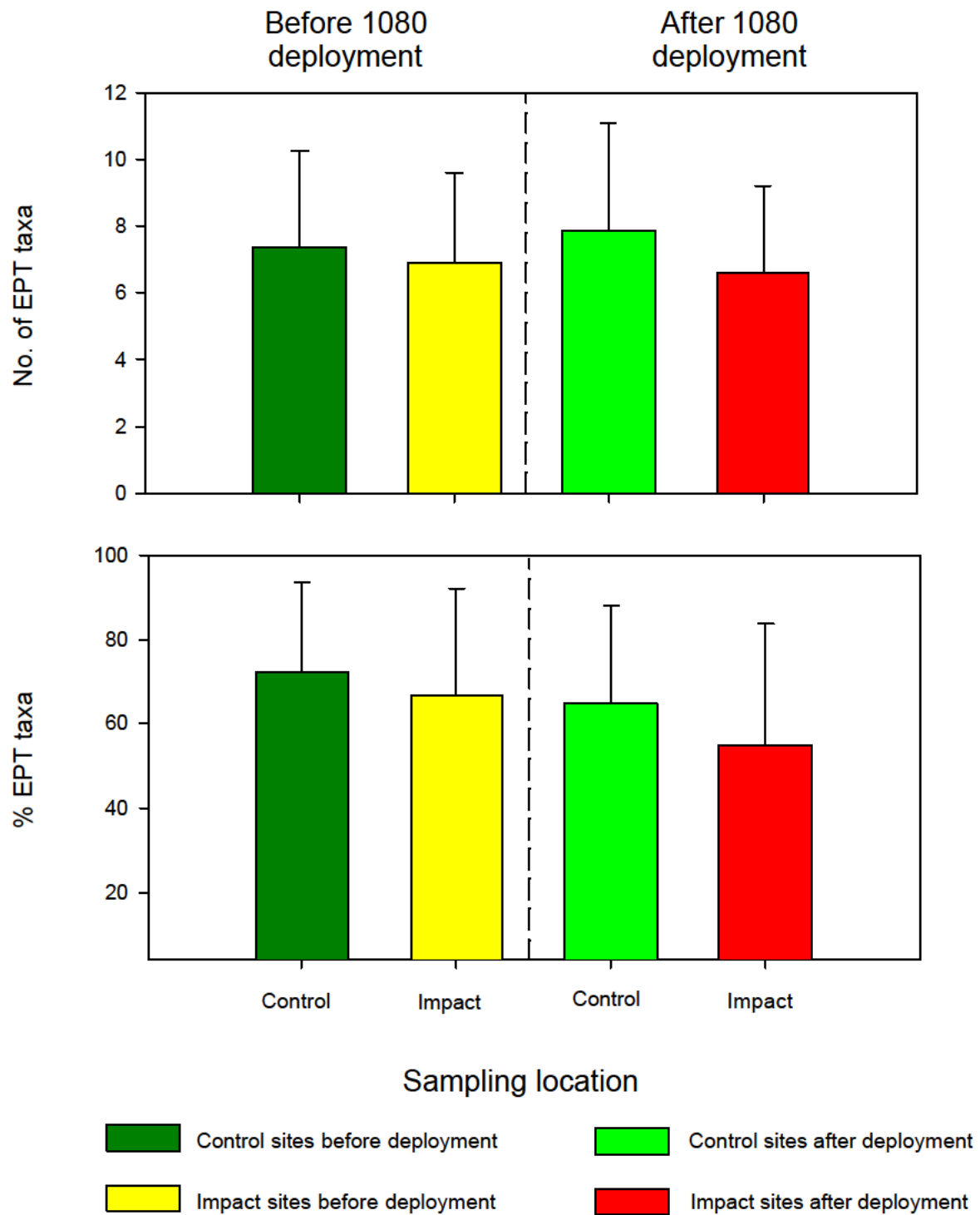


Figure 18: Graph of the number of EPT taxa, and % EPT in the 4 sampling locations ($\bar{x} \pm 1SD$, $n = 40$) during the study showing how densities changed at the control and impact sites before and after the 1080 baits were placed in the streams.

