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native riparian vegetation (Waikato 10) had a fauna that was very similar to that found in the remnant bush gullies in Taranaki. Unfortunately, there are very few such remnants in the highly modified Waikato landscape.

4.4.2 Separating the effects of riparian vegetation type and stock access on springs

The broad-scale survey of Scarsbrook & Haase (2003) indicated associations between community composition and riparian land use, but regional differences, and other sources of variation (e.g. variation in substrate type, underlying geology, stock access) precluded strong inferences on land-use effects. During the summer of 2004, a separate, complementary study was carried out to identify the interacting effects of type of riparian vegetation (native bush v. pasture) and stock access on springs. Sources of variation in other factors were minimised by selecting study sites within a particular hydrogeological setting (i.e. rheocrene springs arising along a narrow elevation gradient along the base of the western Kaimai Ranges; Fig. 21). Springs were chosen to reflect four different land uses:

- Native forest riparian zone with no stock access (NNS; n = 4)
- Native forest riparian zone with stock access (NS; n = 3)
- Pasture riparian zone with no stock access (PNS; n = 4)
- Pasture riparian zone with stock access (PS; n = 4)

Invertebrate community composition varied significantly with riparian vegetation type (i.e. native v. pasture), whereas stock access appeared to produce additional, cumulative effects. There was evidence of a disturbance gradient across the four treatments, with the numbers of taxa unique to a particular treatment decreasing from 20 at native sites without stock access, through to only six at pasture sites with stock access (Fig. 22A).

Relative abundance of the dominant taxon—an indicator of community stress (Barbour et al. 1999)—increased along the disturbance gradient (Fig. 22B). The chironomid *Polypedilum* sp. was the dominant taxon overall, making up 20% on average of all samples. The relative abundance of EPT individuals decreased from an average 44% at native forest springs without stock access, through to only 2% at pasture springs with stock access. Mayflies (e.g. *Zephlebia dentata*, *Z. nebulosa*) and stoneflies (e.g. *Austroperla cyrene*) were major components of the fauna in the native forest springs, whereas amphipods, chironomids and molluscs dominated the pasture springs.

In a study in Switzerland, Zollhöfer (1999) compared the invertebrate faunas of natural springs with those of springs grazed by cattle. He found significant differences in taxon richness between the two spring types. One distinct pattern was that mayflies were absent from springs with unlimited cattle access. In the present study we found a strong land-use effect on the relative abundance of leptophlebiid mayflies. The relative abundance of leptophlebiid mayflies averaged 24% in springs in native forest with no stock access, and was similar at native forest sites with stock access (20%), but decreased to 7% at pasture sites with no stock, and to less than 1% at pasture sites with stock access. Therefore, we suggest that riparian vegetation type may be the principal determinant of mayfly abundance in our Kaimai springs, but stock access can cause additional losses, particularly in open pasture springs.

