Invertebrate communities of the Cass, Ashley & Aparima Rivers & methods for monitoring impacts of management and environmental change

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BRRI Project 3387/1042

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Cover: Selection of invertebrates collected in pitfall and malaise traps on the Cass, Ashley and Aparima Rivers. *Photo: Jessica Chen*

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Contents

1 Summary

Although there are effective aquatic sampling methods to detect and monitor some freshwater invertebrates (E.g. EPT monitoring to indicate stream health, eDNA to estimate diversity) the invertebrate biodiversity values of the wider river braid plains, and how to monitor their response to management and environmental change, are very limited. This project, jointly resourced by the Department of Conservation and Environment Canterbury under the Braided River Research Initiative Programme, aimed to 1) test a set of standardised land-based monitoring protocols, 2) increase our understanding of braided river invertebrate biodiversity, and better understand invertebrate responses to 3) weed management and 4) flooding.

Invertebrate sampling was undertaken on the Cass, Ashley | Rakahuri and Aparima rivers using malaise and pitfall traps between 2019 and 2023. Traps were opened for one week per month from November to February annually. The Aparima River (Southland) was sampled for two years (2019-2021), and the Cass (Mackenzie) and Ashley (Canterbury) for four. Three sampling lines, each comprising one malaise trap and five pitfall traps, were used on the Cass, a small, relatively weed-free upland river. On the Ashley, a weedy but multi-channelled lowland river, three lines were installed in an area where mechanical weed management ('ripping') had been undertaken, and three in an area of no weed management. On the Aparima, an almost single-channel and weedier lowland river, six lines were installed in unmanaged areas. Three of these lines were within an area with a gravel extraction consent, with the intent to provide baseline data for a future before/after extraction comparison. As gravel extraction has not yet occurred this comparison has yet to be made.

Key results:
Over 91,000 specimens > 2mm in length were identified from the three rivers over the duration of the study representing 27 invertebrate orders and 1,099 morpho-species (Aparima = 662 species, Ashley = 700, Cass = 533). Mean species richness per line was highest for the Aparima ($\bar{x}$ = 80 species), and 20% and 25% lower for the Cass ($\bar{x}$ = 64) and Ashley ($\bar{x}$ = 60) respectively. Both Shannon and Simpsons diversity were also slightly higher on the Cass than Ashley. On average, pitfall traps detected fewer species per line compared to malaise traps (Cass = 65% fewer, Ashley = 66%, Aparima = 58%). On the Cass and Ashley in years 1-3, species richness detected with malaise traps tended to increase each month to peak in February, but in year 4 it peaked in December. No such pattern was detected from pitfall trap data.

For all rivers, flies accounted for >1/3 of species detected and 40-60% of all specimens caught. Chironomid flies alone accounted for 42%, 20% and 7% of specimens captured on the Ashley, Cass and Aparima respectively. Hymenoptera (bees, wasps and ants), moths, beetles, and spiders were moderately diverse, with each group representing 8-18% of species on each river. Higher total diversity on the Aparima was partly driven by larger numbers of non-insect arthropods as there were no detections of slaters, exotic segmented worms, slugs, snails, or flatworms on the Cass, or of landhoppers or sausage millipedes on the Cass or Ashley. This may reflect the higher prevalence of exotic grasses, weeds and associated silt, soil and litter on the Aparima relative to clean gravels. The proportion of species represented by flies, caddisflies and sucking bugs on the Aparima was similar to the other rivers, but their proportional abundance was lower, while the abundance of orthoptera, specifically field

crickets, was much higher. For all rivers, around half of species for which feeding guild could be determined were predators (feeding guild ratio = 1 detritovore : 4 herbivores : 5 predators).

Community composition clearly varied by river and trap type, but spatial and temporal variation within rivers was more limited. Half of all species were detected from only one river, and the proportion of unique species per river was relatively consistent (Aparima = 33%, Ashley = 28%, Cass = 24%). Only 34% of species were found using both trap types, 51% with only malaise and 15% with only pitfalls. On the Aparima, there was a small difference in community composition between the upper and lower three sampling lines, particularly when using pitfalls, but there was little evidence for spatial differences on the Cass or Ashley. Annual variation in composition was most prominent on the Ashley, with moderate differences detected between the first and last two years using both trap types. Malaise trapping also found differences in the communities present in months before vs. after Christmas at all three rivers.

On the Ashley, there was no evidence that weed management by mechanical ripping affected invertebrate abundance, diversity or community composition. However, regular flooding likely re-set the weed load at all sampling lines on an annual basis, such that the intended weed 'treatments' quickly became invalid. Flooding itself did not appear to have consistent or long-term impacts on invertebrate diversity or abundance. After two of five moderate-to-large summer flood events, sampling two weeks later detected a significant but temporary increase in the abundance of chironomid flies, which would be consistent with the increased presence of sediment and decaying organic matter. However, neither malaise or pitfall sampling detected a response in species diversity to these events or showed any clear correlation with either high or low flows over the full study period. Overall, our data suggests that, like aquatic invertebrates, the terrestrial invertebrate community recovers very quickly after flooding.

Occupancy modelling with Ashley and Cass pitfall data indicated that the median probability of detecting a given species at least once in a season was < 0.3 at the sampling effort used, but for the most common species could reach almost 0.9. Adding traps or lines increased detection probabilities but gains diminished with each addition. Achieving a median probability of > 0.5 for detecting any species present required at least 5 sampling lines on the Cass and 7 on the Ashley; at this sampling effort common species would be detected with a probability of 0.8-1.

Monitoring recommendations:

The tested monitoring design is ready to be applied to other rivers to assess baseline biodiversity values and trends. The design can be easily modified to fit logistical or financial constraints or to address different questions such as population trends for specific subsets of invertebrates. Using malaise and pitfall methods provides coverage of most braid plain habitats but may miss sessile invertebrates. For baseline studies, 5 to 7 sampling lines are recommended. Sets of 2 or 3 lines at intervals of 5 km or targeted to specific habitat types should be sufficient to capture spatial variation in the invertebrate community on longer rivers. If processing time is a constraint, reducing malaise traps down to 1 for every second line is possible, although one session with a full complement of traps will aid extrapolation and interpretation. For baseline diversity, sampling over 4 months provides good coverage of seasonal variation in species emergence. For trend studies, sampling in the same months annually is critical, but could be reduced to one month before and one month after Christmas (E.g. November and January). Sampling for 5 nights is practical and dampens weather effects. To determine diversity values and trends, specimens must be identified to morpho-species. Naming these at least to family level will allow ecological inferences to be drawn. Identifying juveniles and specimens < 2mm is not recommended.

2 Background

Invertebrates make up a significant proportion of all biodiversity, especially in braided river habitats where plants cover is relatively low. Insects, spiders and other invertebrates are innately important given their extremely high levels of endemism in New Zealand, but they also have a range of ecological functions including as food for fish and for native birds that feed both from the land and water. Despite this, standardised methods for measuring and monitoring invertebrates are almost entirely limited to the aquatic parts of braided river ecosystems, limiting our ability to measure important components of the community and to assess broad biodiversity values and trends.

This project used land-based sampling methods to assess the invertebrate communities of three braided rivers over the summer months from 2019 to 2023 and was funded by Environment Canterbury (ECan) and the Department of Conservation (DOC) under the Braided River Research Initiative joint agreement. The initial aim of the project (ECan ref: Project 3387) was to test and refine invertebrate monitoring methods previously developed by DOCs Project River Recovery programme on the Tasman River (Murray & Anderson, 2019) and use these methods to characterise the invertebrate diversity, abundance, and community composition of three smaller braided rivers (Ashley | Rakahuri River, North Canterbury; Cass River, Mackenzie Basin, and Aparima River, Southland). The inclusion of the Ashley River, where mechanical weed-control experiments had been established several years prior, provided an opportunity to test for difference in the invertebrate communities associated with areas of weedy and non-weedy habitat and better understand impacts of weed management.

Following a large flood on the Ashley River in 2021, funding was extended (ECan ref: Project 1042) to obtain a further two summers of data on the Ashley and Cass, so that the impact of the flood and recovery of the invertebrate community could be assessed. Sampling from the Aparima was paused after two years, with plans to return after gravel extraction (permitted in the area where half the sites were located) was enacted, however this has not yet occurred.

This report summarises work completed over the full duration of the project (November 2019-February 2023) and presents key findings including recommendations for effective invertebrate biodiversity monitoring on braided rivers.

2.1 Objectives:

1. Test the recommended sampling approach developed on the Tasman River and set this up as a best practise technique for identifying terrestrial braided river invertebrate biodiversity values.
2. Gather baseline information on invertebrate biodiversity of three small, braided rivers in different geographic locations to learn about the degree of similarity among them.
3. Identify variation in invertebrate diversity between weedy and non-weedy sites within the Ashley River.
4. Record the impact on the invertebrate community of severe flooding that occurred in the Ashley River in 2021.

3 Methods

In October 2019, six sampling sites were established on the Aparima and Ashley rivers, and three on the Cass (Appendix 1, Fig. 1). Sites on each river were 100-300 m apart. Each of the 15 sites consisted of a line of five pitfall traps (each with 3 x 500 mm long metal guide arms to increase capture area, and a plastic rain cover), and one Marris style malaise trap, all spaced 6 m apart (Fig. 2, Appendix 2).

On the Aparima, all six lines were established in unmanaged 'weedy' areas, but three were in an area zoned for future gravel extraction to provide baseline data for a before-after gravel extraction comparison. The Cass River is relatively free from weeds, so only three 'non-weedy' unmanaged sites were established to provide baseline data from a small (compared to the previously sampled Tasman River) braided river in a relatively pristine environment. On the Ashley River, three lines were situated in a stretch of river where no weed control was undertaken ('weedy' sites located near Groyne 2 off Merton Road), and three in a stretch that had been mechanically cleared of weeds (non-weedy 'ripped' sites). Mechanical weed management trials were established on the Ashley River in 2016 following a period without any major floods between May 2014 and Winter 2017 (Ledgard & Davey, 2020). The associated weed growth reduced the area of bare gravel and was strongly correlated with a drop in bird numbers. In July 2019 (the winter prior to invertebrate monitoring), the ripping method used was switched from tines, which were quickly clogged with weed debris, to a purpose-built undercutting bar (Fig. 3).

To avoid flooding, many monitoring lines had to be moved slightly (± 20m) year-on-year because of changes in the position of river channels. On the Ashley, a more significant move was required in February 2021 as a gravel extraction consent was enacted over the Tull's area where the three original ripped monitoring lines had been placed. These lines were moved to the Marchmont ripped area for the remainder of the study (Fig. 1d).

Between early November and late February annually, pitfall and malaise traps were filled with 2-3 cm of 100% propylene glycol preservative and opened for five nights at approximately 6-week internals, giving a total of 16 sampling sessions (Table 1). At the end of each sampling week, the contents of each trap were poured into separate specimen jars and later rinsed and preserved in 70% ethanol. All adult specimens ≥ 2 mm in length were individual identified as morpho-species and named using a Recognisable Taxonomic Unit (RTU) convention where the most specific name possible is given; for example, if a native bee were collected and identified to species level it would be named *Leioproctus fulvescens,* but if the exact species could not be determined it would have be named *Leioproctus* sp. 1, and if the genus could not be determined it would have been named Colletidae sp. 1. Each unique morpho-species (hereafter 'species') was also given an RTU number starting at RTU001.

Specimens < 2mm in length, juvenile specimens, and damaged specimens were noted and retained (recorded as RTU000), but were not identified or counted. Dark brown or black chironomid flies 2-3mm in length with visible upper wing veins were collectively recorded as RTU049 as they were on the cusp of the size threshold for identification but extremely numerous, and unique species within the group would be difficult to separate with any confidence.