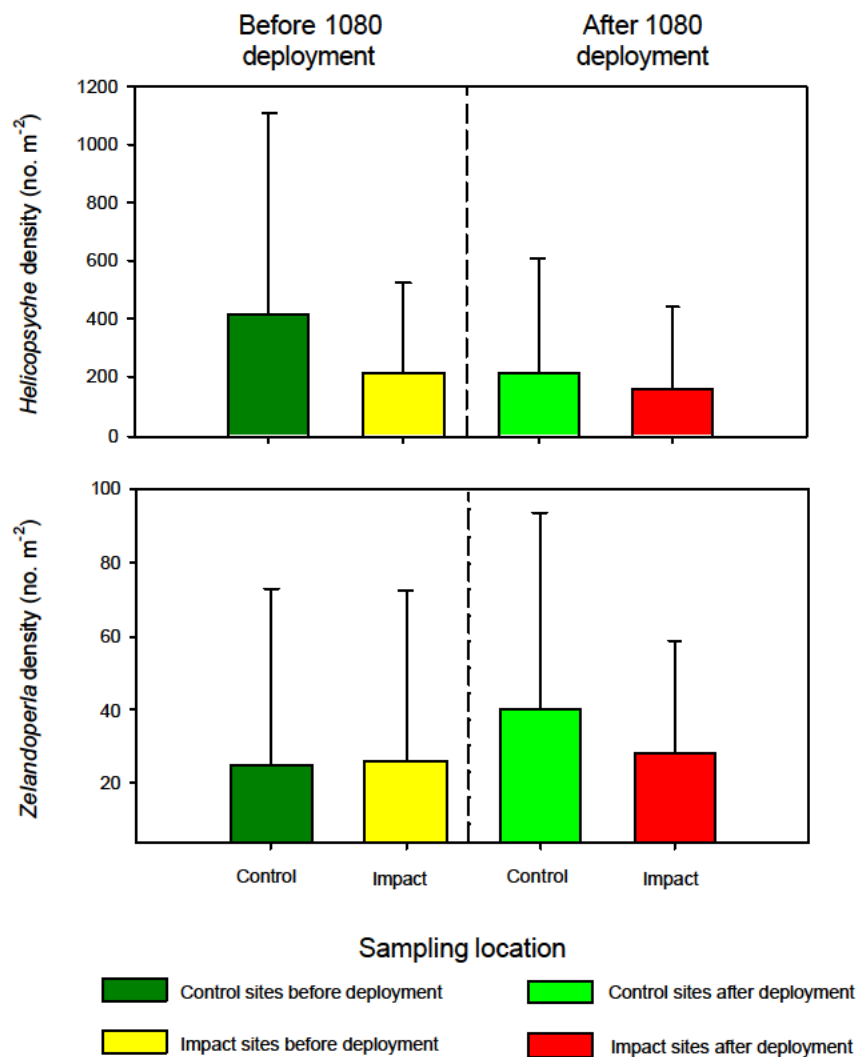


Figure 19: Graph of *Helicopsyche* and *Zelandoperla* densities in the 4 sampling locations ($\bar{x} \pm 1SD$, $n = 40$) during the study showing how densities changed at the control and impact sites before and after the 1080 baits were placed in the streams.

4. Discussion

This study was designed to increase our knowledge of the effect of 1080 baits on stream life. Although there have been many assertions that 1080 would be diluted to toxicologically insignificant amounts in streams (e.g., Eason et al 1999; Green 2003), no studies have corroborated this. Given the public concern about 1080 in the environment, and the upcoming ERMA reassessment of 1080, the results of this study are expected to be of interest to both advocates and opponents of the aerial application of 1080 baits.