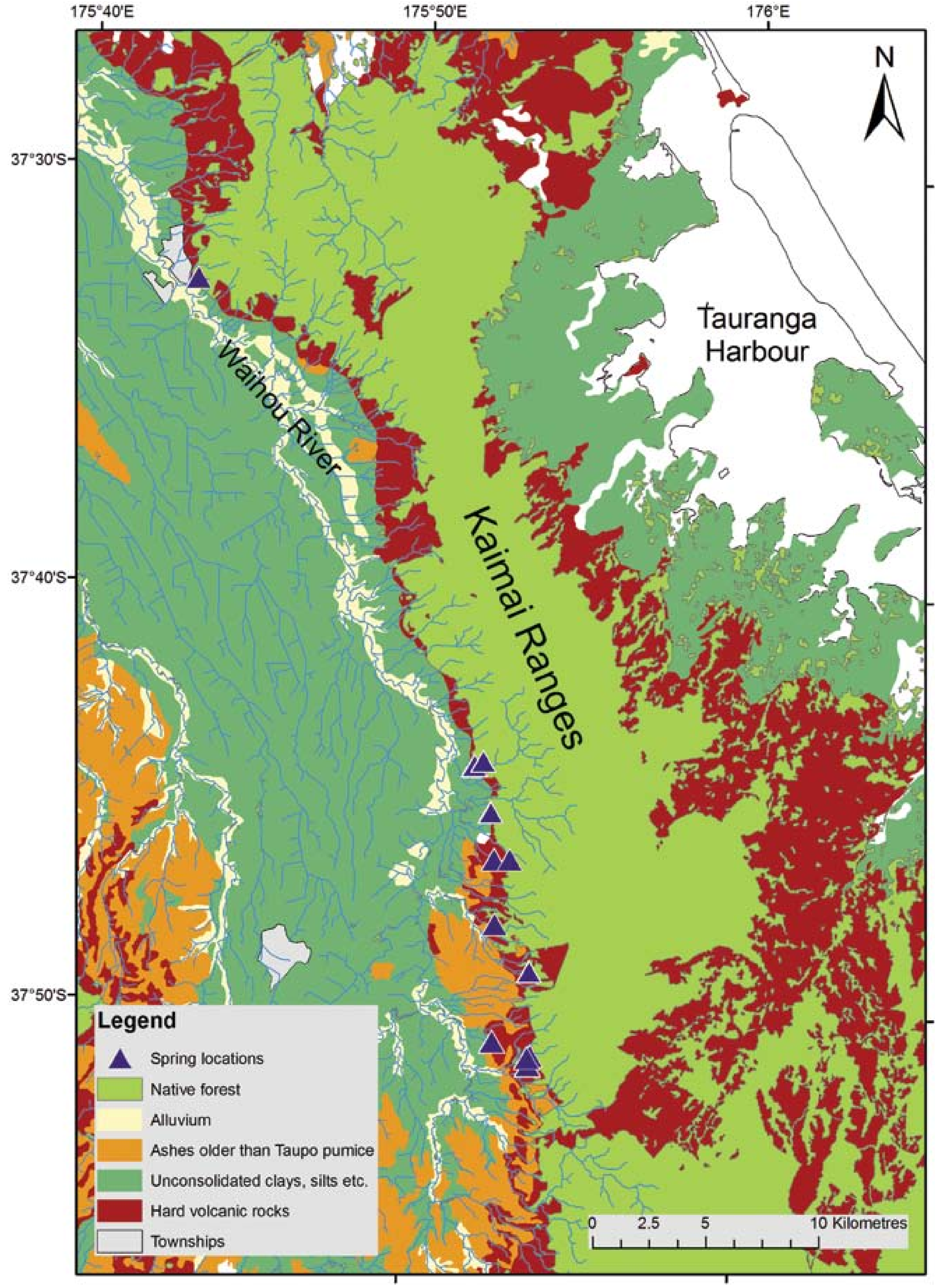

Figure 21. Locations of 15 springs along the base of the Kaimai Ranges.