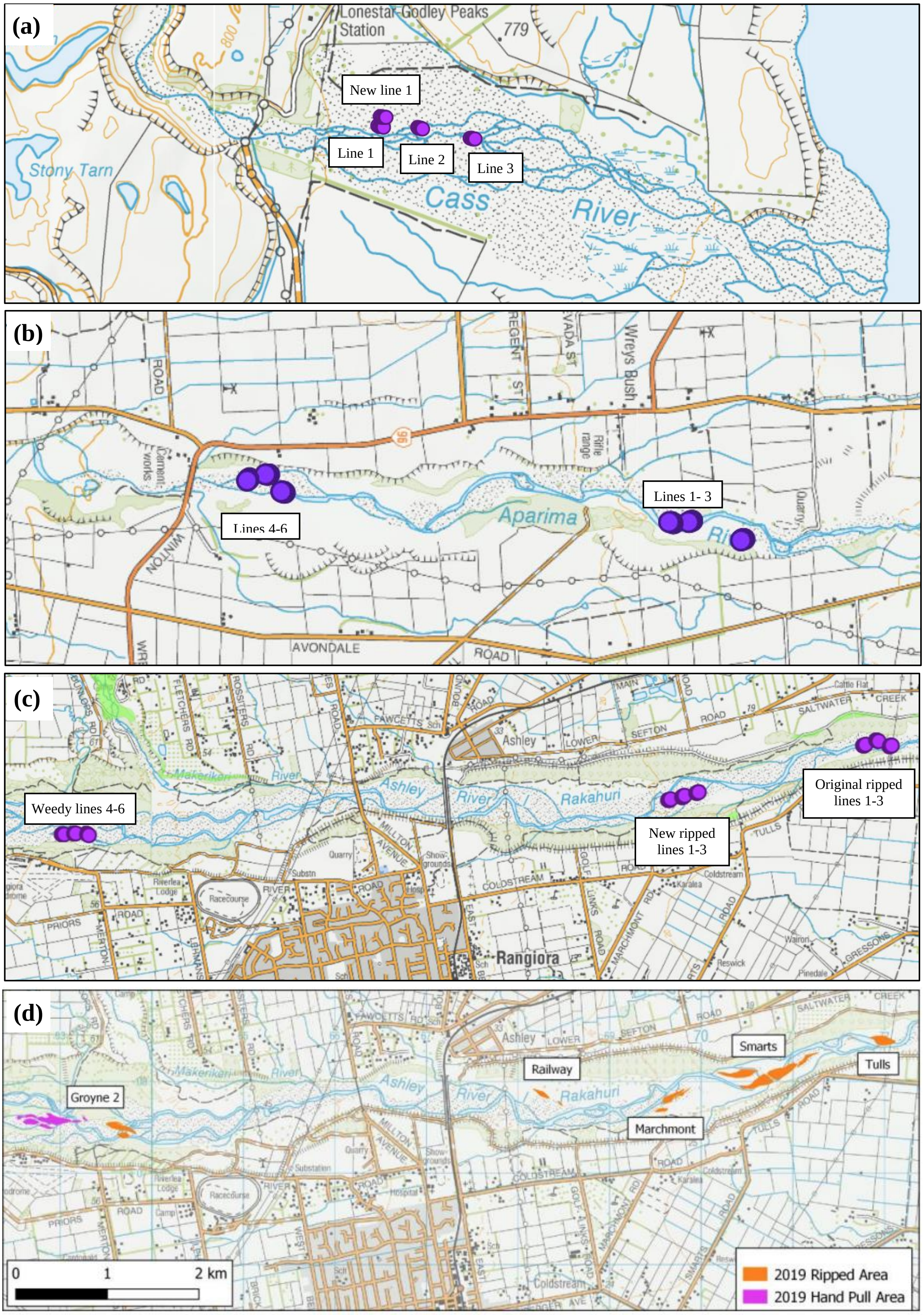

Figure 1: Locations of sampling lines on **(a)** Cass, **(b)** Aparima and **(c)** Ashley Rivers including new lines that were established during the study. **(d)** Areas of the Ashley where weed control ('ripping') was undertaken in July/August 2019 (re-used with permission from Ledgard & Davey 2020).

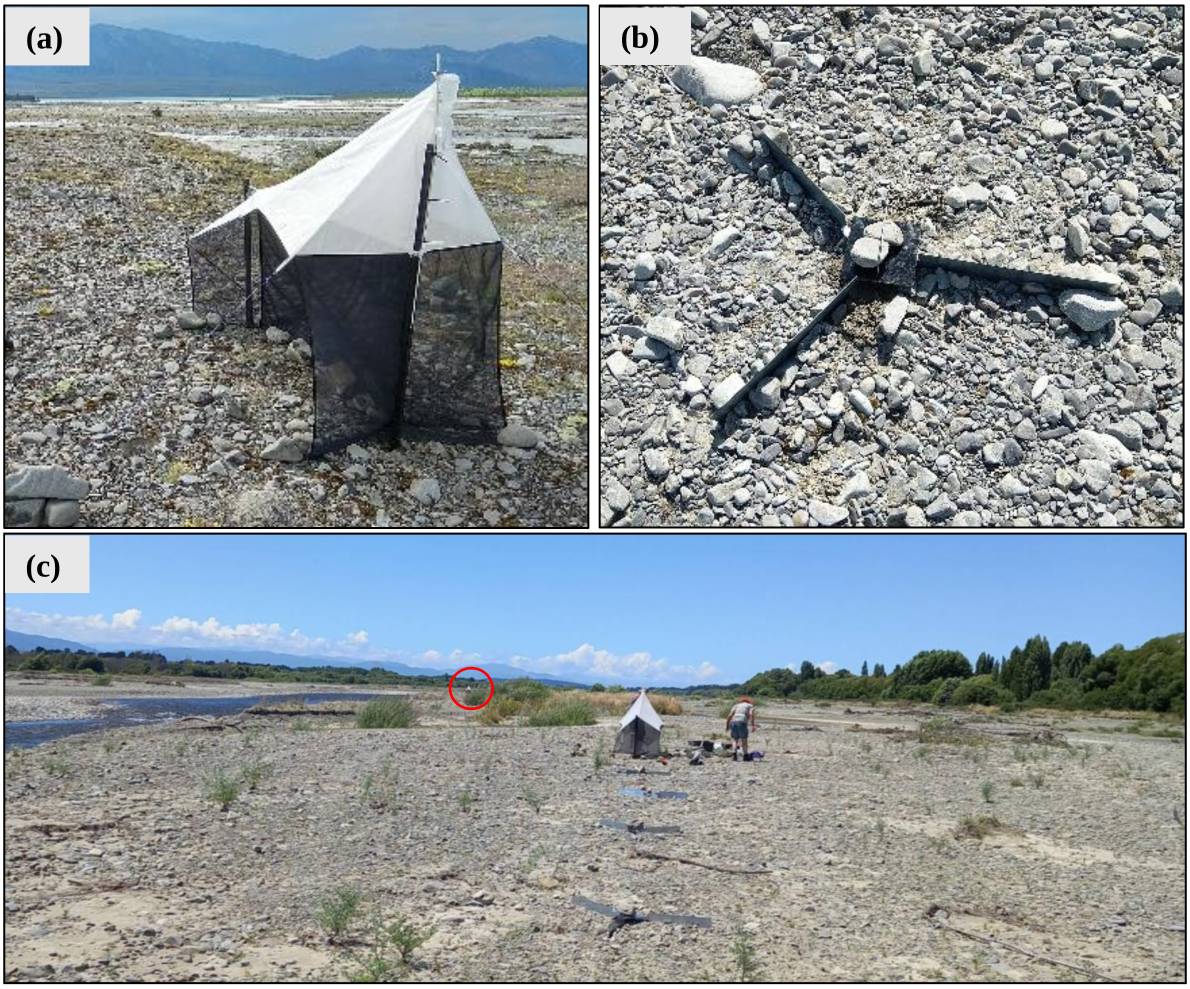

Figure 2: (a) Malaise trap and **(b)** pitfall trap with rain cover and guiding arms set on the Cass River. **(c)** Line of one malaise trap and 5 pitfall traps set at 6 m intervals on the Ashley River. Malaise trap for a second sampling line can be seen in the distance (red circle). ***Photos:*** *T. Murray*.

Figure 3: Tractor pulling purpose-built undercutting bar for mechanical weed control on the Ashley Rakahuri River. ***Photo****: Nick Ledgard*.

Table 1: Dates sampling was completed on the Ashley Rakahuri, Cass, and Aparima Rivers over summer months in each of four sampling years (November 2019 to February 2023). # lines = number of sampling lines planned per month. Total M = total combined number of malaise samples successfully recovered per year on each river. Total P = total combined number of pitfall samples successfully recovered per year on each river. Flooding resulted in the loss or non-deployment of 11 malaise and 46 pitfall traps over the entire study, mostly in the Ashley. Flooding in the Aslhey also caused February 2022 sampling to be delayed to early March in 2022.

Site	Year	Nov	Dec	Jan	Feb	# lines	Total M	Total P
Aparima	Year 1	21.11.19	22.12.19	12.01.20	26.02.20	6	22	105
Aparima	Year 2	8.11.20	8.12.20	14.01.21	5.02.21	6	24	120
Ashley	Year 1	15.11.19	6.12.19	10.01.20	21.02.20	6	24	120
Ashley	Year 2	23.11.20	13.12.20	23.01.21	26.02.21	6	24	120
Ashley	Year 3	1.11.21	5.12.21	9.01.22	6.03.22	6	23	120
Ashley	Year 4	No sample	4.12.22	15.01.23	20.02.23	6	17	90
Cass	Year 1	18.11.19	18.12.19	21.01.20	15.02.20	3	12	60
Cass	Year 2	6.11.20	5.12.20	21.01.21	18.02.21	3	11	60
Cass	Year 3	4.11.21	2.12.21	13.01.22	6.02.22	3	12	60
Cass	Year 4	4.11.22	12.12.22	15.01.23	13.02.23	3	12	60
						Total:	**181**	**915**

4 Analysis

RTU000 (small, damaged and juvenile specimens) and RTU049 (2-3 mm chironomid flies) were excluded from analysis as they include multiple taxa. Diversity metrics and descriptive statistics were calculated for each river to compare total and mean diversity and abundance per sampling line, trap and month.

Variation in species composition between sites, and between factors within sites (trap type, sampling line, year, month and weed treatment) were compared visually using non-parametric multi-dimensional scaling (nMDS) sample ordinations run in the computer package PRIMER 7 (V7.0.23). Analysis of Similarity (ANOSIM) was used to test for statistical differences between groups in ordinations and run with 999 permutations based on matrices of ranked pairwise similarities calculated between samples using the Bray-Curtis similarity measure (Bray & Curtis, 1957). Matrices were calculated from abundance data subjected to 4th root transformation to increase the weighting given to less abundant taxa, thereby ensuring ordinations were not overly biased by the abundances of a few very common species (Somerfield & Clarke, 1995). Ordinations were run with 100 iterations. In the resulting two-dimensional nMDS ordinations, points represent the species composition of a sample and those closer together are more similar in species composition (Clarke, 1993; Clarke & Warwick, 1998). Stress values indicate how well an ordination represents the underlying data. Values < 0.1 represent a strong ordination unlikely to be mis-interpreted, values up to 0.2 indicate the finer details of the plot may be subject to misinterpretation, and values > 0.2 must be interpreted with caution. ANOSIM *R*-values range from 0-1, tending towards 0 if the null hypothesis of no difference between groups is true. *R*-values with significance levels < 1.0% indicate statistically significant differences between the pairs of groups tested.

Using the Ashley and Cass data only, multi-season multi-species occupancy models (MacKenzie et al., 2008) were used to estimate the site occupancy of each species across the sampling lines over the duration of the study (full methods and results in report delivered by Proteus: Bertoia & MacKenzie, 2024). The model was implemented using the r package *spOccupancy* (Doser et al., 2022) and included trap line and sampling session (month within year) as covariates for the occupancy portion, but no covariates in the detection portion. This allowed occupancy probabilities for each species to vary by trap line and time, while detection probabilities remained constant over space and time for all species.

Shannon diversity, Simpson's diversity, and species richness were calculated from a community matrix generated by the occupancy model using the *vegan* package (Oksanen et al., 2004). This resulted in average estimated diversity and richness indices for each pitfall line and sampling session from each iteration (n = 3,150) of the model. This was repeated for the groups spiders-only, beetles-only, and then all 'other' invertebrates (i.e. all groups excluding spiders and beetles).

The detection probability estimated by the occupancy model for each species was applied to the equation $p^* = 1- (1 - p)^k$ to determine how varying pitfall trap replication (k) influenced the probability of detecting each species ($*p$). The same approach was used to explore how increasing or decreasing the number of pitfall lines influenced detection of each species.

5 Results

Monthly summer sampling (November to February) was only attempted in years 1 and 2 on the Aparima (Table 1). On the Cass and Ashley, monthly sampling was completed in all four years, except November 2022 when flooding prevented access to the Ashley. Over the whole study, flooding resulted in the loss of 11 malaise samples (1 lost in the Cass, 2 lost in the Aparima, 2 lost and 6 never set in the Ashley) and 45 pitfall samples (15 lost in the Aparima, 30 never set in the Ashley). Flooding (see Appendix 3 and 4) also delayed sampling on the Ashley on several occasions resulting in samples being collected later in the month than on the Cass, and on one occasion February sampling was delayed until early March (Table 1). Only 18 pitfall traps that were deployed on lines that were not flooded returned no invertebrates, primarily because of individual trap failures (e.g. cups either broken or filled with sand) rather than absence of invertebrate activity in the environment.

5.1 Overall abundance and diversity

More than 91,347 individual invertebrates > 2mm in length were sorted and identified as RTUs over the duration of the study. These comprised of 1,099 morpho-species (see Appendix 5 for species list), although the total number of unique species will be slightly different because some RTUs may represent different sexes of the same species (e.g. this is likely for some flies), while others will be morphologically cryptic species that have been lumped together (e.g. this may be the case for some of the small flies, small carabid beetles and spiders because they very difficult to consistently distinguish based on external morphology).

More than 50% of specimens were collected from the Ashley, but the site had close to twice the sampling effort compared to the other rivers (Fig. 4a). Similarly, the total number of species detected (Total RTUs Fig. 4b) was highest on the Ashley (700 species), followed by Aparima (662) and Cass (533). Correcting for sampling effort showed mean abundance per line was highest in malaise traps on the Ashley, and in pitfalls on the Aparima (Fig. 4c). Mean diversity (total species) per sampling line was highest on the Aparima ($\bar{x}$ = 80), followed by the Cass ($\bar{x}$ = 64) and Ashley ($\bar{x}$ = 60), a pattern consistent for both trap types (Fig. 4d). Mean species per trap was also highest on the Aparima, and 26% and 37% lower for malaise and pitfalls respectively on the Cass, and 33% and 46% lower respectively on the Ashley (Fig 4e, f).