4.1. Quantifying the extent of contamination

The application rates of 1080 baits are set as part of standard operational conditions, so it should have been possible to estimate the average number of baits falling into 100 m stream sections, based on the total area of streambed. However, no correlation existed between bait application rate and the number of baits found in streams, nor between stream size and the number of baits within a stream. These results suggest that the chance of baits falling into streams was random. Moreover, the distribution of baits in streams was generally non-uniform. Such non-uniform distribution could be caused by a number of reasons, such as baits being washed from high-energy areas such as riffles and accumulating in low-energy environments such as pools or behind boulders or logs. These findings have a number of implications for water sampling programmes that are used to monitor 1080 contamination in streams.

If baits randomly fall into streams, then water quality monitoring programmes may by chance be sampling streams without any 1080 baits in them. The large number of “zero” detection levels in sampling programmes to date may in part reflect absence of baits within streams. Lack of baits in streams is important as it suggests that not all streams within operational zones become contaminated with 1080. This contention is supported by observations that 5 streams in both the Lewis Pass (out of 20 streams) and Moana-Ruru (out of 11 streams) 1080 drops had no baits in them. Moreover, at the Moana-Ruru sites, the streams often flowed through steep-sided limestone gorges, and this morphology may have reduced the likelihood of baits falling into streams. Local geomorphology may thus play an important part in influencing the number of 1080 baits that fall into streams.

4.2. Fate of baits landing in water

Baits progressively lost weight over a 4-5 day period until they had completely fragmented. 1080 was also rapidly leached from baits when submerged, with a very rapid initial loss within the first 8-12 h. Almost all the 1080 had leached from the baits within 30 h in the experimental flow tank. Extrapolation of the regression curve of 1080 concentration against time from the field experiments showed that all the 1080 would have leached from the baits within 20 h. The quicker leaching rate in the field experiment may reflect the faster and more turbulent water velocities in Deadhorse trib than in the more controlled conditions of the flow tank.

The water sampling protocol used in the fish and invertebrate sampling programmes corroborated the results of the leaching experiment. In the fish programme, there was a reduction in dissolved 1080 over a 2-4 h period in three of the streams, and no 1080 was detected in samples collected 8 h after addition of baits. In Wheelbarrow Creek,

1080 was detected in samples collected after 8 h, but not after 24 h. The longer time for 1080 to leach out of baits in this creek may have been an artefact caused by the tighter ‘clumping’ of baits within the mesh bags placed in this stream, which held 27 baits. Water velocities around these baits may not have been as high as around baits placed at lower densities in the mesh bags in the other streams. In the invertebrate monitoring programme, no 1080 was detected in the water during the experiment, but the first water samples were collected 24 h after the baits had been submerged. Lack of detectable 1080 24 h after the introduction of the baits in the invertebrate study again illustrates the rapid leaching rate of 1080 from submerged baits.

This rapid leaching has implications for water sampling programmes. If the intent of sampling programmes is to quantify the potential contamination loading of 1080 in streams following an aerial drop, then sampling should occur at least within 24 h (and preferably within 8-12 h) of the stream having potentially been contaminated by 1080 baits. If a water sampling programme were commenced 24 h after a drop, then any 1080 present in submerged baits would have leached out before the sampling programme had started.

Many resource consent conditions stipulate that water sampling must be carried out 1 day (or more) after aerial operations. For instance, the West Coast Regional Council used to have conditions that require water sampling within 36-48 h after a drop. Other sampling programmes (e.g., Taranaki Regional Council 1993, 1994b; Fowles and Williams 1997) sampled streams 1-2 days following aerial operations. Not surprisingly, they found no evidence of 1080 contaminating streams in these programmes.

