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Effects of small ponds on stream water quality and macroinvertebrate communities

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Abstract Six small constructed ponds (surface area 500–7500 m^2, catchment area 28–158 ha) in rural and native forest catchments in the Auckland region had poorer water quality than the streams they replaced. Temperature (24°C) and dissolved oxygen (DO) (4 mg/litre) criteria were exceeded for up to 46% and 84% of days, respectively, during a critical 40-day summer period. The poor conditions found in ponds, even within undeveloped native forest catchments, indicated that the physical characteristics of ponds (e.g., lack of shade, organic sediments) affected water quality independently of other factors (e.g., land use, riparian protection). The frequency and severity of the exceedences were related to pond size, retention time, and catchment land use; the most degraded conditions were found in rural ponds with largest surface areas and longest retention times. Ponds affected water quality and macroinvertebrate communities downstream. Exceedences of temperature and DO criteria occurred more frequently and were more severe downstream than upstream of ponds. Ponds in rural catchments increased mean daily stream temperatures 3.1–6.6°C during the critical summer period, and temperature differences were three times higher than those in bush catchments (0.8–2.0°C). Elevated temperatures were observed for hundreds of metres downstream owing to the slow rate of cooling (1°C/100 m), expanding the extent of adverse effects well beyond the "footprint" of the pond. Macroinvertebrate community composition (sample area 1–3 m^2) and values of four commonly used metrics appeared to be significantly affected by ponds in rural and native forest catchments. These finding have important management implications that should lead to modifications (e.g., breaching dams) of the estimated 4500 existing ponds in the Auckland region, where possible, and restrictions on proposals for new "on-line" ponds.

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INTRODUCTION

Rivers and streams are dammed for many reasons, and there is extensive literature on the effects of impoundments on downstream ecosystems (see reviews in Baxter 1977; Petts 1984). However, most research has focused on the effects large reservoirs have on river ecosystems, and there appears to be very little information available on the effects of small impoundments on headwater streams. The focus of this study was on small dams (ponds) constructed in stream channels ("on-line") for stock watering, irrigation, and their perceived aesthetic amenity. Ponds are also a common treatment device for trapping sediment from construction sites and for stormwater treatment in urban areas. We defined small ponds as having a dam crest height of <5 m, maximum depth <5 m, surface area <10 000 m^2 (1 ha), and catchment area <300 ha.

The damming of water alters numerous physical, chemical, and biological processes in streams. Sediment and associated contaminants accumulate, and areas of shallow open water promote radiant heating and increase primary production (Wetzel 2001). Physical (e.g., temperature, sedimentation) and chemical (e.g., dissolved oxygen) changes affect biological communities directly (e.g., toxicity) and indirectly (e.g., altered food webs) across local (e.g., in ponds) and catchment (e.g., migration barriers) scales (Baxter 1977; Petts 1984).

Ponds and the streams affected by ponds represent a significant freshwater resource in the Auckland region of New Zealand's North Island. The location and catchment area of ponds with a surface area >200 m^2 that appeared on 1:10 000 scale colour aerial photographs (taken in 1999) have been recorded and digitised (ARC 2003). There were c. 4500 ponds in the 5000 km^2 Auckland region, with a total upstream catchment area representing c. 9% of the region. Approximately 30% (1500) were on streams mapped on 1:50 000 scale topographic maps (NZMS 260 series) with inter-quartile ranges (25th to 75th) in surface area and catchment area of 532–1876 m^2 and 5.9–36.6 ha, respectively.

Most ponds in the Auckland region are constructed (ARC 2003). Natural ponds and shallow open water wetlands are limited to the sand hills along the east and west coasts where windblown sand has blocked the drainage of small catchments (ARC 2003). Most on-line ponds were built by the placement of earth across the gullies of streams, often at the point where a farm track crosses a stream. They are robust structures that can last for many decades with little or no maintenance (ARC 2003). They are rarely removed, and thus their numbers tend to increase over time. Many have no present use or their intended use has been abandoned.

The ecological functions of many New Zealand streams are strongly linked to the sea, particularly in the Auckland region with a high proportion of small catchments and short distances to the sea. Many native fish species (McDowall 1990) and the New Zealand freshwater shrimp *Paratya curvirostris* (Carpenter 1983) depend upon access to the sea to complete their life cycles. Although many native fish are good climbers, rural ponds often have outlet structures that are a significant barrier to the passage of climbing and non-climbing native fish species (Boubée et al. 1999).

No published studies were found in the New Zealand literature that addressed the water quality conditions either upstream or downstream of small constructed rural ponds. Overseas literature on the subject appears to be limited to a single study of nine rural ponds in Michigan, United States (Lessard & Hayes 2003). In that study, stream temperatures increased up to 5°C below ponds affecting macroinvertebrate and fish communities, and elevated temperatures persisted for considerable distances (2–3 km) downstream. Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa and coldwater fish species were most affected. Literature on the effects of beaver (*Castor canadensis*) ponds on stream ecosystems in North America suggests that ponds can significantly alter stream hydrology, chemistry, temperature regimes, and biota downstream (Naiman et al. 1988, 1986; Alexander 1998; Collen & Gibson 2000).