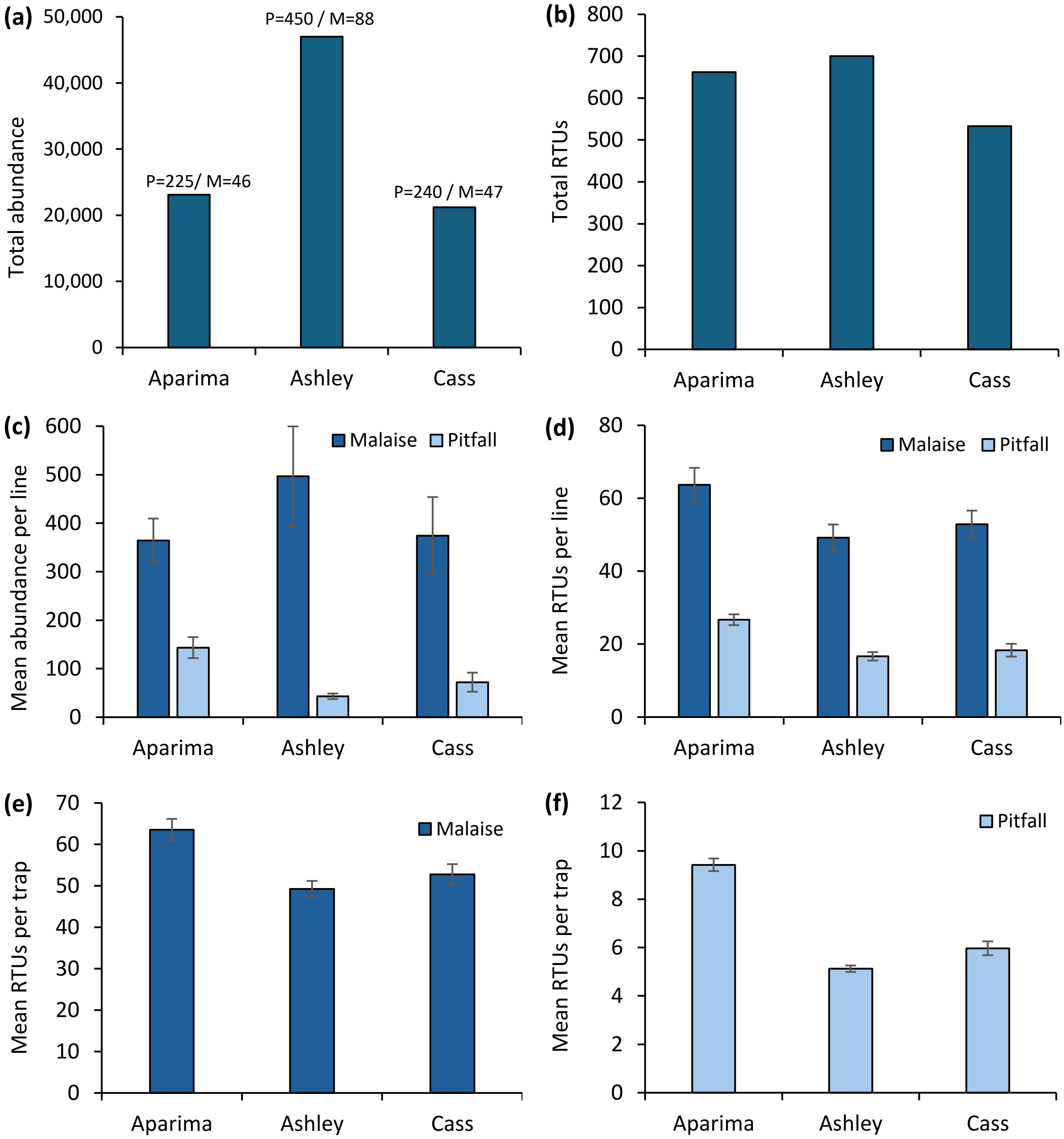

Figure 4: Total abundance **(a)** and diversity (number of different RTUs) **(b)** of invertebrates identified from each river inclusive of both trap types and all sampling sessions (note the variation in sampling effort: P = number of pitfalls, M = number of malaise traps). Mean (± SE) abundance **(c)** and diversity **(d, e, f)** of invertebrates per line **(c, d)** and per malaise or pitfall trap **(e, f)** sampled across all sessions. *Sampling sessions*: Aparima = 8 months sampled over 2 summers, Ashley/Cass = 16 months sampled over 4 summers. *Total number of samples recovered*: Aparima = 46 malaise, 225 pitfall, Ashley = 88 malaise, 450 pitfall, Cass = 47 malaise, 240 pitfall).

In agreement with findings for diversity per-unit-sampling-effort generated from raw species counts, occupancy model estimated Shannon diversity, Simpson diversity and species richness were also all slightly higher for the Cass than the Ashley (no Aparima estimates calculated) (Fig. 5). The higher Shannon's index suggests a wider variety of species on the Cass with a more balanced distribution of individuals, while the higher Simpson's index suggests less dominance of a few very common species. For all three indices, the difference between Cass and Ashley was detected when 'all species' and 'all species excluding spiders and beetles' were modelled together, but not clearly or consistently observed for spiders or beetles when these two groups were modelled independently.

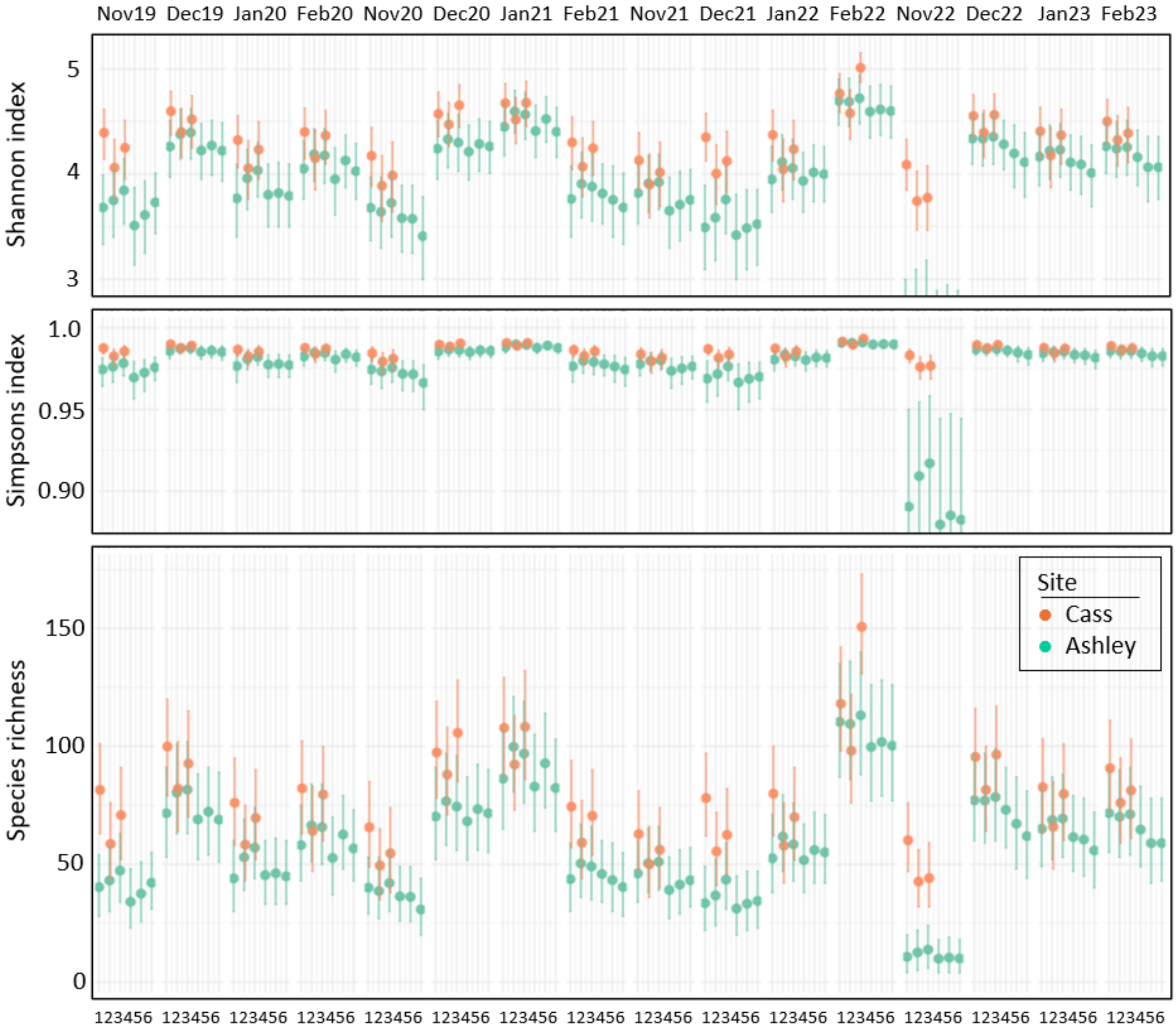

Figure 5: Average estimated **(a)** Shannon diversity **(b)** Simpsons diversity and **(c)** Species Richness calculated for each sample line (1-6) and each sampling month using occupancy modelling with all species included. Note the exceptionally low modelled estimates for the Ashley in November 2022 result from the absence of sampling data in that month because of flooding.

For the first three sampling years on both the Ashley and Cass, observed diversity tended to increase over the summer, usually peaking in February (Fig. 6a). However, in year 4 (2022-23) this trend did not hold, with diversity peaking in December. These trends were reflected in the malaise-only data as most diversity was detected using these traps. For pitfalls, however, observed diversity varied more randomly from month to month (Fig. 6b).

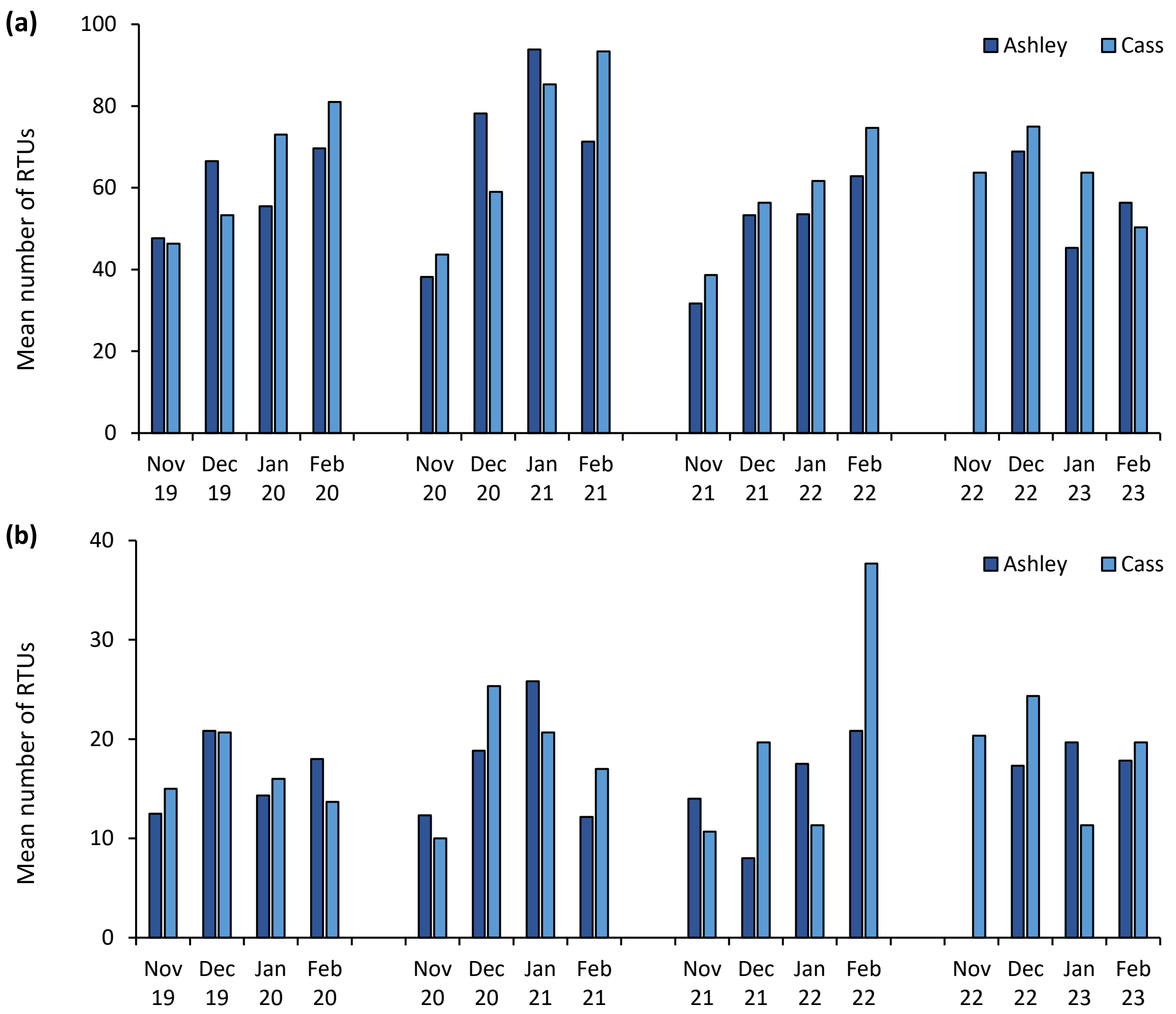

Figure 6: Mean species diversity (number of unique RTUs) per sampling line detected per month from **(a)** all pitfall and malaise traps combined and **(b)** pitfall traps only on the Cass and Ashley over 4 summers of sampling. Date from malaise traps alone repeats the patterns observed in (a).