The importance of collecting samples on the same day as aerial drops occur was highlighted by the results of a monitoring drop conducted by ECan in the Ashley Forest near Mt Grey. Six water samples were collected for this monitoring programme, and positive results were found for 2 samples: the Kowhai River contained $1 \mu\text{g l}^{-1}$, and a small tributary into Bushy Creek contained $0.8 \mu\text{g l}^{-1}$. Fortuitously, these same streams were surveyed on the same day to count the number of baits in them. A total of 21 baits were found in the Kowhai River, near from where the water sample was collected. Discharge of this stream was approximately 30 l s^{-1} , so the estimated 1080 concentration if all 1080 baits had occurred in a small area and the 1080 had leached from the baits within 4 h would have been $0.47 \mu\text{g l}^{-1}$. This is less than what was found in the water sample, suggesting that either there were more bait in the river than we counted, or that the 1080 had leached from the baits in a shorter time. Nineteen baits were found in a tributary into Bushy Creek, close to from where the water sample had been collected. Discharge here was approximately 15 l s^{-1} , so the estimated 1080 concentration over a 4 h period would have been $0.8 \mu\text{g l}^{-1}$.

which is what the water sample contained. These results again highlight the importance of collecting water samples within a few hours of streams being contaminated by 1080 baits.

4.3. Effects on native fish

No mortality in any of the three fish species tested was observed when the fish were exposed to 1080 leaching from the baits. The only mortality that occurred in the study was for upland bullies (*Gobiomorphus breviceps*) that died as a result of flooding between the 1st and 2nd sampling occasions before the baits were added. Although native fish are regarded as being relatively tolerant to floods (Jowett 2000), it is likely that the upland bullies could not cope with conditions in the cages in Wheelbarrow Creek. The streambed here was characterised by a mixture of cobble-dominated riffles, and slower flowing runs with fine gravel and sand, into which the cages were placed. During the high flow event between the 1st and 2nd sampling trips, these fine gravels would have become mobile, and this may have resulted in mortality of some of the upland bullies. The upper cage containing upland bullies was also dislodged from its site, despite being anchored in position, and all fish subsequently died. The other fish species were more tolerant of conditions within the cages during this time, with only 1 dead koaro being found in the cages at Wheelbarrow Creek following this flood.

We placed 80 Wanganui No 7 baits in Wheelbarrow Creek, and monitored fish mortality at a site approximately 10 m below this. To give an indication of how realistic 80 baits within a 10 m block was, we compared this loading with those observed from the surveys of streams flowing in operational areas. The maximum number of RS5 baits found in a 10 m block was 8, in a stream at Mt Grey in the Ashley Forest. We therefore had applied 10 times this amount at a single location in Wheelbarrow Creek. The number of baits placed in Wheelbarrow Stream was 4 times higher than the total number of baits found in a 100 m section of stream at Mt Grey. Our application of 80 baits consequently represents an “extreme” case of contamination by 1080 baits over a short distance.

Such an extreme case would have resulted in a concentration of only 0.51 $\mu\text{g l}^{-1}$ at the site (Table 10), assuming a 4 h leaching rate. Previous overseas studies have shown that fish (bream and bass) can survive in water with a concentration of 370 $\mu\text{g l}^{-1}$ of 1080 (King and Penfound 1946): 725 times higher than the concentration that we were able to achieve in our field trials. To reach similar concentrations in streams with even a relatively modest flow of 105 l s^{-1} (the flow in Wheelbarrow Creek at the time the 1080 was added) would require the addition of 58 040 baits, or 370 kg of the 6.4 g

Wanganui No7 baits. Such a scenario could only occur if a relatively full monsoon bucket carrying 1080 baits fell into a stream.

Table 10: Estimated 1080 concentrations (assuming a 4 h leaching period) in the four streams during the fish and invertebrate studies.