The Auckland region was almost completely covered in trees before human settlement (Beever 1981), and it is likely that much of the endemic flora and fauna evolved in shaded environments. As a result, lack of shade vegetation along streams and its effect on temperature has been identified as a key stressor in New Zealand streams (Quinn et al. 1994; Rutherford et al. 1999; Quinn 2000).

The primary objectives of this study were to assess the adverse effects of ponds on water quality and living resources downstream, to complement what is already known about ponds as potential migration barriers (Joy & Death 2001). Investigations of the adverse effects of ponds in the Auckland region were considered important because of the lack of relevant studies in New Zealand, the large number of ponds, the unique aquatic fauna, and indications from laboratory studies that many New Zealand invertebrate and fish species are adversely affected by elevated water temperatures common in ponds and streams in the Auckland region.

Our study addressed the following questions: (1) what are the water quality conditions (temperature and dissolved oxygen (DO)) in ponds; (2) what effects do ponds have on downstream water quality; (3) what is the extent of elevated temperature and depressed DO downstream, and at what rate (°C/m stream) does temperature recover; and (4) do ponds affect the stream biota (macroinvertebrates)?

MATERIALS AND METHODS

Sampling sites

Six ponds were selected that were behind earthen dams along the course of perennial 1st order streams (i.e., on-line) as shown on 1:50 000 scale (NZMS260 series) topographic maps (Fig. 1). The ponds ranged in catchment area (28–158 ha), surface area (500–7500 m^2), and maximum depth (1.5–3.0 m) (Table 1), and were comparable in size to the on-line ponds on mapped streams in the Auckland region (ARC 2003). Photographs of the six ponds are shown in

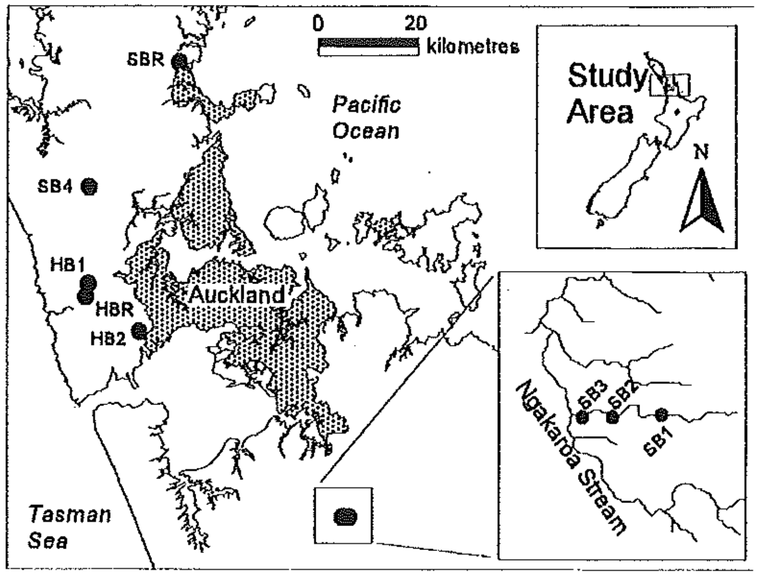

Fig. 1 Locations of sampling sites in the Auckland region, New Zealand; stippled area is the Auckland urban area. (SB, soft-bottomed; HB, hard-bottomed.)

Fig. 2. Streams were selected that had continuous flow during dry weather conditions and had similar physical characteristics upstream and downstream of the ponds. All ponds discharged continuously during the survey period based upon continuous water level recorder data and biweekly visits to the sites. Streams were wadeable with mean wetted width range of 0.8–2.1 m and mean Thalweg depth range of 0.2–0.4 m (Table 1). All unpublished data are available from the Auckland Regional Council (ARC, 21 Pitt Street, Auckland).

Ponds SB1, SB2, SB3, and SB4 were in unshaded rural streams with soft-bottomed (SB) substrata, and were selected to represent the most common type of pond in the region in size, geology, land use, and physical habitat. Catchment land use was dominated by livestock, commercial row crops, and vineyards. Livestock had direct access to both streams and ponds, and there was little or no native riparian vegetation. Rural ponds SB1, SB2, and SB3 were in sequence on an unnamed tributary of the Ngakaroa stream, and were used to measure temperature changes in the streams between the ponds. Their relative position (moving downstream) is summarised as follows: SB1 pond >550 m stream >SB2 pond >300 m stream >SB3 pond >100 m to Ngakaroa stream confluence (Fig. 1). This stream was in geology of the Franklin volcanic formation. Bed material was fine sediment with coarse detritus covered with macrophytes and overlying hard clay and occasional volcanic rocks (ARC unpubl. data). Rural pond SB4 was in geology of the Waitemata sandstone and siltstone formation on an unnamed tributary of the Waikoukou stream (tributary of the Ararimu stream). Bed material was similar to the Ngakaroa stream (ARC unpubl. data).

Ponds HB1 and HB2 were in undeveloped native forest catchments, and chosen to assess conditions with minimal human disturbance (Fig. 1). These sites had hard-bottomed (HB) substrata; there were no ponds in native forest catchments with SB substrata. Catchments were 99% indigenous forest, and vegetation along the stream network and around the ponds comprised mature native trees. Development was limited to isolated houses and a single secondary road in each catchment. Bush pond HB1 was located on an unnamed tributary of the Waitupu stream (tributary of the Waitakere River). Bush pond HB2 was located on Cochrane stream (tributary of the Oratia stream), and was slightly more developed than pond HB2. Both catchments are along the eastern slope of the Waitakere Ranges in geology of the Waitakere Group formation. Bed material was dominated by cobbles and boulders.