Twenty-seven invertebrate orders were detected across all rivers (Fig. 7). Taxa from 13 insect orders were detected on the Cass and Aparima, and one more (Cockroaches) was detected on the Ashley. Of the non-insect groups, spiders and harvestmen (an order of arachnids that includes both predatory and scavenging species) were found on all rivers, but amphipods (landhoppers), collembola (springtails) and sausage millipedes (chordeumatida) were not detected on the Cass or Ashley, and isopods (e.g. slaters), segmented worms, slugs, snails and flatworms were also absent from the Cass. Diptera (flies), were both the most diverse and abundant group detected on all rivers, followed by smaller proportions of hymenoptera (primarily wasps, including parasitic species, but also some bees and ants), coleoptera (beetles), lepidoptera (moths) and arachnids (spiders and harvestmen) (Fig. 7). Relative proportions of taxonomic diversity were similar for the two trap types although slightly more hymenoptera and moth species (both predominantly flying groups) were collected in malaise traps and more beetles and spiders (predominantly crawling groups) in pitfall traps (Fig. 7).

Flies, caddisflies and the group ‘other insects’ (Orders orthoptera, blattodea, dermaptera, ephemeroptera, neuroptera, plecopteran, psocoptera, thysanoptera) were relatively more abundant than diverse, while hymenoptera, coleoptera and arachnids were less abundant than diverse, particularly on the Ashley River.

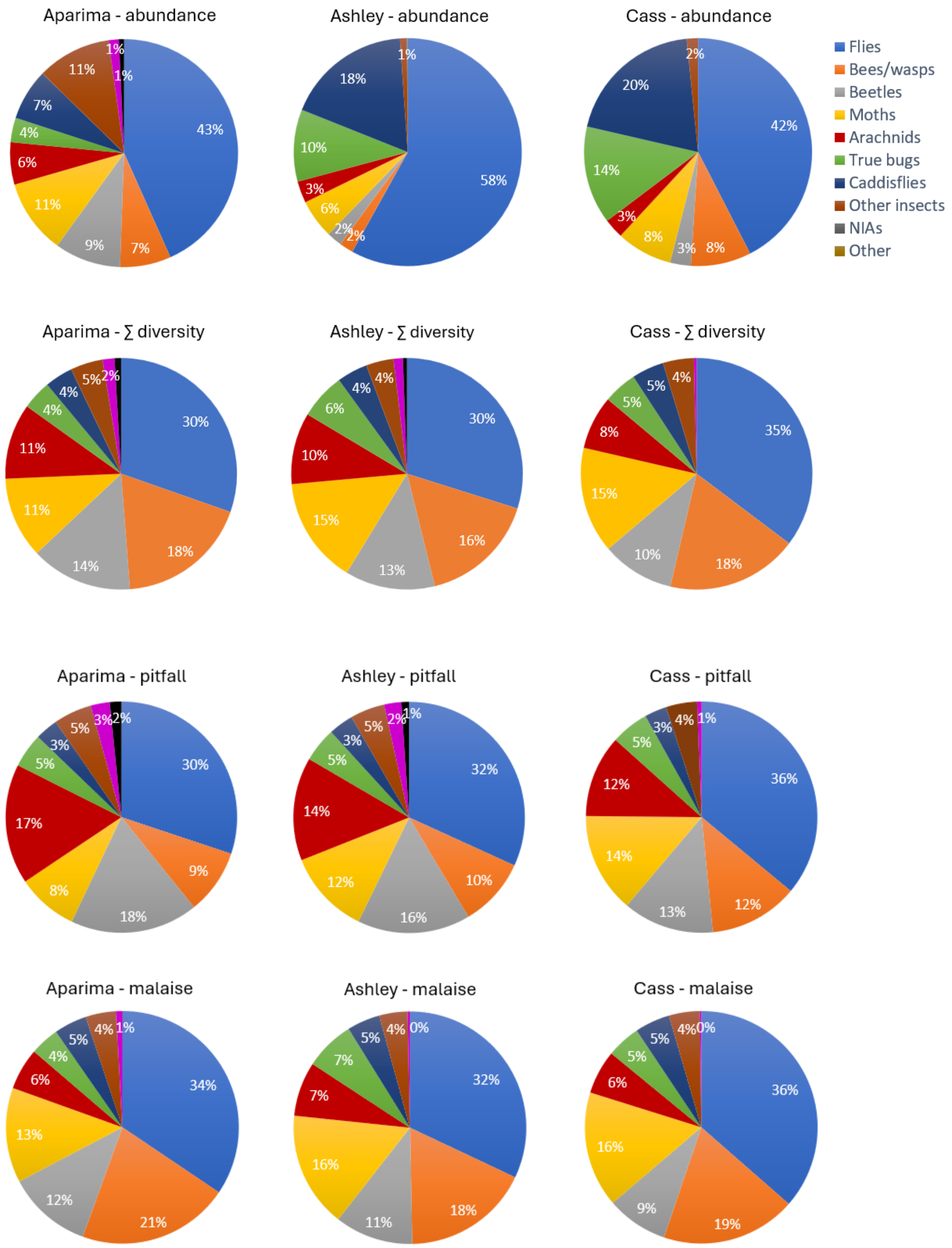

Figure 7: Proportion of total abundance, total (∑) diversity, and total diversity by trap type, represented by different taxonomic groups at each River inclusive of all sampling sessions. Other insects = insects in the orders orthoptera, blattodea, dermaptera, ephemeroptera, neuroptera, plecopteran, psocoptera, thysanoptera. NIAs = Non-insect arthropods in the groups amphipoda, isopoda, collembola, diplopoda, chilopoda. Other = platyhelmitds, annelids, molluscs.

5.2 Functional diversity

Of the 1,099 species identified, 81% could be confidently assigned as detritivores, herbivores, fungivores, omnivores, predators or parasitoids. Simplifying this to detritivores (feeding on dead or decaying organic matter), herbivores (feeding on live plant, fungi or algae) and predators (free-living predators as well as parasitoids) gave a feeding guild ratio of approximately 1:4:5 consistently across all rivers. For all three rivers, approximately half of the assigned species were predators (Fig. 8a). On the Ashley and Cass ~40% were herbivores and 10% detritivores, while on the Aparima there were slightly fewer herbivores and more species that feed on dead organic matter. As expected, slightly more herbivores (especially moths and sucking bugs) were collected using malaise traps while more detritivores were collected in pitfalls (Fig. 8b).

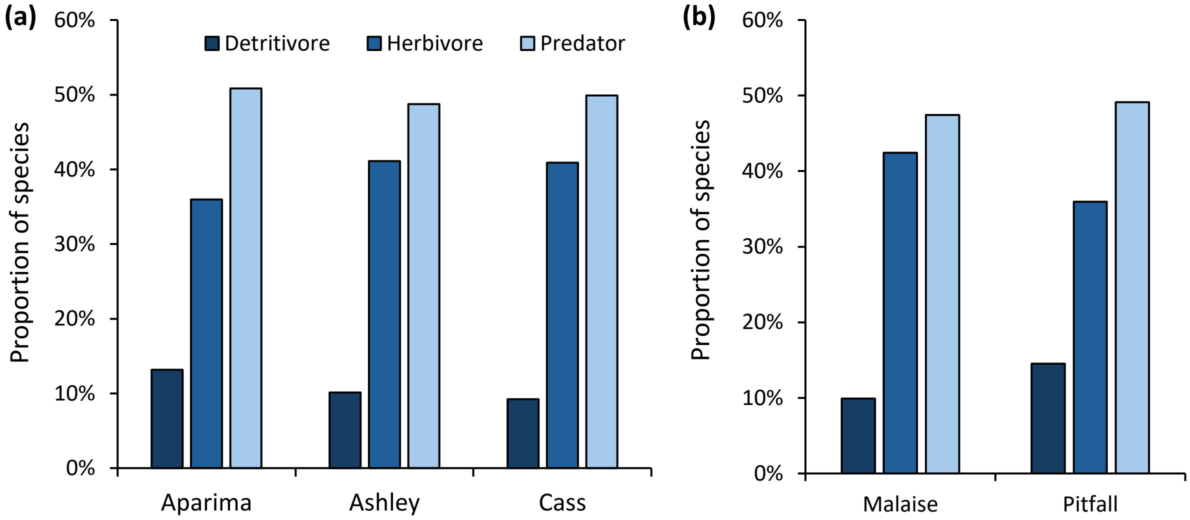

Figure 8: Proportions of the 888 taxa that could be assigned a larval feeding guild and were collected from each of **(a)** the three rivers and **(b)** the two trap types and classed as detritivores (feeding on dead organic matter of any kind), herbivores (live plant, fungi or algae) and predators (free-living and parasitoids).

5.3 Community composition

There was a small but significant difference in the community composition of each of the three rivers over the full sampling period (nMDS *R* statistic = 0.238, significance level 0.1%, Table 2; Fig. 9). Half of all species were found at only one of the rivers (Aparima = 218, Ashley = 197, Cass = 129). As a proportion of total diversity observed at each river, the number of these location-specific species was similar (Aparima = 33%, Ashley = 28%, Cass = 24%). Most were beetles, flies, wasps, moths and spiders. Proportionally, there were fewer unique spiders and beetles on the Cass, but more flies and caddisflies, while on the Ashley there were more unique true bugs (hemiptera) and fewer wasps. The Aparima had more unique non-insect invertebrates including earthworms, flatworms, slugs, snails and millipedes, all of which were rarely encountered on the other rivers, especially the Cass (1 millipede species detected).

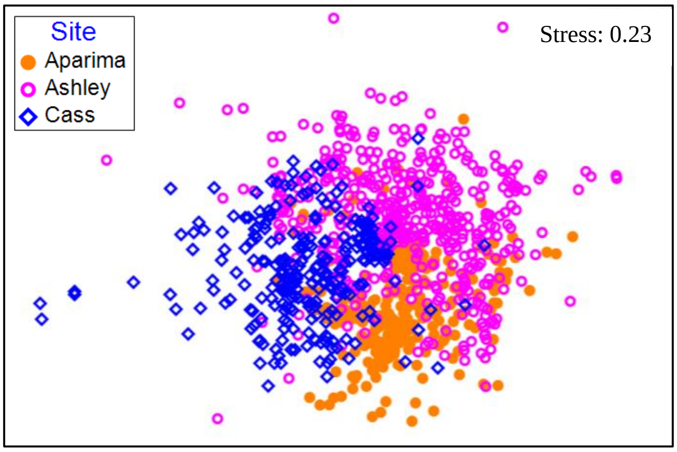

Figure 9: MDS sample ordination showing community composition of the Cass, Ashley and Aparima Rivers where each point represents the species composition of a single pitfall or malaise sample. The high stress level (> 0.2) indicates only the course spatial patterns (e.g. between sites rather than individual samples within sites) are reliable.

There was a significant but moderate difference between invertebrate communities captured by the two trap types at all three rivers (Fig.10a-c; Table 2). This difference was weakest for the Ashley (R = 0.144), likely because of the higher sampling effort and subsequent higher diversity driving greater variation in samples within the site. Overall, 51% of species were detected only in malaise traps, 15% only in pitfalls and 34% in both. The proportion of species that were not found in both trap types was 57% in the Aparima, 56% in the Ashley and 47% in the Cass.

Comparing the species composition of each sampling line (Fig. 10d-i) showed no clear spatial influence on species composition within the Ashly or Cass, but on the Aparima there was evidence for a difference in the invertebrate communities observed in the upper three lines (1-3) compared to the lower three lines (4-6). This difference was more evident for the invertebrate community sampled by pitfall traps than by malaise traps (Fig. 10d, g). Ordinations indicated a weak temporal influence on species composition (Fig. 11, 12). This effect was strongest for the pitfall and malaise communities observed on the Ashley in Years 3 and 4 compared to Years 1 and 2 (Fig. 11c, 12c) followed by the Cass malaise community (Fig. 12e). For the malaise community, there was a clear effect of month on species composition in the Aparima and Cass, with a distinction between samples from November and December compared to January and February (Fig. 12b, f).

Table 2: One-way analysis of similarity results testing for differences in sample species composition between rivers, trap types, and between ripped and un-ripped weed treatments on the Ashley.