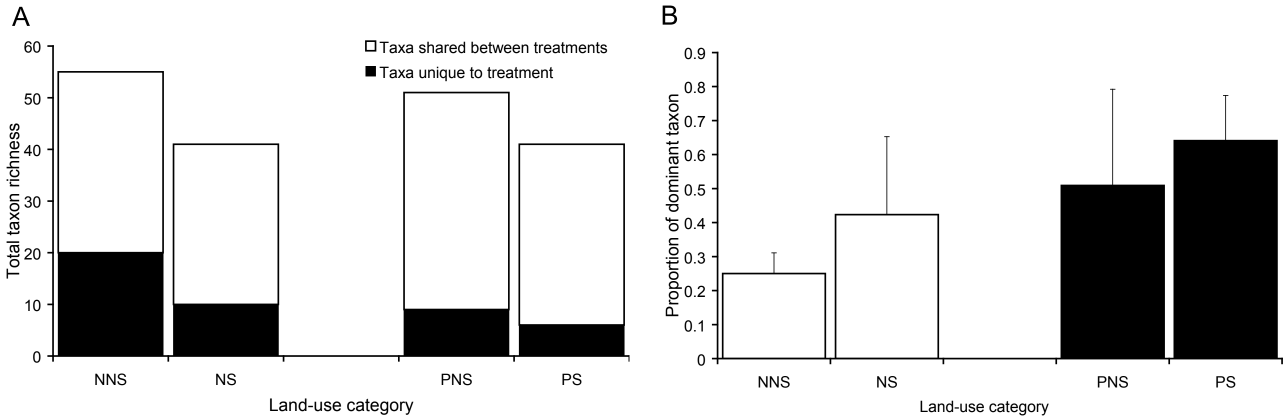

Figure 22. (A) Taxon richness patterns across springs in different land-use categories; (B) Proportions of dominant taxons across springs in different land-use categories.

5. Anthropogenic threats to springs

As ecotones, springs integrate ecological processes occurring in groundwater, surface water and terrestrial ecosystems. However, this integrating property also makes springs particularly vulnerable to impacts caused by a range of direct and indirect human activities (Fig. 23). The range of threats has been summarised by a number of authors (e.g. Danks & Williams 1991; Smith 2002). Water permanence is a key factor influencing the number of crenobiontic species, endemism and biodiversity in spring habitats (Danks & Williams 1991; Erman & Erman 1995). Thus, many threats to the integrity and biodiversity of spring ecosystems are related to the reduction of their flow. Groundwater abstraction for mining and pastoral purposes is the major threat to the mound springs of the Great Artesian Basin (GAB), Australia, and has already lead to the extinction of many springs and their associated flora and fauna (Fatchen 2000; Ponder 2002). Land drainage for agricultural and forestry activities has caused a fall in water table levels which, in turn, has reduced the density of springs to less than half on the Swiss Plateau and in the Jura Mountains (Zollhöfer 1999). In the USA, water extraction for municipal and irrigation uses has reduced springs to approximately 15% of historical levels in the Funeral Mountains, Death Valley (Erman 2002), and water abstraction for municipal supply is the major threat to numerous springs of the state of Florida (Hartnett 2000). Modification of surface flows can also alter spring flow. For example, canalisation and regulation of rivers can reduce the inter-connectivity between surface flows and groundwater storage (Hancock 2002) and negatively affect spring flows, especially of alluvial springs. Urbanisation upstream from a source of the Avon River/Otakaro, Canterbury, has produced a downstream shift of its springs as land development advanced (Marshall 1973). It has been suggested that water extraction impacts on spring flows may be a greater problem in valley and desert systems, which depend largely on artesian-regional aquifers, than in mountainous areas (Sada et al. 2001; Erman 2002).

The value of natural spring waters is illustrated by the growth of the bottled water industry, both in New Zealand and internationally. However, Smith (2002) suggested that the capture of springs for bottled water operations is a significant threat, worldwide, to their integrity and persistence.