Sampling protocol

Water chemistry

Eighteen temperature loggers were deployed over a 175-day summer/autumn period from 21 December 2001 to 18 June 2002 (Stowaway Tidbit, Onset Computer Applications; accuracy ±0.2°C). There were three sites at each pond (upstream, in pond, and

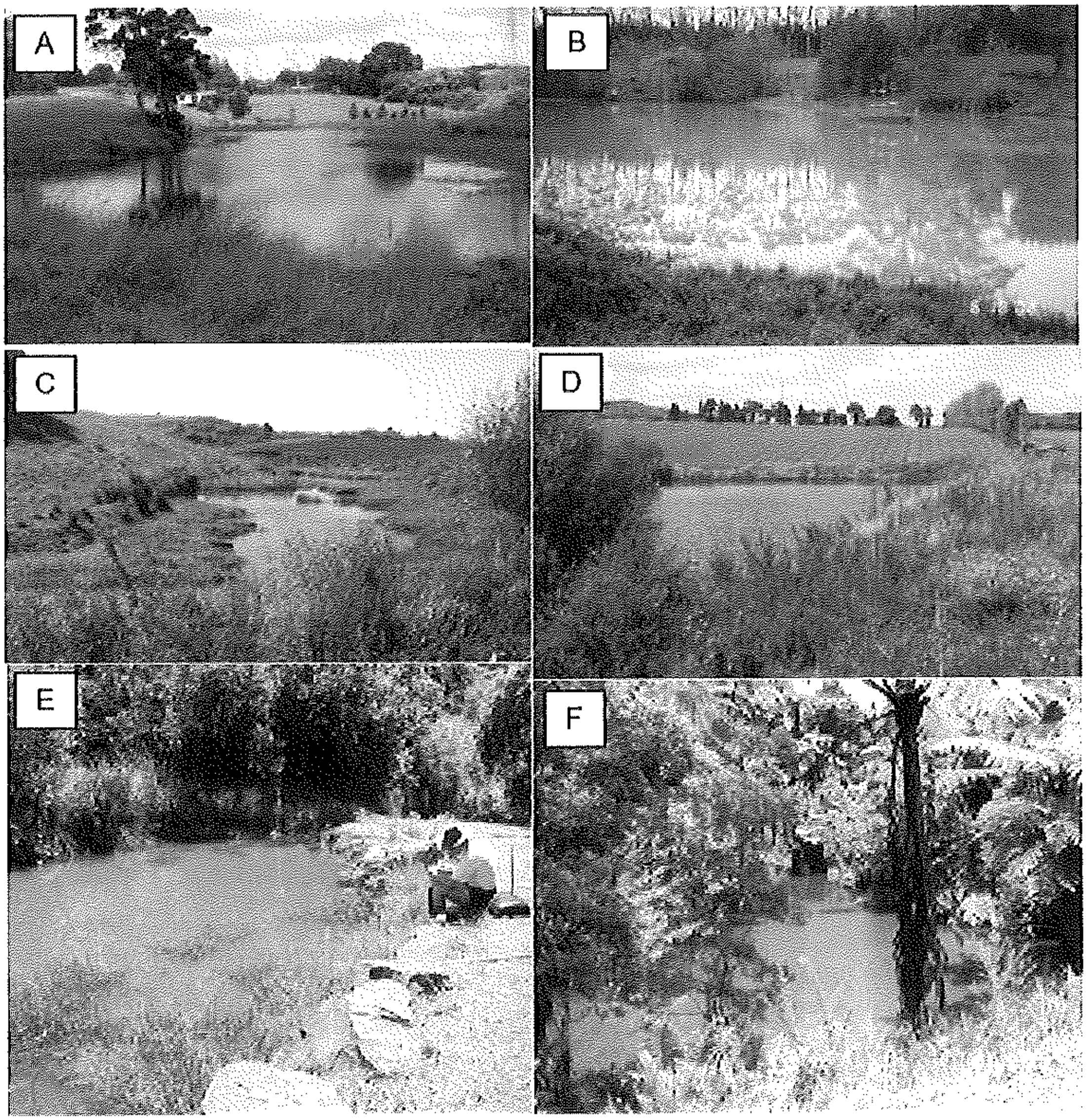

Fig. 2 Ponds: A, SB4; B, SB3; C, SB2; D, SB1; E, HB1; and F, HB2. (SB, soft-bottomed; HB, hard-bottomed.)

downstream), and loggers recorded every 15 min. Loggers in ponds were placed midway between the inlet and outlet, 100–500 mm below the surface, at least 200 mm above the bottom, and 2–5 m from the pond edge. Loggers in streams were placed in the main flow of the channel. Stream sites upstream of the pond had flow and no influence of the pond (30–50 m from the inlet of the pond), and sites downstream of the pond were 10–20 m from the pond outlet. Data quality was assessed fortnightly by comparison with a hand-held meter (Yellow Springs Instruments model 85; accuracy temperature ±0.1°C, DO ±0.3 mg/litre).