Factor	Pair	R statistic	% significance
Site	Aparima - Ashley	0.212	0.1**
	Aparima - Cass	0.298	0.1**
	Ashley - Cass	0.240	0.1**
Trap – Aparima	Pitfall - Malaise	0.516	0.1**
Trap – Ashley	Pitfall - Malaise	0.144	0.1**
Trap – Cass	Pitfall - Malaise	0.418	0.1**
Weed – Ashley	Ripped - Not-ripped	0.032	0.1**

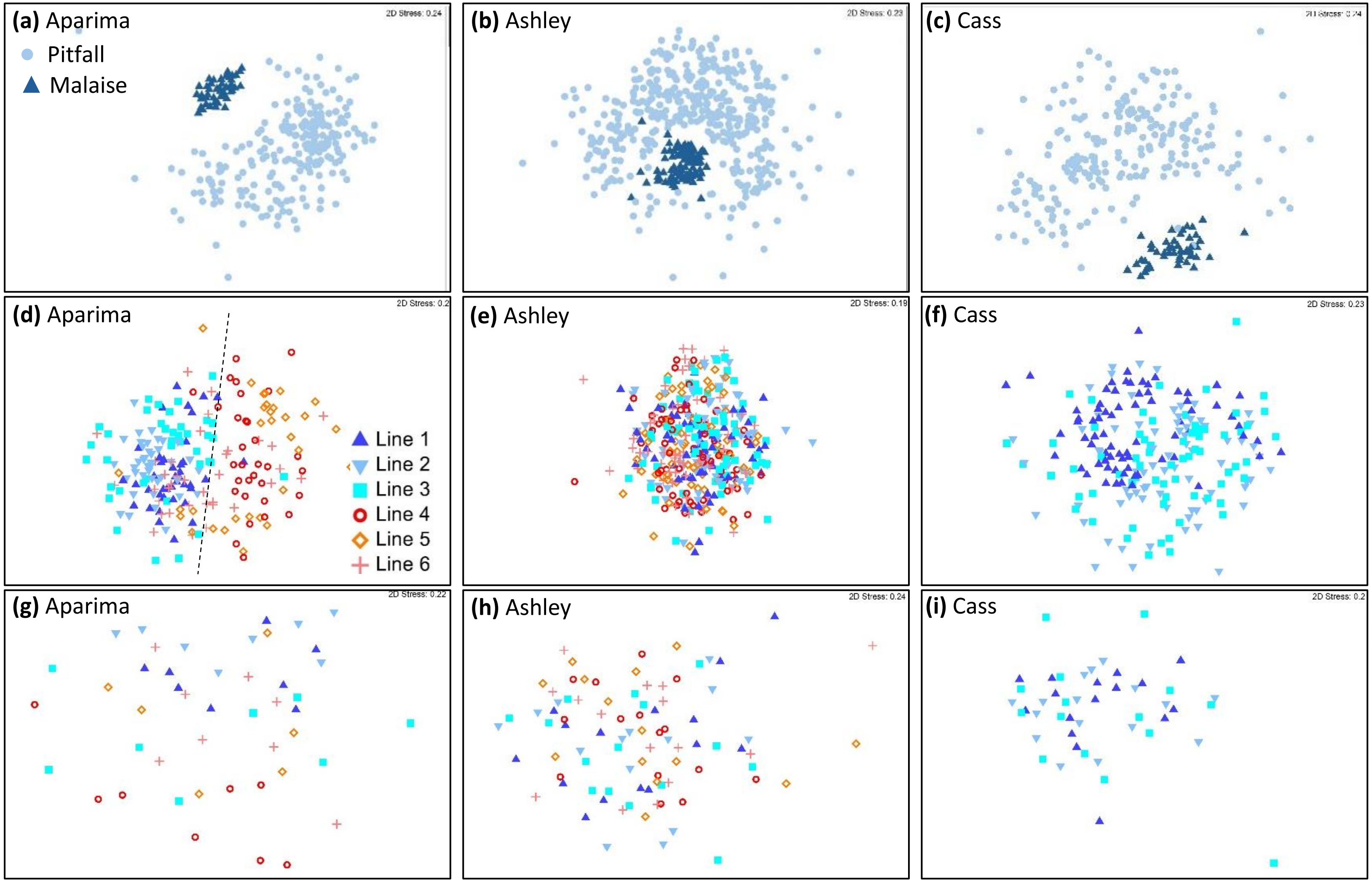

Figure 10: Non-parametric MDS sample ordinations for each river showing relative species composition of **(a, b, c)** the two trap types (▲= malaise, ● pitfall), **(d, e, f)** each line of pitfalls and **(g, h, i)** each line of malaise traps. Black dashed line in (d) shows the separation of samples from lines 1-3 from those of lines 4 and 5 (and some of 6) which were located further down-river on the Aparima. Legend in (d) applies to (d-i) Stress range = 0.22-0.24.

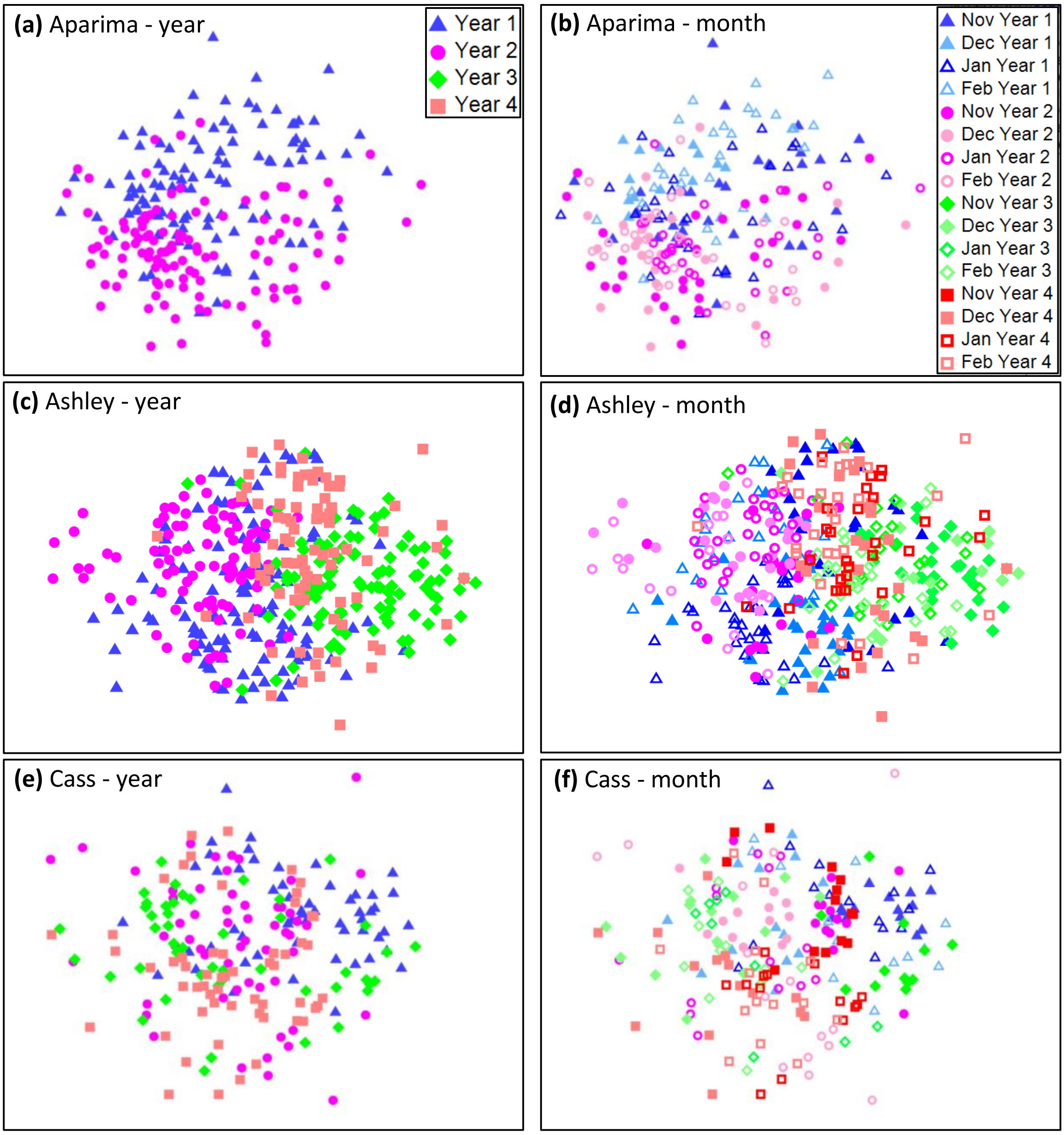

Figure 11: Non-parametric MDS sample ordination for each of the three rivers showing the relative species composition of pitfall samples collected in each year (left, see legend in (a)) and each month-within-year (right, see legend in (b)); **(a, b)** Aparima (sampled in years 1 and 2 only), **(c, d)** Ashley, **(e, f)** Cass. Stress range 0.22-0.24.

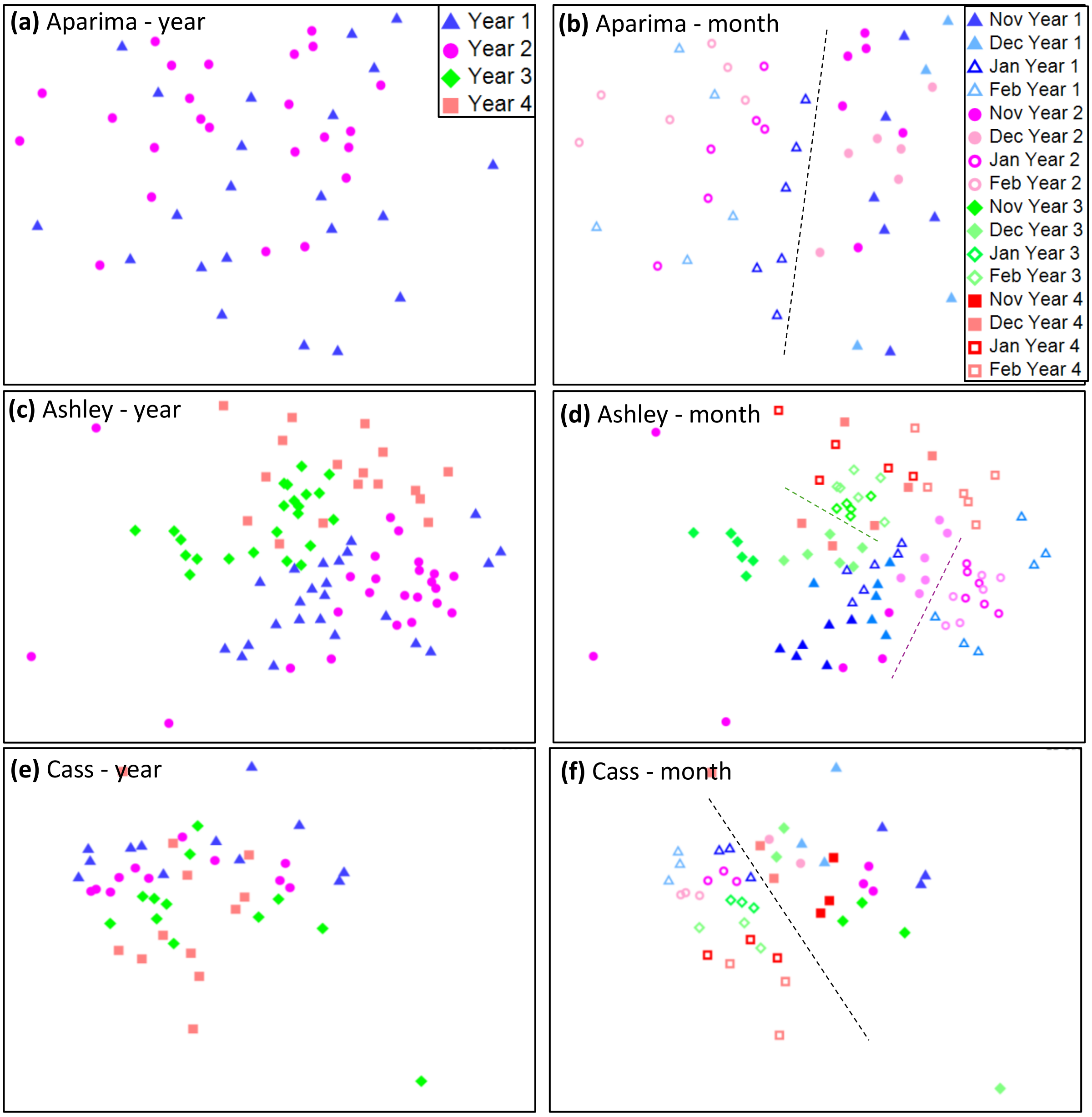

Figure 12: Non-parametric MDS sample ordinations showing the relative species composition of malaise samples collected in each year (see legend in (a)) and in each month-within-year (see legend in (b)) on the Aparima **(a, b)**, Ashley **(c, d)**, and Cass **(e, f)**. Dashed lines in b, d and f show the separation of samples from months before vs. after Christmas at each site.

5.1 Impacts of weed control

On the Ashley, there was no evidence of any significant impact of mechanical weed control (ripping) on mean species abundance or diversity per sampling line. Abundance was slightly, but not significantly, higher in the un-managed sampling area in year 1 and 2, but this was reversed in years 3 and 4 (Fig. 13a). The average number of species detected per line of pitfall traps and per malaise trap tended to be slightly higher in the ripped sites (Fig. 13b, c) but again the difference was not significant. Similarly, no significant difference in community composition was observed between samples from ripped vs. un-managed sections of the river for either trap type (Fig. 14; Table 2).