Site	Study	Flow (l s ⁻¹)	Volume (l in 4h)	Number of baits added	Weight of 1080 added (mg)	Estimated concentration (µg l ⁻¹)
Deadhorse	Fish	86	1238400	50	480	0.39
Wheelbarrow	Fish	105	1512000	80	768	0.51
Washout	Fish	51	734400	30	288	0.39
Crooks	Fish	56	806400	30	288	0.36
Deadhorse	Invertebrates	14	201600	20	192	0.95
Wheelbarrow	Invertebrates	17	244800	25	240	0.98
Crooks	Invertebrates	16	230400	20	192	0.83
Small	Invertebrates	6	86400	15	144	1.67

Although we tested only 3 species of native fish to 1080, it is highly unlikely that other species of the same genera would be sensitive to 1080, especially given the reported high resistance of other fish to this compound. Consequently, there seems no risk to native fish from the accidental contamination of streams by 1080 as a result of aerial operations.

4.4. Effects on invertebrates

No effects of 1080 were detected for the naturally occurring invertebrate communities present in the 4 streams. Although fewer baits were added to the streams that were used in the fish study, stream discharge was much lower for the invertebrate study, resulting in 1080 concentrations in the streams three times higher than found in the fish study (Table 10). Despite the higher concentrations, the water sampling protocol did not detect any 1080 in the water, as the first samples were collected 24 after the addition.

Notwithstanding the large number of baits placed in the streams, none of the common taxa (from a wide range of taxonomic groups) we examined displayed any consistent change to their density that could be attributed to the effects of 1080. Neither were there any large consistent changes to other metrics such as EPT or MCI as a result of the 1080. Although we observed reductions in the MCI scores, these reductions were ecologically insignificant when compared to the natural variability of MCI scores in

the sites (ranges from 88 – 186). Interestingly, the highest MCI score (186) was found at the site immediately below the 1080 baits in Wheelbarrow Creek, 1 day after the baits were added to the stream. We also observed reductions in the QMCI scores over time, but these reductions were found at sites above and below the 1080 baits. Moreover, the magnitude of the reduction was negligible when considering the average QMCI score and its range found during the study.

Being a toxin, we expected that the densities of some taxa would have declined following addition of 1080 baits to the streams. Such a decline would have changed at least some of the metrics we calculated. This was not observed. Although there were some differences in the invertebrate metrics between sites above and below where the baits were placed, these differences were unrelated to either the time that the baits were added, or to the location of the samples above or below the 1080 baits. As such, the small differences we observed appeared to represent natural variations in invertebrate density along each of the 4 streams over the duration of the experiment. For example, total invertebrate density changed by only approximately 24% between sites above and below the 1080 baits, and the magnitude of this change was similar before and after the baits were added. This variation in invertebrate density is slightly less than that observed in pristine alpine streams near Arthurs Pass. Here, mean monthly densities varied up to 30% from the long-term mean (Suren 1991, unpublished data). We thus consider a 24% variation in invertebrate density at sites above and below the 1080 baits due to natural variability and not the presence of 1080 baits. This contention is reinforced by the fact that the density differences between the upstream and downstream sites was similar both before and after the 1080 baits were added. If the 1080 caused significant invertebrate mortality, then we would have expected a much greater reduction in invertebrate densities at the sites below the 1080 after it had been added.

4.5. Management implications

There are two important implications arising from this study. First, water-monitoring programmes designed to quantify potential contamination of surface waters by 1080 need to be undertaken within the first 8-12 h of a particular stream being potentially contaminated by 1080 baits. Within this time there is the greatest potential for collecting the short-lived pulse of 1080 that rapidly leaches from submerged baits that have fallen in streams.

Second, the intent of placing buffer strips around waterways needs to be reconsidered. There is currently no national standard that we are aware of for imposing buffers around waterways during 1080 operations. Some regional councils require buffers around all waterways, others have buffers around all waterways > 3 m in width, while

other councils have no buffers around waterways that are not used for drinking water. If the intent of placing buffers around streams is to minimise potential adverse effects of 1080 on the environment, then placing buffers on only relatively large streams >3 m in width will not prevent the smaller headwater streams from being exposed to accidental contamination of 1080 baits. Smaller streams have a much lower dilution capacity than larger ones, and often support higher invertebrate densities than larger streams, so it could be argued that buffers should be placed around these smaller streams. However, requiring buffers around all streams within an operational area would reduce the cost-effectiveness of aerial 1080 operations. This requirement would mean large areas of catchments would become no-drop zones, increasing the operational difficulty. Additionally, buffer zones around waterways may create refuge zones for the target species. Creation of such refuge zones may diminish the overall effectiveness of 1080 operations in specific areas.