Nine multi-parameter datasondes recorded temperature and DO every 15 min at the same locations as the temperature loggers (Greenspan CS304 Smart-Sensor or Multi-Parameter Sonde; accuracy temperature ±0.4%, DO ±1.5%). Datasondes were deployed over two time periods: 20 December 2001 to 1 March 2002 (ponds SB1, SB2, and SB3) and 1

Table 1 Physical characteristics of pond and stream sites; altitude in m a.s.l. (SB, soft-bottomed; HB, hard-bottomed; ref, reference; NA, not applicable; ND, no data.) Data available from the Auckland Regional Council (ARC database, 21 Pitt Street, Auckland.)

										Stream			
ID	Type	Geographic coordinates (easting:northing)	Altitude (m)	Distance from tide (km)	Catch area (ha)	Pond area (m^2)	Max. depth (m)	Dam height (m)	Aspect ratio (L:W)	Wetted width (m)	Thalweg depth (m)	Measured flow (litre/s)*	Residence time (days)†
SB1	rural SB	2686779:6443451	182	18	116	7200	3.0	4.0	2:1	1.0	0.3	0.6	208.0
SB2	rural SB	2686102:6443451	170	17	140	600	1.5	2.0	5:1	1.2	0.3	1.1	4.7
SB3	rural SB	2685754:6443503	155	17	158	7500	3.0	4.0	2:1	1.4	0.2	3.2	41.0
SB4	rural SB	2646225:6494552	20	38	125	4400	3.0	5.0	4:1	2.0	0.3	3.0‡	25.5
SBR	ref SB	2660330:6513934	30	3	47	NA	NA	NA	NA	0.8	0.3	ND	ND
HB1	bush HB	2646152:6479040	100	12	100	500	1.5	2.4	2:1	2.1	0.4	1.1	3.8
HB2	bush HB	2654097:6472194	200	13	28	1000	3.0	5.0	3:1	1.2	0.2	2.9	6.9
HBR	ref HB	2646100:6478300	100	13	238	NA	NA	NA	NA	2.9	0.5	ND	ND

*Averages of upstream and downstream flows, from 1 to 3 gaugings carried out under low flow conditions (except SB4).
†(pond volume) / (measured flow), where volume estimated as (pond area) × (max. depth) / 2.
‡Catchment area-weighted estimate from gauged site 1 km downstream.

March 2002 to 2 June 2002 (ponds SB4, HB1, and HB2). The units were calibrated before deployment following manufacturer's specifications, and cleaned and checked for accuracy fortnightly. Data quality for DO was assessed by comparison with readings taken with the calibrated hand-held meter.

Macroinvertebrates

Biotic effects were assessed by comparison of the macroinvertebrate communities in stream reaches that had comparable substrata and habitat conditions upstream and downstream of the ponds. Comparable sites sampled for macroinvertebrates included above/below the two bush ponds (HB1, HB2), and above rural pond SB1 and below rural pond SB3. The SB1/SB3 comparison was used to assess the cumulative effects of ponds SB1, SB2, and SB3 that were in close proximity on the same stream. Macroinvertebrate samples were also collected at reference sites that met the following criteria: perennial flow; 100% of catchment in regenerating native bush at least 50 years old; human use limited to recreation (e.g., walking tracks); no ponds, dwellings, reservoirs, or roads; and remote (>300 m) from roads and other access points. The HB reference site was in the Cascade stream (site HBR) and the SB reference site was in the Otanerua stream (site SBR) (Fig. 1). Sites with HB and SB substrata were sampled and analysed separately because they have different organic and inorganic substrata, support different macroinvertebrate communities (Maxted et al. 2003), and used different sampling methods (Stark et al. 2001).

Macroinvertebrate samples were collected between 27 May and 17 June 2002. Three replicate samples were collected at each site with one exception; a single sample was taken upstream of pond HB1 because the area of riffle habitat was limited. The sampling method for the SB and HB sites followed Protocol C2 and C1 of the New Zealand national protocols, respectively (Stark et al. 2001).

At SB sites, samples were collected from bank margins and macrophytes by aggressively jabbing a 0.33 m wide D-frame dip-net (500 μm mesh) over a distance of 1 m, followed by 2–3 cleaning sweeps to collect dislodged organisms. Ten such collections were combined in a 500-μm-mesh sieve bucket for a total sample area of c. 3.0 m^2. The SB reference site (SBR) was sampled using five collections of bank margin as above, and 5 m of submerged wood (50–300 mm diam.) gently removed, placed over the mouth of the sieve bucket and brushed by hand under pouring water; total sample area c. 3.0 m^2.

At HB sites, macroinvertebrate samples were collected in riffle habitats (turbulent flow) using the same net as above. HB sites were sampled by disturbing (kicking and brushing by hand) c. 0.17 m^2 of cobble/boulder substratum 0.5 m upstream of the net (0.33 m × 0.5 m = 0.17 m^2). Six separate riffle areas covering the range of velocities were sampled and combined in a 500-µm-mesh sieve bucket for a total sample area of c. 1.0 m^2.