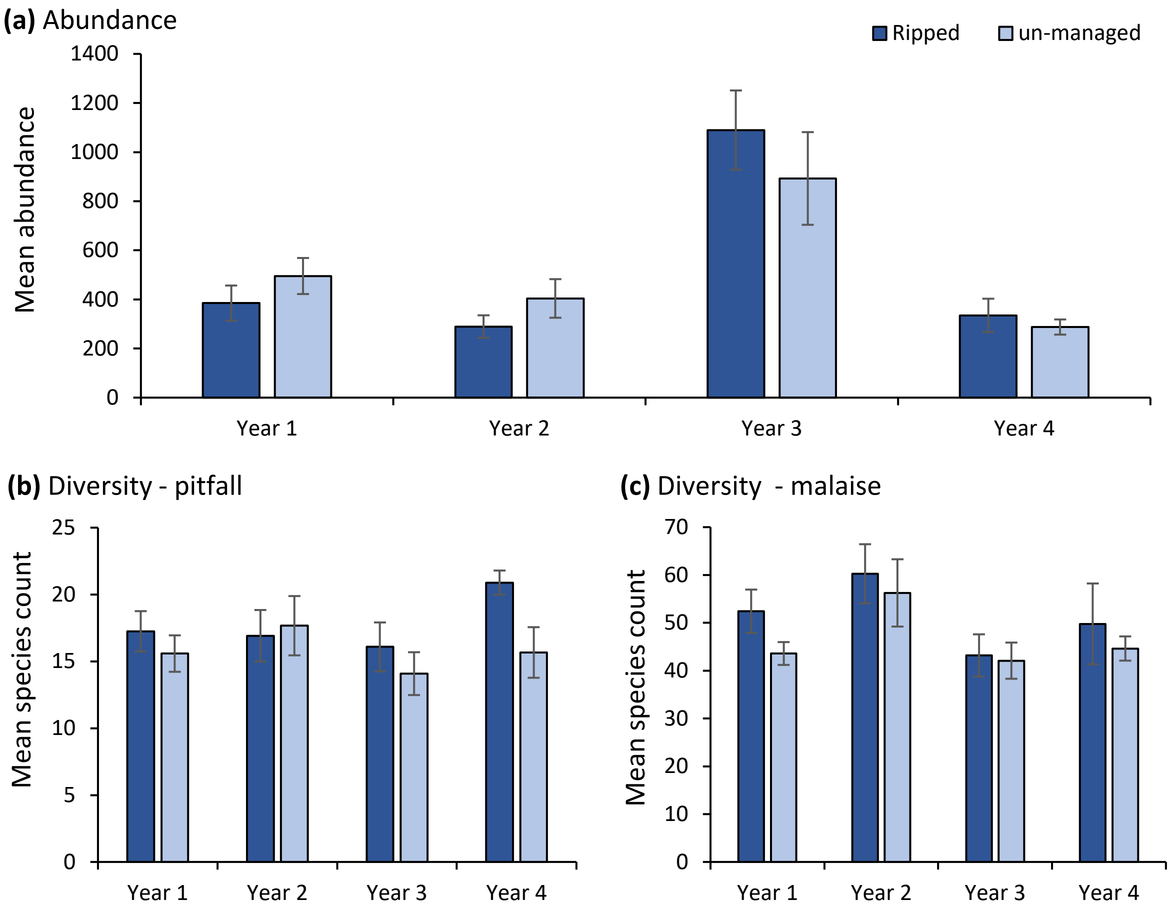

Figure 13: Mean (± SE) **(a)** abundance (all traps) and mean (± SE) number of species detected per sampling session in **(b)** pitfalls and **(c)** malaise traps set in ripped vs. unmanaged sections of the Ashley River over 4 sampling years.

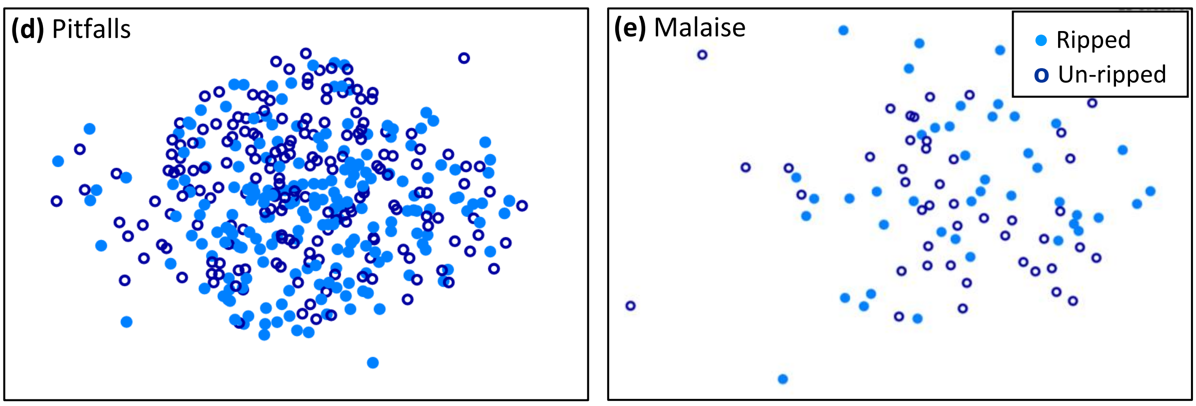

Figure 14: Non-parametric MDS sample ordinations showing the relative species composition of **(a)** pitfall samples and **(b)** malaise samples collected from sections of the Ashley River where mechanical weed-ripping had occurred compared to un-ripped sections where no weed control was undertaken. Data is inclusive of all 15 sampling sessions conducted on the Ashley over four years.

5.2 Impacts of flooding

No consistent correlations were observed between river flow and either the abundance or diversity of invertebrates captured on the Ashley River (Appendix 3). In the first two years abundance was generally below 400 specimens per sampling line per session, and two higher peaks did appear to occur after periods of low flow. In year 3, very high abundances (> 1400 specimens per sample line) were observed in January 2022 and late March 2022, both approximately 2 weeks after large flood events. However, low abundances (< 375 insects per sample line) were subsequently recorded throughout year 4 (2022-23 season) which was marked by a series of more moderate floods in mid and late November, mid-December and mid-February.

The average number of species detected per pitfall line and per malaise trap was relatively consistent in each of the sampling sessions between January 2022 and February 2023, despite multiple flooding events (Appendix 3 b, c). The number of species caught in malaise traps in December 2021 and January 2022 were very similar despite the December period being preceded by low flow (6-82 m^3/s), and the January period occurring just 2 weeks after extremely high flows (462 m^3/s). In contrast, the number of species detected in pitfall traps, was twice as high in January (post-flood) than December (pre-flood).

5.3 Probability of detection and sampling effort

Occupancy modelling applied to pitfall data from the Ashley and Cass rivers indicated that the probability of detecting any given species in a sampling session was low, although that for spiders was consistently higher than for beetles. Varying the modelled number of traps per line, or the number of lines resulted in detection probabilities increasing continually as traps or lines were added but the size of these gains diminished with each addition (Fig. 15,16).

At the current effort of 5 pitfalls per line, the median probability that a species would be detected at least once in the season was < 0.2 for beetles and almost 0.3 for spiders (Fig. 15). However, the spread around this median indicates that very common species have much higher detection probabilities, extending up to almost 0.7 for common beetles and almost 0.9 for common spiders and for all ‘other’ invertebrate groups combined (Fig. 15). Reducing the number of pitfalls from 5 to 3 per line predicted a reduction in detection probabilities of between 45% and 50% for the three species groups (see change in upper extent of 95% lines in Fig. 15), while increasing to 7 or 9 pitfalls predicted smaller gains (7 pitfalls = 24.3-28.8% increase in detection probability; 9 pitfalls = 15.9-19.6% increase in detection probability).

The probability of detecting a species at least once in a session increased most when increasing from 1 sampling line to 2 lines but gains quickly diminished to less than 10% after 5 lines (Fig. 16; Table 3). To achieve a median probability of detecting a species that is present to more than 50% for all groups (spiders, beetles and others) required at least 5 sampling lines on the Cass and 7 lines on the Ashley, although for spiders alone just 4 lines were required on the Aslhey (Fig. 16).

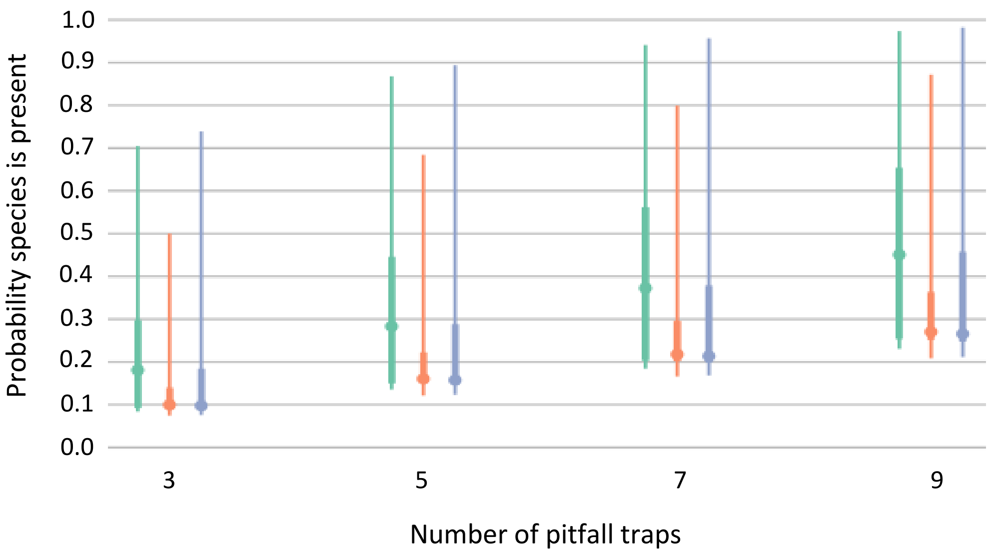

Figure 15: Modelled probability of detecting a species at least once in a sampling session when varying the number of pitfall traps per line from 3 to 9. Probabilities modelled independently for (from left to right); spiders (green), beetles (orange), and all 'other' groups combined (blue). Circles indicate median probabilities, thick lines show the middle 50% of data and thin lines extend to 95% of the data.

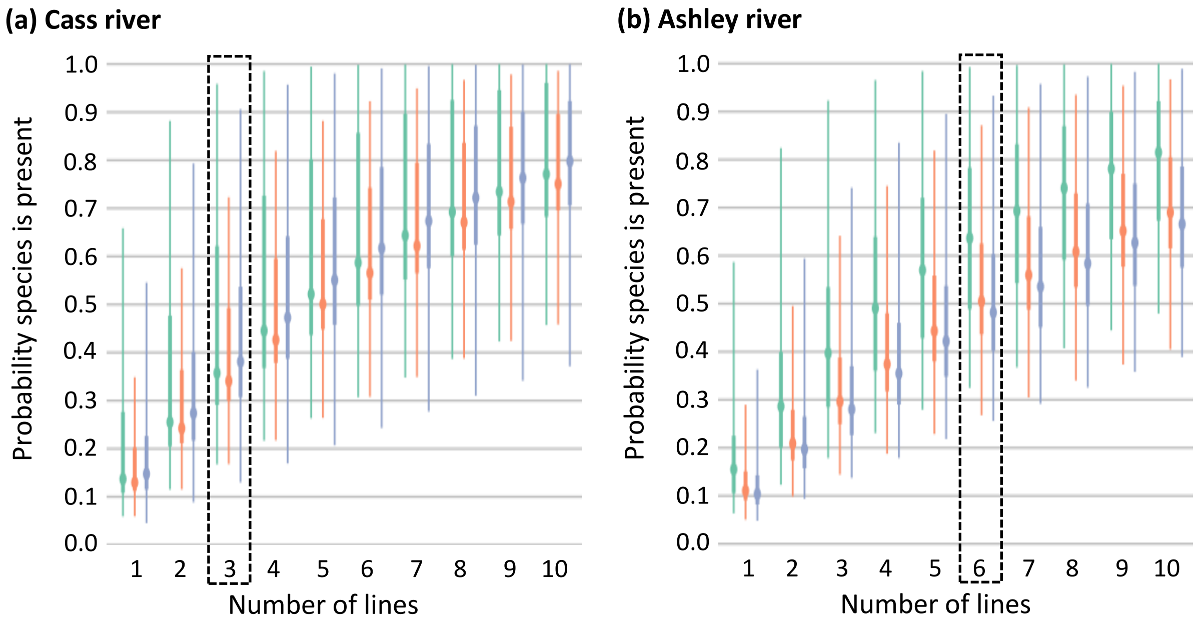

Figure 16: Model-estimated probability of detecting a species at least once in a sample session as the number of sampling lines (each containing 5 pitfall traps) is varied from 1 to 10 at **(a)** Cass and **(b)** Ashley rivers. Probabilities are modelled independently for (from left to right); spiders (green), beetles (orange), and all 'other' groups combined (blue). Circles indicate median probabilities, thick lines represent the middle 50% of data and thin lines extend to 95% of the data. Dashed-line boxes indicate the actual sampling effort for this study.

Table 3: Modelled estimate of the percent change in the probability that a species is present at least once in a line of pitfall traps as the number of lines is increased from 1 to 10. Data are presented for spiders alone, beetles alone, and all 'other' remaining invertebrate groups combined.