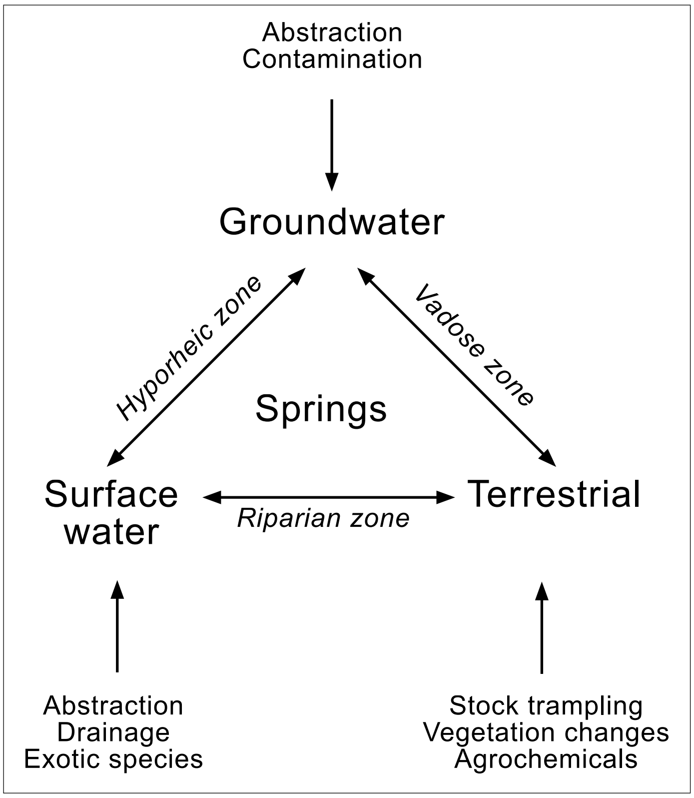

Figure 23. Diagram showing how springs integrate anthropogenic impacts on groundwater, surface water and terrestrial ecosystems.

Spring water quality largely depends on the underlying geology and the land uses of the recharge basin (van der Kamp 1995). Contamination can occur from point sources such as septic tanks (e.g. Blue Spring, Florida) (Hartnett 2000), or from diffuse sources such as the application of agrochemicals (e.g. springs of the Wairau River plains) (Young et al. 2002). Additionally, heavy metals and other chemicals associated with mining can pollute groundwaters and aquifers (Hancock 2002), and are the cause of water contamination in some western USA springs (Sada et al. 2001). Another source of pollution can be road and urban runoff. For example, water quality in Sulphur Springs, Florida, was seriously impaired after the city of Tampa directed stormwater into sinkholes connected to the spring (Hartnett 2000).

Removal of riparian vegetation affects invertebrate community composition in New Zealand springs (Scarsbrook & Haase 2003) and Californian (USA) springs (Erman 2002). The absence of riparian cover changes stream energetic budgets (Quinn 2000) and temperature patterns (Collier et al. 2001), and can produce algal and macrophyte blooms as more light reaches the channel (e.g. springs of the Wairau River plains) (Young et al. 2002). Elimination of riparian forest also destabilises banks and diminishes the filtration properties of the riparian zone, which contribute to increased nutrient and sediment loads from surface run-off (Quinn 2000; Parkyn & Wilcock 2004). Finally, logging activities have the potential to damage the physical environment of a spring, by increasing the sediment load reaching the spring channel (Erman & Erman 1995; Zollhöfer 1999; Erman 2002), and reducing woodland environments, which can provide shelter, feeding and mating areas for some adult aquatic insects (Collier & Smith 1998, 2000).

Livestock affect streamside vegetation, stream channel morphology, shape and quality of the water column and the structure of stream-bank soil (Kaufmann & Krueger 1984; Fleischner 1994). In arid regions, cattle grazing changes riparian vegetation community composition (e.g. at mound springs) (Fatchen 2000; Ponder 2002), and overgrazing can virtually eliminate all riparian vegetation (e.g. springs in the western USA) (Sada et al. 2001; Erman 2002). Moreover, livestock trampling erodes spring banks, and degrades habitats for aquatic flora and fauna by filling the interstitial spaces around rocks and gravel and by compacting mud and clay (Minckley & Unmack 2000; Sada et al. 2001; Erman 2002). In contrast, it has been suggested that coarse substrate (cobbles, boulders and bedrock) may provide some protection from stock trampling in some New Zealand springs (Scarsbrook & Haase 2003).

Piping and diversion of spring flows is one of the most common threats to the physical diversity of spring habitats (Shepard 1993; Erman & Erman 1995; Cianficconi et al. 1998; Fatchen 2000; Sada et al. 2001), and can reduce or eliminate flows altogether (Erman 2002). Many high-discharge springs are sites for swimming, fishing and camping (Shepard 1993; Hartnett 2000). The recreational use of springs often involves capture of the spring and water diversions for aesthetic reasons, damming of spring pools, removal of riparian vegetation, compaction of soils and usage of chemicals such as chlorine (van Everdingen 1991; Shepard 1993; Zollhöfer 1999; Sada et al. 2001).