Samples were processed following Protocol P1 (coded abundance) of the New Zealand national protocols (Stark et al. 2001). Sample material was separated into 2–4 size fractions using stacked 0.5–4.0 mm Endecott® sieves, and sorted in white-bottomed trays. One to five specimens of each taxon were removed, identified to the taxonomic level required for the macroinvertebrate community index (MCI) (Stark 1985; Stark et al. 2001), and given one of the following abundance codes: rare (R) = 1–4 animals/sample; common (C) = 5–19; abundant (A) = 20–99; very abundant (VA) = 100–499; and very very abundant (VVA) = 500+ (Stark 1998).

Assessment criteria (temperature and DO)

Criteria for temperature and DO were selected to assess adverse effects. Bioassay data for New Zealand aquatic fauna are limited (Quinn et al. 1994; Richardson et al. 1994). We have reported results at three levels of effect to avoid misleading results that might occur when using a single criterion, and to allow for future interpretations as new effects data become available. These criteria corresponded to slight, moderate, and severe adverse effects.

The available literature on New Zealand fish and invertebrates is largely based on controlled laboratory tests that derived % mortality data for the test organism after exposure to constant conditions for a specified duration (Cox & Rutherford 2000a). In natural systems these variables fluctuate owing to radiant heating and cooling, and to photosynthesis and respiration. However, few studies have addressed the effects of variable temperature and DO exposures to New Zealand native fish or invertebrates. Data on daily variable exposures are limited to a single study of temperature effects on two native invertebrate species (*Potamopyrgus antiopodarum* and *Deleatidium autumnale*) (Cox & Rutherford 2000a), whose authors concluded that adverse effects were bounded by the daily mean and maximum values (Cox & Rutherford 2000b). Considering this lack of data, we assessed adverse effects by comparing daily extreme values (maximum temperature, minimum DO) with criteria derived from acute exposure bioassay results.

Acute mortality for most native New Zealand fauna tested to date occurred above 25°C. LT_{50} values (lethal temperature that killed 50% of the test organisms over 10-min duration) for nine species of native fish (excluding *Anguilla* species) ranged 27.0–31.9°C (Richardson et al. 1994). Juvenile and adult *Anguilla* species (eels) were considerably more tolerant than other fish species (LT_{50} ranged 34.8–39.7°C), and thus were not used in setting assessment criteria. Native invertebrate species were more sensitive than fish, where LT_{50} values (24-h exposure) for 12 species ranged 25.9–32.4°C (Quinn et al. 1994). Simons (1986) recommended that a maximum value 3°C below the lowest LT_{50} would allow for a margin of safety, and would be 22.9°C in this instance. No effect levels (LT_0) ranged 23.6–26.0°C (Quinn et al. 1994) for several invertebrate taxa commonly found in Auckland SB streams, indicating that adverse effects may begin to occur above 22°C. Based upon the test data and these interpretations, slight, moderate, and severe adverse effects were estimated to occur above 22°C, 24°C, and 26°C, respectively.

Dean & Richardson (1999) studied the sensitivity of seven native fish, rainbow trout (*Oncorhynchus mykiss*), and the native shrimp (*Paratya curvirostris*) to 1, 3, and 5 mg/litre DO (48-h exposure). There was no mortality to any tested species at 3 and 5 mg/litre, except rainbow trout with 14.3% mortality at 3.0 mg/litre over 36 h. There was 100% mortality to banded kokopu (*Galaxias fasciatus*), common smelt (*Retropinna retropinna*), common bully (*Gobiomorphus cotidianus*), and rainbow trout at 1 mg/litre over 12 h, and 61.1% mortality to inanga (*Galaxias maculatus*) over 48 h. Behavioural effects (live fish surfacing or out of water) were observed at 5 mg/litre over 12 h for common smelt. They concluded that full protection would be provided for native fish and invertebrates by adopting the United States Environmental Protection Agency criteria for salmonid waters, which is 6 mg/litre DO as a daily minimum (USEPA 1986). Based upon the test data and these interpretations, slight, moderate, and severe adverse effects were estimated to occur below 6.0 mg/litre, 4.0 mg/litre, and 2.0 mg/litre, respectively.

Data analysis

Tidbit and datasonde temperature data were combined to produce a complete record over the study period; gaps in datasonde data were filled using Tidbit data. Mean values calculated for each day (mean daily temperature) were used for upstream/downstream comparisons. Stream temperature changes owing to the presence of a pond were determined by subtracting the daily mean value upstream from the daily mean value downstream.

Each 15-min measurement in temperature and DO was compared to the three effect criteria. An exceedence was defined as a measurement that was less than the DO criterion or greater than the temperature criterion. The total number of exceedences and the number of days with at least one exceedence provided "% of time" and "recurrence" exposure measures, respectively. Statistics on the number and percentage of exceedences were compiled for the critical period in mid-summer when flows were low and stress on the aquatic community from high temperature and low DO was expected to be most severe. The combination of three severity levels (effect criteria) and two exposure periods provided a more accurate assessment of ecological effects than would result using a single effect level and exposure period. Pond volume was used to relate water quality conditions to pond size, and estimated by multiplying half the maximum depth × the pond surface area.