Our experiments were performed in small streams all <3 m in width, which would have all been subject to potential contamination by 1080 baits during aerial operations. Despite placing very large quantities of baits into these streams (in the order of 10 times what would be expected to occur normally), no adverse effects on the natural invertebrate communities or three native fish species were found. If there were no adverse effects of 1080 on animal life in small streams where the dilution capacity is low, there would also be no adverse effects on stream life in larger streams where there is a much greater dilution capacity. Placing buffer zones around these larger streams therefore seems unjustifiable. However, the use of buffer zones around streams that are used for human drinking water or for stock supply is justified, as the public needs to be aware that all steps to prevent 1080 baits from landing in such streams have been taken.

5. Acknowledgements

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From:
Sent: Wednesday, 19 August 2020 4:45 PM
To: Martyn Cole
Cc:
Subject: Northern Taranua Livestock Map of Otaki Bore Locations
Attachments: Northern Taranua Kapiti Coast DC Map 19Aug20.pdf

Hi Martyn

Attached is a map showing the location of Kapiti Coast bores in relation to the treatment area.

The distance from the treatment area to the Hautere bores is about 10 kilometres, not 13 kilometres as stated in my previous email.

Regards

From:
Sent: Wednesday, 19 August 2020 12:20 PM
To: Martyn Cole <Martyn.Cole@kapiticoast.govt.nz>
Cc:
Subject: Northern Taranua Livestock and Taonga Protection - Research on 1080 in Water

Hi Martyn

Thanks for speaking with me today. I wish to provide you with the following information.

The Ministry of Health guidelines require us to notify operators of public water supplies within three kilometres of an aerial treatment that an operation is taking place.

The MOH guidelines specify that no toxic bait should be applied to a waterway less than 400 meters upstream from a public water way. 400 metres has been determined to be sufficient to avoid risk to human health.

The Hautere infiltration galleries near the Otaki River (those closest to the operational area) are located approximately 13 kilometres downstream of the proposed treatment area.

Attached are some research papers on the fate of 1080 in waterways and ground water. The NIWA paper on effects of 1080 on fish and stream invertebrates in West Coast Streams contains relevant information. In particular, points

3 and 6 in the Executive Summary discuss the rapid leaching of 1080 from cereal baits and that despite the large quantities of bait placed in the streams during this trial, 1080 concentrations were all lower than the Ministry of Health's PMVA.

Please note that the Ministry of Health's Provisional Maximum Accepted Value (PMAV) of 4 parts per billion represents the concentration of sodium fluoroacetate (1080) in water that, on the basis of present knowledge, is not considered to cause any significant risk to the health of the consumer over their lifetime of consumption of that water.

Within the Otaki River catchment it is proposed that cereal pellet bait containing 1080 will be applied at a rate of 1.5 kilograms per hectare. The bait has a 1080 content of 0.15%. This means that 2.25 grams of 1080 active ingredient will be applied per hectare. The relatively small amount of bait that falls into water ways will be diluted to undetectable levels due to the large volumes of water involved and the fact that 1080 is released from submerged baits within a 24 hour period.

Please let me know if you require further information.

Regards

From:

Sent: Thursday, 25 June 2020 2:30 PM

To: 'martyn.cole@kapiticoast.govt.nz' <martyn.cole@kapiticoast.govt.nz>

Cc: Subject: Notice of Proposal to Use Vertebrate Toxic Agents for TB Vector Control: Northern Taranaki Range

Dear Martyn

Epro Limited of Taupo has been contracted by TBFREE New Zealand and The Department of Conservation to undertake an animal predator control project within the Northern Taranaki Range.