All but one of the 12 spring-associated hydrobiid species described from artesian springs in Queensland by Ponder & Clark (1990) were considered to be endangered because the springs they reside in had no conservation status and are threatened by pastoral activities and extraction from parent aquifers. Ponder & Clark (1990) suggested that the threats posed by stock access were less serious than those posed by the land managers themselves. Damage by stock may be short-term, whereas activities of land managers (e.g. damming, digging out, over-pumping of an aquifer) may result in the complete disappearance of a spring. Ponder & Clark (1990) noted that many of the springs surveyed by Habermehl (1980) in the late 1970s no longer existed when visited in 1984. In New Zealand, our diverse hydrobiid fauna may similarly be threatened by land-use intensification. Fig. 24 shows a small spring arising from limestone hills in central Southland. This spring contained two new species of hydrobiid snail (Scarsbrook & Haase 2003), yet was open to trampling by stock.

Introduced species can also have a significant impact on the biodiversity and ecological function of springs. For example, vertebrates such as the mosquitofish and many plant species recognised as noxious weeds are now found in springs throughout the world (Shepard 1993; Minckley & Unmack 2000; Sada et al. 2001; Ponder 2002; Young et al. 2002). Exotic plants reduce overall plant and animal diversity and alter site hydrology, whereas non-native fishes, crayfish and toads may reduce or even extirpate native aquatic species (Sada et al. 2001; Ponder 2002; Young et al. 2002).

Because springs have groundwater sources, spring water quality is obviously affected by activities that impact groundwater quality (see Close et al. (2001) for a review). This suggests that the fauna of springs may have potential as relatively inexpensive indicators of groundwater quality (Williams & Danks 1991). Indeed, springs and their fauna may act as indicators of both the sustainability of groundwater abstraction and groundwater quality.

5.1 IMPACT OF RIVER MANAGEMENT AND REGULATION ON BRAIDED RIVER SPRINGS

The springs of braided rivers, and their fauna, are dependent on the spatial and temporal dynamics of flood-plain elements, and connectivity within the braided river flood plain (Gray 2005). Different habitat types (e.g. springs, springbrooks, side braids and main channels) are maintained in more-or-less constant proportions by the natural flow regime that drives the shifting mosaic of flood-plain elements (Arscott et al. 2002; van der Nat et al. 2003; Hauer & Lorang 2004). Anthropogenic activities such as diversion, channelisation and impoundment can have severe impacts upon the relative proportions of these flood plain elements, and thereby pose a significant risk to discrete habitats such as springs.

Many large New Zealand rivers have been channelised to create farmland and prevent river migration (Young et al. 2004). Constriction of the active river channel can cause changes in local aggradation and degradation, which can affect the channel's interactions with the aquifer and thence water supply to springs. A 0.5-m drop in the bed of the lower Motueka River was predicted to reduce summer aquifer recharge by 24% (Young et al. 2004). Furthermore, disconnection of the river from its flood plain tends to reduce habitat heterogeneity at the landscape scale and alter successional dynamics within existing flood-plain habitats (e.g. springs). Following the construction of flood control barriers, extant springs are likely to have a reduced probability of disturbance, and the formation of new springs by the reworking of alluvial gravels and river channel migration is likely to be reduced. Spring-fed habitats therefore will tend towards later successional stages (see section 4.2), with subsequent implications for biodiversity across the riverscape.

The effects of flood retention works are not universally negative. In the lower Selwyn River/Waikirikiri, flood retention practices have included both the construction of retention banks and planting of riparian willows to constrict the active channel. Spring habitats often occur at the head of

Figure 24. A small Southland spring showing obvious damage from stock trampling. Despite the damage from stock, the spring contained two new species of hydrobiid snail.

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