The critical period was determined using rainfall data from the nearest Auckland Regional Council rain gauge and stream temperature data recorded at site HBR (ARC database, 21 Pitt St, Auckland). Summer stream temperatures at site HBR were highest and most stable (14.5–16.5°C) from 15 January to 1 April 2002. Rainfall was variable during this period but generally lacked significant events (>20 mm) between 20 January and 2 March; there were 10 events <10 mm and one 25 mm rainfall event. The period 21 January to 1 March, therefore, was used as the critical period for temperature data analysis at all sites, and for DO data collected on the first datasonde deployment (SB1, SB2, SB3). A later period (1 March to 1 April 2002) was used for DO data analysis on the second datasonde deployment (SB4, HB1, HB2). A shorter and less variable period (8 days, 29 January to 5 February) was used to assess the recovery of elevated temperature between ponds SB1, SB2, and SB3. Rainfall during this period was limited to a single 5 mm event. Mean values over this period were used to relate temperature change with reach length.

Non-metric multi-dimensional scaling (MDS) of macroinvertebrate data was used to illustrate patterns in community similarity between sites (Systat 10, Systat Software Inc. 2000). Separate ordinations were carried out for sites with SB and HB substrata. The coded abundance data were converted to percentages using the lower range for each category (e.g., common, C = 1.01%) as set out in Stark (1998). Spearman rank correlation (Datadesk 6.1, Data Description Inc. 1999) between axis scores and the relative abundance of individual taxa was used to indicate taxa showing monotonic gradients along the MDS axes (Maxted et al. 2003). *Paratya* (shrimp) and Collembola were removed from the data set before MDS analyses. *Paratya* were not included because they migrate to the sea for the development of early life stages, and their presence is influenced by physical barriers (natural and constructed) and distance from the sea (Carpenter 1983). *Paratya* was only found at the SB reference site (SBR). Collembola were not included because they are semi-aquatic.

Macroinvertebrate metrics calculated were number of taxa (total richness), number of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT richness), MCI (Stark 1985), and semi-quantitative MCI (SQMCI) (Stark 1998). MCI tolerance values for each taxon were those given by Stark et al. (2001). Mean and standard deviation (SD) statistics were calculated from replicate data ($n = 3$), and significance of differences ($P < 0.05$) between site pairs was determined using a simple t test (SigmaStat 2.03, 1997, Systat Software Inc.). Site pairs included upstream/downstream, upstream/reference, and downstream/reference. The dominant taxa at each site were used to illustrate taxon-specific relative abundance, which was done by combining replicate samples and compiling taxa with A, VA, and VVA abundance classes.

RESULTS

The summer of 2002 was typical for stream flows in the Auckland region. Stream low flows were equivalent to the mean annual low flow (normal) in the west (HB1, HB2) and northwest (SB4), and equivalent to the 1:5 year low flow return frequency (slightly drier than normal) in the south (SB1, SB2, SB3) (ARC database, 21 Pitt St, Auckland).

Pond water quality

Pond sites had the highest number of measurements and days exceeding criteria for temperature (Table 2) and DO (Table 3) during the critical period. There were no DO data for ponds SB2 and SB3 owing to fouling of the sensors. All data are available from the Auckland Regional Council (ARC database, 21 Pitt St, Auckland).

Table 2 Minimum (min.), maximum (max.), and mean diel ranges in temperature of six ponds and stream waters above and below them. The percentages (%) of 15-min measurements and days exceeding three temperature criteria (22, 24, 26°C) during critical summer period 21 January to 1 March 2002 are also shown; pond values in bold type. (*n*, number of measurements or days.)

		Temp. (°C)			% measurements				% days			
						Temp. criterion				Temp. criterion		
Pond	Site	Min.	Max.	Diel range	*n*	>22	>24	>26	*n*	>22	>24	>26
SB1	up	14.0	18.4	1.6	3494	0	0	0	35	0	0	0
	pond	**19.2**	**26.6**	**2.8**	**3927**	**50**	**13**	**1**	**38**	**95**	**39**	**8**
	down	16.4	27.7	4.5	3916	34	11	4	38	82	42	21
SB2	up	13.3	20.1	2.3	3927	0	0	0	38	0	0	0
	pond	**17.0**	**25.5**	**2.2**	**3930**	**14**	**1**	**0**	**39**	**36**	**5**	**0**
	down	16.3	25.5	4.4	3121	28	6	0	30	77	30	0
SB3	up	14.4	25.0	5.8	3102	10	1	0	32	59	9	0
	pond	**19.5**	**26.5**	**2.8**	**3913**	**62**	**14**	**1**	**39**	**97**	**46**	**10**
	down	18.8	26.6	3.2	3932	57	16	1	38	95	50	11
SB4	up	14.2	22.7	2.6	3936	1	0	0	40	13	0	0
	pond	**19.2**	**26.8**	**1.2**	**3925**	**34**	**4**	**<1**	**40**	**83**	**25**	**5**
	down	19.5	23.5	3.2	335	14	0	0	2	100	0	0
HB1	up	14.2	18.1	0.6	3935	0	0	0	40	0	0	0
	pond	**14.4**	**25.6**	**1.5**	**3934**	**6**	**1**	**0**	**40**	**43**	**13**	**0**
	down	12.8	22.0	3.0	3933	<1	0	0	40	5	0	0
HB2	up	13.5	17.3	0.8	3936	0	0	0	41	0	0	0
	pond	**15.4**	**19.5**	**0.6**	**3936**	**0**	**0**	**0**	**41**	**0**	**0**	**0**
	down	15.2	19.7	1.8	3936	0	0	0	41	0	0	0