Epro give notice of our intention to apply sodium fluoroacetate (1080) impregnated cereal pellet baits by helicopter. The aerial operation will commence with prefeeding in late July 2020 and will be progressively completed between then and 20 December 2020, weather permitting.

A map of the operational area is included in the attached fact sheet.

We seek Council's advice as to whether you have any concerns over public water supplies in the area. We are aware of the Otaki bore supply. This will not be affected.

Please notify us if Kapiti Coast District Council has any queries or concerns relating to this project.

Northern Tararua Combined



Aerial Predator Control 2019/20 - Kapiti Coast DC Water Supply Map

Map drawn 19 August 2020 FINAL NZTopo250 SHEET 14. SCALE:

1:110,967



From:
Sent: Thursday, 20 August 2020 8:33 AM
To: Martyn Cole
Subject: FW: Northern Tararua Livestock Map of Otaki Bore Locations
Attachments: Northern Tararua Kapiti Coast DC Map 19Aug20.pdf; AOCC 2D - Water Supply Agmt Kapiti Coast DC.pdf

Good morning Martyn

Attached is a new version of the Water Supply Agreement for the Otaki town water supply intakes. Once you have had an opportunity to read through the research material I sent you yesterday, could you please sign the Water Supply Agreement, indicating the mitigation measures you require.

Kind regards

From:
Sent: Wednesday, 19 August 2020 4:45 PM
To: Martyn Cole <Martyn.Cole@kapiticoast.govt.nz>
Cc:
Subject: Northern Tararua Livestock Map of Otaki Bore Locations

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Regards

WATER SUPPLY AGREEMENT

FORM 2D

Operation Name(s)	Northern Taranaki Predator Control		
Operation Number(s)	SUP01, SUP02, SUP03	TMA Code	N/A

To: Public Health Unit

Landowner Name	Kapiti Coast District Council, Martyn Cole Water & Wastewater Asset Manager	Phone	027 5554 751
Address	Kāpiti Coast District Council Tel 04 296 4863 Martyn.Cole@kapiticoast.govt.nz www.kapiticoast.govt.nz		
Signature		Date	

I, Martyn Cole have been provided with information by Epro Limited of Taupo on the proposed aerial 1080 operation in the catchment of the Otaki River.

I confirm that the New Zealand Transverse Mercator (NZTM) coordinates these water supply intakes are correct.

Water Supply Name	Type of Water Supply (Stream, Spring, Bore, Roof)	NZTM Coordinates
Hautere Bore KWSN036506	Otaki River	1785538 E 5480056 N
Hautere Bore KWSN036507	Otaki River	1785559 E 5480033 N
Hautere Bore KWSN027648	Otaki River	1785555 E 5480026 N
Rangiuru Road Bore KWSN030196	Otaki River	1779450 E 5486303 N
Rangiuru Road Bore KWSN036916	Otaki River	1779186 E 5487395 N
Rangiuru Road Bore KWSN036931	Otaki River	1779150 E 5487420 N
Rangiuru Road Bore KWSN027677	Otaki River	1782315 E 5485445 N
Rangiuru Road Bore KWSN027678	Otaki River	1782337 E 5485462 N

The Otaki town water supply does not have storage capacity.