	Number of pitfall lines									
	1	2	3	4	5	6	7	8	9	10
Ashley										
Spiders	0	70.6	31.3	18.8	12.8	9.4	7.2	5.7	4.6	3.8
Beetles	0	84.8	39.1	24.1	16.8	12.4	9.6	7.7	6.2	5.1
Other	0	80.8	37.3	23.1	16.2	12.1	9.5	7.6	6.2	5.2
Cass										
Spiders	0	67.3	29.5	17.6	12.0	8.8	6.8	5.4	4.4	3.6
Beetles	0	79.9	36.0	21.7	14.7	10.7	8.1	6.4	5.1	4.2
Other	0	73.5	33.0	19.9	13.5	9.8	7.4	5.8	4.6	3.8

Modelling the interaction between the number of pitfalls and number of sampling lines indicated that increasing the number of traps from 5 (current standard) to 7 could increase the probability of detection by ~10% for any number of sampling lines (Fig. 17). At current the effort of 5 pitfalls per line and 3 lines on the Cass River (Fig. 17a) the median probability that a beetle species will be present and detected is below 0.1, while that for spiders is just above 0.1. At the Ashley, these figures were slightly higher at just below 0.2 for beetles and 0.3 for spiders. Again, variation above the median was large such that detection probabilities ranged from just under 0.4 for some very common beetles, to > 0.7 for common spiders.

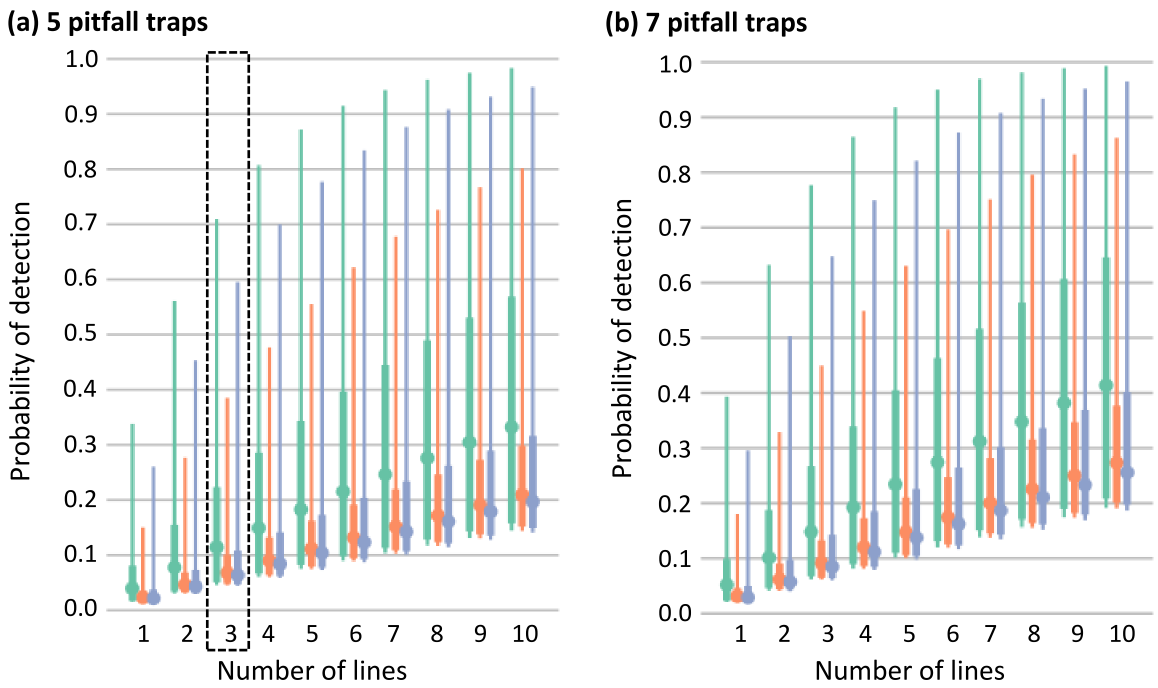

Figure 17: Model-estimated probabilities that a species is present and detected at least once as the number of sampling lines on the Cass River is varied from 1 to 10 based on **(a)** the current sampling effort of 5 pitfalls per line compared to **(b)** an increase to 7 pitfalls per line. Probabilities are modelled independently for (displayed from left to right); spiders (green), beetles (orange), and all other groups combined (blue). Circles show median probabilities, thick lines represent the middle 50% of data and thin lines extend to 95% of the data. Dashed-line box indicates the actual sampling effort for this study.

6 Discussion

This study aimed to test a standardised monitoring approach to measuring braided river invertebrate biodiversity values while gathering baseline data on the invertebrate communities of three very different rivers and exploring whether spatial and temporal variation within the Ashley River community could be attributed to weed management or flooding.

6.1 Objective 1: Test of sampling approach

Test the recommended sampling approach developed on the Tasman River and set this up as a best practise technique for identifying terrestrial braided river invertebrate biodiversity values.

Methods developed for the Tasman River Study were refined and successfully applied to the Aparima, Ashley and Cass Rivers based on several efficiencies recommended by Murray and Anderson (2019). Firstly, sampling was limited to malaise and pitfall traps only. Malaise sampling was selected because the method captured the greatest diversity in the Tasman Study. However, we did not use malaise traps with the recommended collection troughs (as opposed to jars) because such traps are not readily available and are therefore impractical to consider for standardised monitoring. Pitfall trapping was selected as a method that targets a different section of the invertebrate community, particularly beetles, spiders and other flightless (and therefore more locally constrained) species that are less likely to be detected with malaise traps and may be more responsive to local environmental conditions and management actions.

Sampling frequency was reduced from continuous weekly sampling over 14 weeks, to one 5-day period per month (selecting optimal weather at 4–6-week intervals) for the four warmest summer months. Juvenile specimens (which made up > ¼ of specimens caught in pitfalls in the Tasman Study but contributed only 2 RTUs that were not also detected as adults) and specimens < 2mm in length were not processed as this was predicted to reduce the number of specimens requiring identification by 46%, with little ecological meaningful impact on diversity metrics.

6.1.1 Trap type

Because invertebrates are so diverse, trapping method or combination of methods applied must be targeted to meet the objective of any invertebrate study. Here, the objective was to obtain baseline biodiversity and community composition metrics. As expected, the two trapping methods detected slightly different invertebrate communities. Overall, 15% of species were limited to pitfall traps, 51% to malaise and 34% were found in both trap types. Malaise traps caught significantly more invertebrates than pitfalls and more species, but the correlation between abundance and diversity detected was not linear. Although increasing trap number will increase the number of species detected, malaise traps have a tendency to catch very large numbers of common species, such as small flies and caddisflies. As such, it is important not to deploy more malaise traps than necessary or gains may be outweighed by the extra processing time. Here, malaise samples consumed 67% of processing time and detected 85% of species, while pitfall samples took 33% of processing time and detected 49% of species. This means the species detected per processing hour was only slightly lower for malaise (1.9) than pitfall (2.3) samples and the number of malaise samples was not excessive. However, assessment of species accumulation curves (data not presented) indicates the accumulation trajectory would be almost identical if only half the malaise traps had been used, so it may be possible to extrapolate total diversity metrics from a smaller sample size.

6.1.2 Replication

The impact of increasing sampling effort was also clear when comparing species counts from the Ashley River to those from the Aparima and Cass where sampling effort was lower. Occupancy modelling indicated that to achieve median detection probabilities of at least 50% for all species groups, at least 5 sampling lines were required on the Cass, and 7 on the Ashley. However, gains in the proportion of species detected diminish with each additional trap or line. For example, when varying pitfall trap replication, the greatest gains occurred by increasing from 3 to 5, while for sampling lines, the greatest gains were achieved by increasing from 1 line to 2. Is not recommended, therefore, that more traps or lines of traps should be added to measure invertebrate biodiversity in rivers of the size assessed to date.

There was limited evidence for differences in community composition between sampling lines in the Ashley (sites spread over 6 km) and Cass (sites spread over 500 m), but some evidence in the Aparima where the most distinct lower sites were separated from upper sites by 3 km. In the Tasman River, Murray and Anderson (2019) found moderate spatial effects when sampling using pan and pitfall traps (but not malaise) over a much larger area (sites up to 10km apart). This suggests it would be prudent to distribute sets of sampling lines at intervals along the length of particularly long rivers, especially when there is clear variation in habitat representation. Sets of 2 or 3 lines at intervals of 5 km or more apart or targeted to specific habitat types are likely to be sufficient to capture spatial variation.

6.1.3 Frequency and timing

For the first three years of study, total diversity tended to increase from November to February. This was driven by malaise trap catch and not consistent for pitfall traps. There was weak evidence for annual variation in community composition, but stronger evidence for monthly variation within years. For studies aimed at monitoring changes in community conditions over time, both findings reiterate the importance of consistently sampling in the same part of the season, especially when using malaise traps. If sampling less frequently is essential, the community differences detected in November and December compared to January and February indicate that reducing to one month pre- and one month post-Christmas would promote detection of a greater proportion of total diversity than sampling in two months either early or late in the summer. Sampling in only one month per year is not recommended.

Sampling over multiple days and nights dampens natural daily variability in invertebrate activity and reduces the potential for underestimating diversity because of short periods of unfavourable weather (e.g. a single cold or wet night). Sampling for 5 nights was found to be achievable in this study. Longer periods would require leaving traps on rivers over the weekend when visitation rates and potential trap interference are higher. We did not test shorter periods, but reducing to 4 or 3 nights targeting optimal weather conditions may be sufficient as long as done consistently. Fewer than 3 nights is not recommended.

6.1.4 Species identification

Identification to morpho-species level is the most time-consuming aspect of the sampling approach tested. Over the 4 years of study, the average time to process a single sample was 4h 32min per malaise and 20 min per pitfall. However, these averages reduced to 2h 43 mins and 15 min respectively for years 2-4 as the reference collection was established, new species were encountered less frequently, and staff skills and familiarity increased. Identifying specimens to order level only is not recommended as proportional diversity at the order level was very consistent between the three rivers sampled here, as well as the Tasman sampled previously. As such, little understanding of biodiversity values can be gained in the absence of more detailed species identification. Processing time could be further reduced by assigning morpho-species at the order level only (e.g. Coleoptera 1, 2….99), rather than the lowest possible taxonomic level as attempted here. This would result in the same number of 'species' identified but provide no resolution at the family, genus or species level, so no information on ecological characteristics like feeding guild could be determined.

6.2 Objective 2: Baseline community metrics

Gather baseline information on invertebrate biodiversity of three small, braided rivers in different geographic locations to learn about the degree of similarity among them.

Metrics of species diversity and community composition were successfully compiled for the Ashley, Aparima and Cass, three rivers that are quite different in their current condition. Coupled with data already gathered for the Tasman River, we now have a foundational understanding of the invertebrate biodiversity values of a large and a small upland river, both in good condition with few weeds, a reasonably free-flowing lowland river with moderate and variable weed density, and a constrained lowland river with substantially more exotic weeds.

The proportional composition of different invertebrate groups was markedly similar for all three rivers assessed here and matched what was previously observed on the Tasman. Flies made up approximately 1/3 of species in all cases, followed by hymenoptera, moths, beetles and spiders. Proportionally, flies were more abundant on the Ashley. Interestingly, as a function of search effort, diversity observed on the Cass and Ashley was very similar but that on the Aparima was higher. Half of all species detected in this study were observed on only one of the three rivers. On the Aparima, there were more non-insect arthropods (E.g. landhoppers, slaters, millipedes and centipedes), as well as flatworms, exotic worms, slugs and snails (all absent from the Cass), reflecting the higher prevalence of exotic grasses, weeds and associated silt, soil and litter build up. The proportion of species represented by flies, caddisflies and sucking bugs on the Aparima was similar to the other rivers, but their proportional abundance was lower, while the abundance of orthoptera, specifically field crickets, was much higher.

6.3 Objective 3: Impacts of weed control on the Ashley River

Identify variation in invertebrate diversity between weedy and non-weedy sites within the **Ashley River.**

Safavian (2022) assessed the effectiveness of two consecutive years of mechanical-ripping as a method of weed control on the Aslhey River from 2019 to 2020. The study was conducted in the same location as one of the three sampling lines used for invertebrate monitoring in the current study, and the other two invertebrate monitoring lines were in adjacent ripped areas. These three lines had to be moved when a gravel extraction permit was enacted, but they were relocated to an area where the same method of weed ripping had been undertaken at the same time, and using the same method.

Ripping was undertaken in August 2019 and repeated in August 2020. Safavian (2022) found that mean weed species richness was lower after ripping and weed cover remained low relative to control sites. Immediately after control weed cover was significantly reduced, but recovery began immediately, increasing from 5% in September 2019 to a peak of 45% in December. In adjacent un-ripped plots, weed cover ranged from 60-100%. Lupin and Broom contributed the most to weed cover, and ripping reduced lupin from 30% down to 5% but as above it quickly resprouted.

Although ripping occurred just 3-4 months before invertebrate monitoring commenced in each of the first two study years, no effect of weed ripping on invertebrate abundance, diversity or community composition was detected. However, from May 2021 onwards, there were regular winter and summer flood events which constantly reset the weed-load on all sites and any difference in habitat caused by weed-ripping would have quickly been lost. As such, the impact of weed cover and mechanical weed-control on invertebrates remains to be tested with sufficient rigor, and no substantial conclusions can be made with confidence.