Table 3 Minimum (min.), maximum (max.), and mean daily diel range in dissolved oxygen (DO) of six ponds and stream waters above and below them. The percentages (%) of 15-min measurements and days exceeding three DO criteria (6, 4, 2 mg/litre) during critical summer periods 21 January to 1 March 2002 (SB1, SB2, SB3) and 1 March to 2 April (SB4, HB1, and HB2) are also shown; pond values in bold type. (*n*, number of measurements or days.)

		DO (mg/litre)			% measurements				% days			
						DO criterion				DO criterion		
Pond	Site	Min.	Max.	Diel range	*n*	<6	<4	<2	*n*	<6	<4	<2
SB1	up	4.0	6.2	0.5	2718	93	0	0	26	100	0	0
	pond	**4.1**	**11.7**	**3.3**	**3496**	**6**	**0**	**0**	**34**	**26**	**0**	**0**
	down	–	–	–	–	–	–	–	–	–	–	–
SB2	up	0.0	5.7	1.4	3116	100	50	11	29	100	66	34
	pond	–	–	–	–	–	–	–	–	–	–	–
	down	–	–	–	–	–	–	–	–	–	–	–
SB3	up	6.4	9.1	1.6	2487	0	0	0	24	0	0	0
	pond	–	–	–	–	–	–	–	–	–	–	–
	down	1.9	7.4	1.9	489	96	61	1	3	100	100	33
SB4	up	–	–	–	–	–	–	–	–	–	–	–
	pond	**0.0**	**9.5**	**2.4**	**2985**	**90**	**64**	**34**	**31**	**97**	**81**	**58**
	down	4.9	9.3	1.8	2448	12	0	0	25	40	0	0
HB1	up	7.7	9.6	0.3	2938	0	0	0	30	0	0	0
	pond	**3.8**	**8.4**	**1.4**	**2852**	**52**	**1**	**0**	**28**	**86**	**7**	**0**
	down	8.1	9.5	0.5	3113	0	0	0	32	0	0	0
HB2	up	8.6	9.4	0.2	2639	0	0	0	27	0	0	0
	pond	**0.9**	**7.2**	**1.1**	**2582**	**96**	**69**	**9**	**25**	**100**	**84**	**24**
	down	4.6	8.9	0.7	2637	56	0	0	27	81	0	0

Rural ponds (SB1, SB2, SB3, SB4) had higher temperatures and a greater percentage of temperature exceedences than bush ponds (HB1, HB2). Maximum daily temperatures in rural ponds ranged 25.5–26.8°C compared with 19.5–25.6°C in bush ponds (Table 2). Rural ponds exceeded the "moderate" criterion (24°C) on 5–46% of days compared with 0–13% in bush ponds. Rural ponds SB1, SB3, and SB4 exceeded the "severe" criterion (26°C) on 8%, 10% and 5% of days, respectively (Table 2). The frequency of exceedence was associated with pond size as measured using volume (Fig. 3). The two largest ponds (SB1, SB3) had the highest percentage of criteria exceedences (Table 2). Bush pond HB2 was the only pond that did not exceed a temperature criterion.

Exceedences of temperature criteria were associated with diel variations. Ponds with the highest percentage of exceedences also had the highest diel range. For example, mean diel temperature range over the critical period was 1.2–2.8°C in rural ponds compared with 0.6–1.5°C in bush ponds (Table 2). The highest diel range on an individual day was 4.7°C (pond SB3).

Exceedences of DO criteria were more variable between sites and appeared to be less influenced by pond size and catchment land use than temperature. The large rural pond SB4 and the small bush pond HB2 were hypoxic, exceeding the severe criterion (<2 mg/litre) on 58% and 24% of days, respectively (Table 3). These two ponds also exceeded the moderate criterion (<4 mg/litre) on most days (81% and 84%, respectively). Pond SB4 was anoxic with 0 mg/litre DO. Rural pond SB1 and bush pond HB1 had minimum DO values of 3.8 mg/litre and 4.1 mg/litre, respectively, and few exceedences of the 4.0 mg/litre criterion. Exceedences of the slight criterion (<6.0 mg/litre) occurred frequently (26–100% of days) at all pond sites (Table 3). Exceedences of DO criteria were also associated with diel variations. Mean diel ranges in ponds over the critical period were 1.1–3.3 mg/litre (Table 3). The highest diel ranges on an individual day were 5.1 mg/litre (pond SB1), 4.6 mg/litre (SB4), 3.1 mg/litre (HB1), and 4.0 mg/litre (HB2).