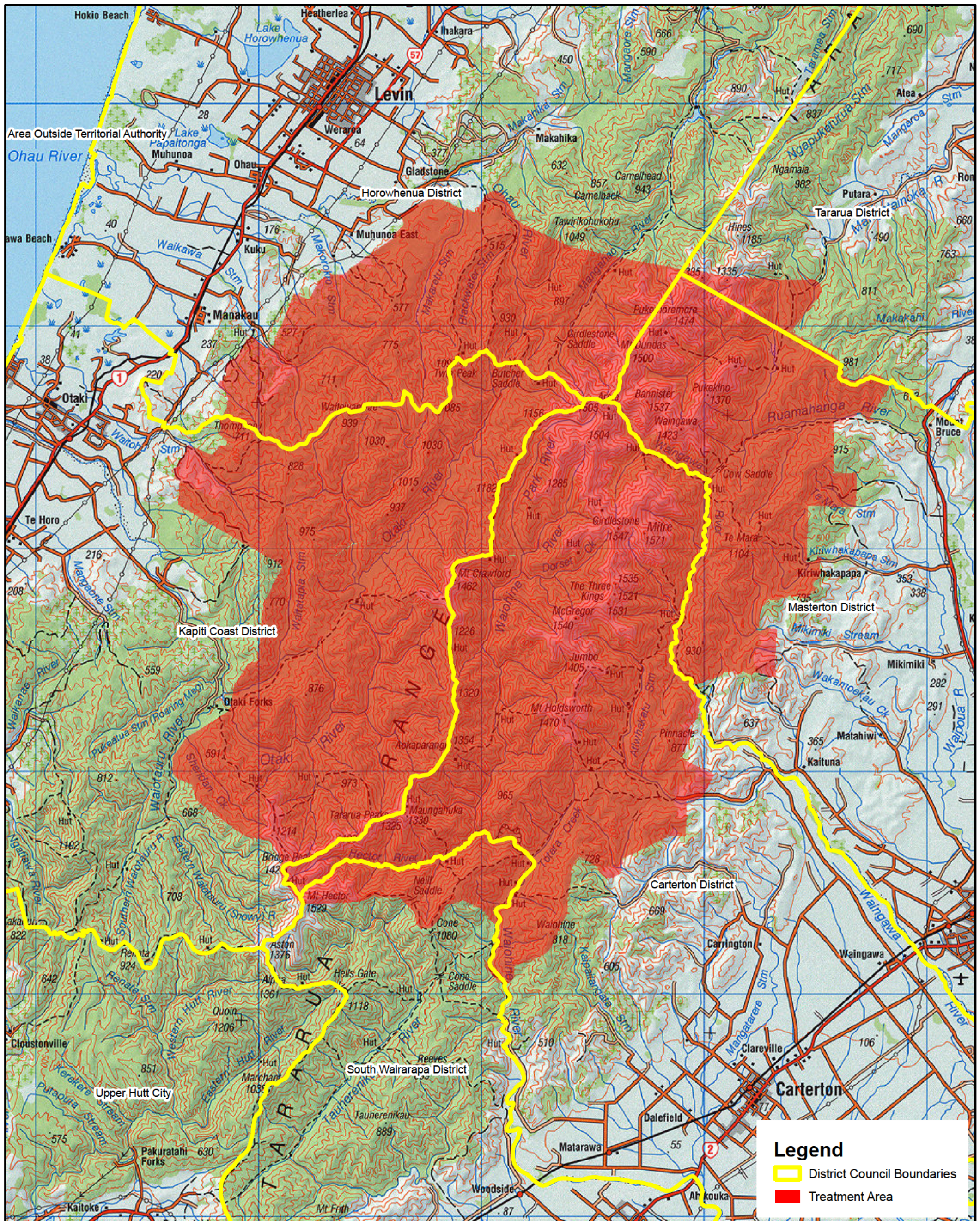
My water supply can be temporarily disconnected.

No

I will require Epro Limited to supply me with the following mitigation measures.

- ☐ No alternative water or water test, as my supply will not be affected by this operation.
- ☐ No alternative water or water test, on condition an exclusion is placed around my supply location.
Exclusion dimensions to be _____ metres either side of the intake for a distance of _____ metres upstream.
- ☐ A water test only of my water following the operation.
Water test sample to be collected _____ hours after the application of toxic bait.

Northern Tararua Combined



Aerial Predator Control 19/20 - District Council Map

Map drawn 25 June 2020 NZTopo250 SHEET 14. SCALE:

1:156,919