6.4 Objective 4: Impacts of severe flooding on the Ashley River

Record the impact on the invertebrate community of severe flooding that occurred in the Ashley River in 2021.

There were no strong and consistent correlations between the abundance or diversity of invertebrates detected during each sampling session on the Ashley River and flood events, or river flow more generally. However, abundance was exceptionally high following two large flood events in December 2021 and February 2022. In both cases the sharp increase in abundance could be attributed to flies in the family Chironomidae, which accounted for 87% of the 7,355 invertebrates collected in January 2022 and 76% of 8,453 specimens collected in early March 2022. Some chironomid flies are tolerant of low oxygen environments and are often indicators of increased sediment and decaying organic matter, which is consistent with conditions immediately after flood events. Interestingly, chironomid abundance was much lower in the three subsequent sampling sessions, each of which occurred after moderate flood events. The increase in a small number of chironomid species following the two large floods explains why species richness did not increase in proportion to abundance during these sampling periods.

Notably, flooding did not appear to have a negative impact on the number of different species detected per sampling session. In particular, diversity detected with malaise traps before and after the December 2021 flood did not differ, while pitfall diversity was substantially lower in December 2021 after a period of low flow, compared to January 2022 two weeks after flows exceeded 460 m^3/s. Overall, our data suggests floods did not have long lasting effects on the abundance or diversity of the Ashley River invertebrate community.

7 Acknowledgements

Sampling on the Ashley River was undertaken by the Ashley Rakahuri Rivercare Group who persisted in the face of regular floods and trap vandalism and consistently delivered to the highest standard. We would particularly like to thank Judith Hughey and Nick Ledgard for rallying the troops each year and Grant Davey for drone footage of the monitoring sites. Thanks also to the DOC Twizel Project River Recovery team, and especially rangers Jennifer Schori and Sam Turner and seasonal staff Ella Sussex, Maddy Pye and Jolene O'Connor for monitoring on the Cass and Aparima Rivers. Behind the scenes, Jessica Chen (DOC) was key to the success of this project, providing many hundreds of hours sorting and identifying insect specimens. Thank you also to Susan Anderson (DOC) who provided the foundation for this work in leading the Tasman invertebrate project in 2005. This project was jointly funded by the Environment Canterbury Braided River Research Initiative and DOCs Project River Recovery.

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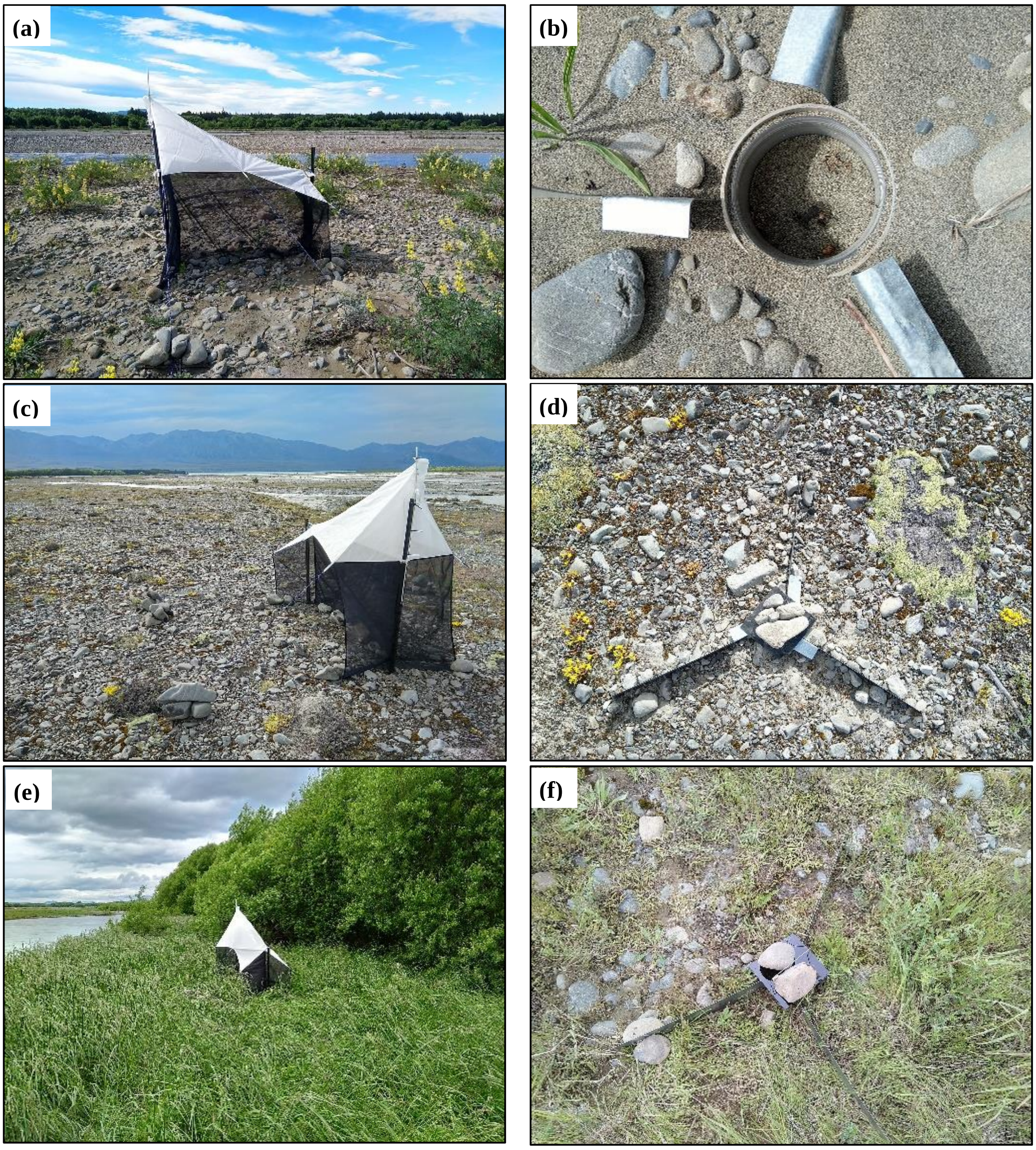
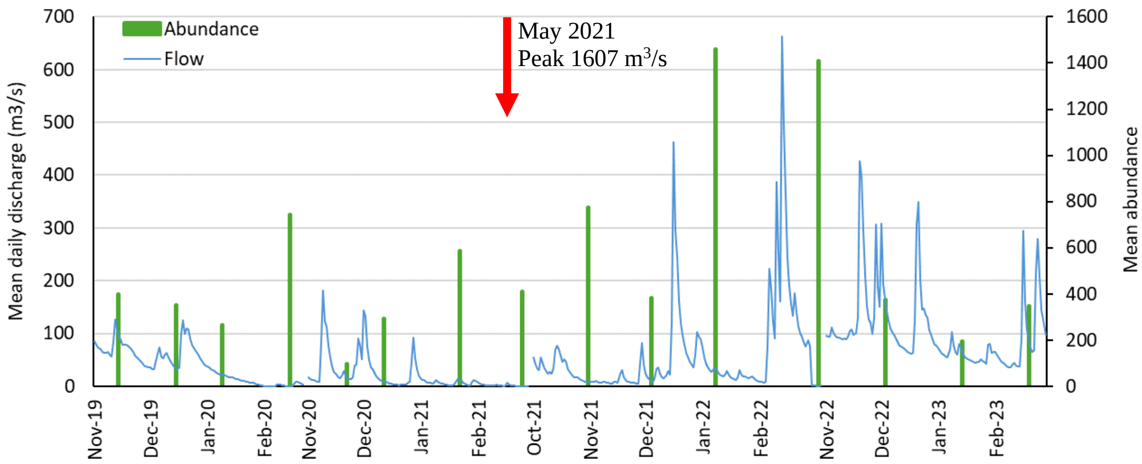
9 Appendix

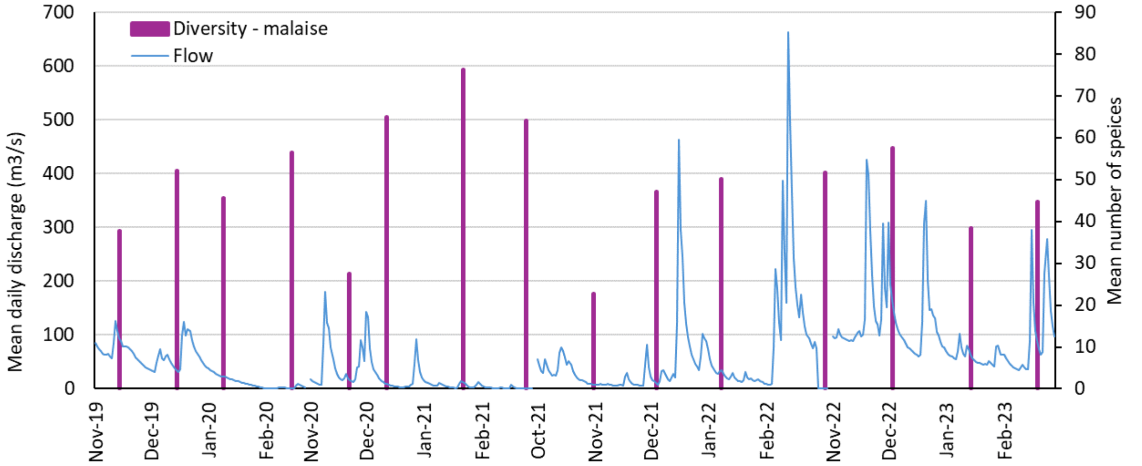
Appendix 1: Relative locations of the Cass, Ashley and Aparima Rivers where invertebrate sampling was carried out for this study.

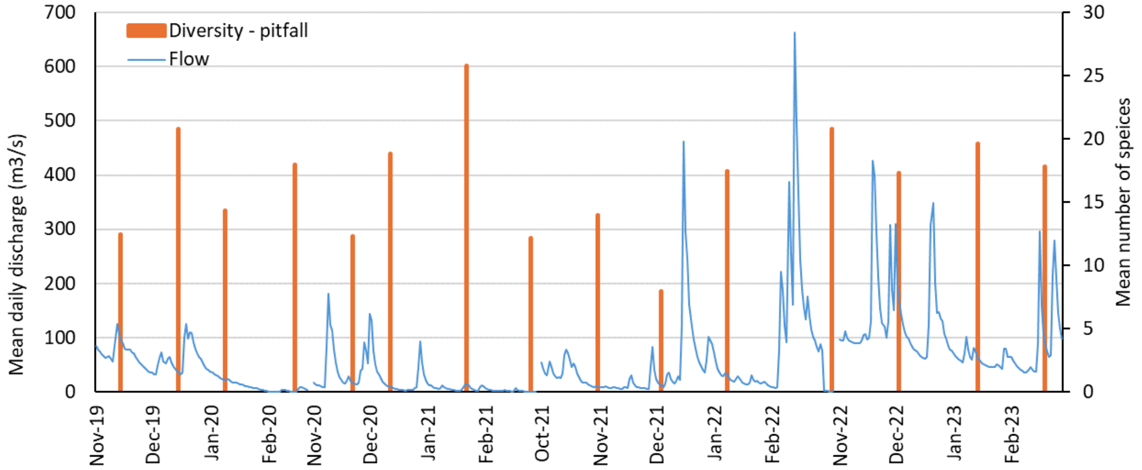
Appendix 2: Malaise traps (left) and pitfall traps (right) set up on the **(a, b)** Ashley, **(c, d)** Cass and **(e, f)** Aparima Rivers showing the different habitat types. Not all sampling areas on the Aparima were as weedy as that shown. Sand filling traps was occasionally a problem on the Ashley River during high winds so microhabitats with larger gravels and pebbles were selected whenever possible. *Photos*: Jennifer Schori (reproduced from interim report June 2021).



Appendix 3: Daily average flow (m^3/s) (recorded at 5 min intervals at the Rangiora Traffic Bridge) for the Ashley River overlaid with **(a)** total abundance of specimens captured in malaise and pitfall traps, **(b)** mean number of species captured per malaise trap, and **(c)** mean number of species captured per pitfall line, over the 15 sampling sessions between November 2019 and February 2023. Large winter floods also occurred outside the sampling season in 2021 and 2022 (largest indicated by red arrow) *Flow data curtesy of Tony Gray, Environment Canterbury.*







Appendix 4: Ashley River in flood February 2022 (Photos: Grant Davey) and sampling sites 1- 6 (Photos: Maddy Pye) at the end of the study in February 2023 showing the buildup of silt on all sites and limited difference in weediness after a season marked by 4 large floods.

Appendix 5: Complete list of morpho-species detected over 2-4 sampling years on the Cass (Cs), Ashley (As) and Aparima (Ap) Rivers. Unique taxa were identified to the lowest possible level and named as a Recognisable Taxonomic Unit (RTU). Group = taxonomic order if know, otherwise class or phylum; Guild = primary functional groups (*Detr = detritivore*, *Fung = fungivore, Herb = herbivore, Omni = omnivore, Para = parasitoid*, *Pred = predator, Ukn = unknown*); Traps in which taxon was detected are indicated but *P = pitfall and M = malaise*.

SEPARATE FILE