Effects of ponds on downstream water quality

Mean daily stream temperatures increased below all ponds throughout summer and autumn (Fig. 4). Increases varied between ponds but were generally higher below rural ponds than bush ponds, and higher in summer than autumn. Stream temperature increases >1°C continued through late summer and

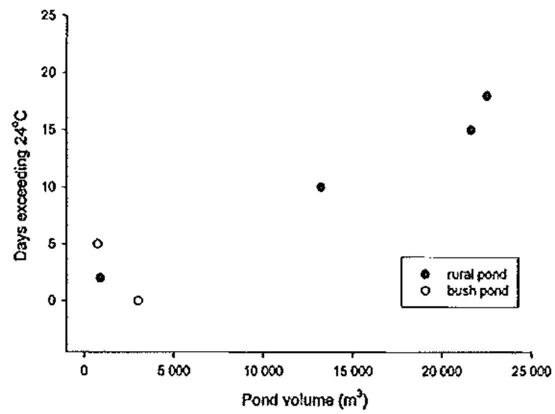

Fig. 3 Pond size (volume, m³) in relation to the number of days temperature exceeded 24°C during a 39-day critical summer period (21 January to 1 March 2002).

autumn (March–June) below rural pond SB4, but increases were minimal (<1°C) after 1 April below the two bush ponds (HB1, HB2). Rural pond SB1 had a cooling effect on stream temperature during autumn (Fig. 4). The interquartile range (25–75%) in mean daily stream temperature increase was three times higher below rural ponds (3.1–6.6°C) than bush ponds (0.8–2.0°C) during the critical period (Fig. 5). Maximum stream temperature increases below bush and rural ponds were 2.7°C (HB2) and 7.5°C (SB1), respectively.

Temperature criteria were also exceeded in streams during the critical period, and were more severe and occurred more frequently below ponds. The moderate criterion (24°C) was exceeded 6–16% of the time and on 30–50% of days below rural ponds SB1, SB2, and SB3; there were few exceedences upstream of these ponds (Table 2). The severe criterion (26°C) was exceeded on 21% and 11% of days below the two largest rural ponds (SB1, SB3, respectively). The slight criterion (22°C) was exceeded 14–57% of the time and on 77–100% of days below the four rural ponds, whereas exceedences upstream and downstream of the bush ponds were negligible (Table 2).

Two stream sites downstream of the ponds exceeded the moderate and severe criteria for DO during the critical period. The moderate criterion (4 mg/litre) was exceeded 50% and 61% of the time and on 66% and 100% of days below ponds SB1 (upstream of pond SB2) and SB3, respectively (Table 3). The severe criterion (2 mg/litre) was exceeded 11% and 1% of the time and on 34% and 33% of days at these sites, respectively. These sites

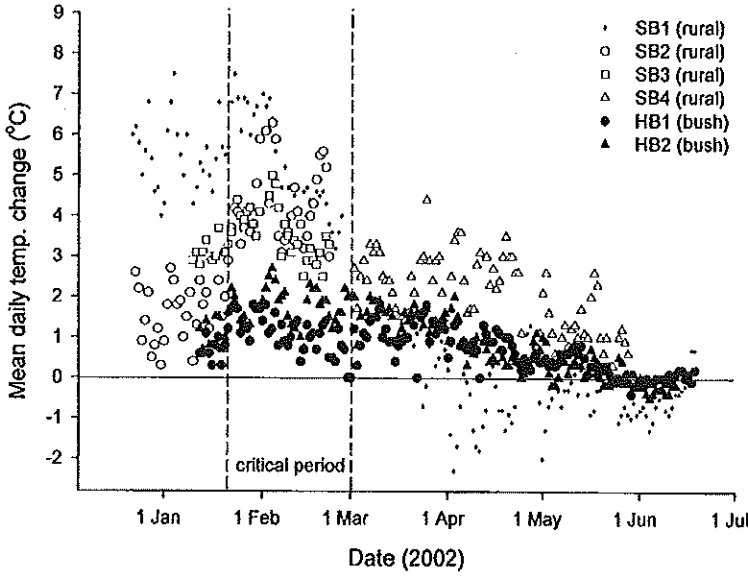

Fig. 4 Change in mean daily stream temperature downstream of rural ponds (SB1, SB2, SB3, SB4) and native bush ponds (HB1, HB2) relative to upstream values over the full sampling period (21 December 2001 to 18 June 2002); the critical (dry summer) period (21 January to 1 March) is within the dashed lines.

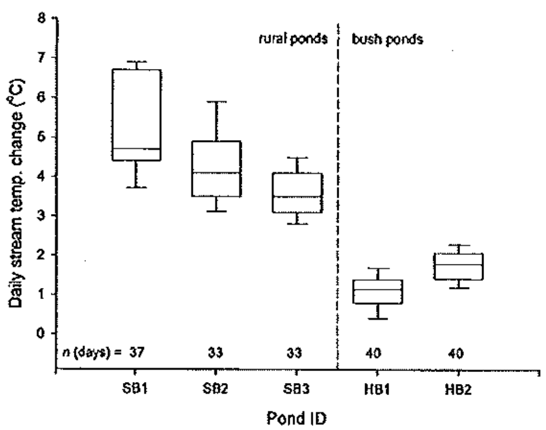

Fig. 5 Percentile distribution of mean daily increase in stream temperature below rural ponds (SB1, SB2, SB3) and native bush ponds (HB1, HB2) during the critical summer period 21 January to 1 March 2002. (Boxes, 25th and 75th percentiles; lines, median values; wiskers, 5th and 95th percentiles.)

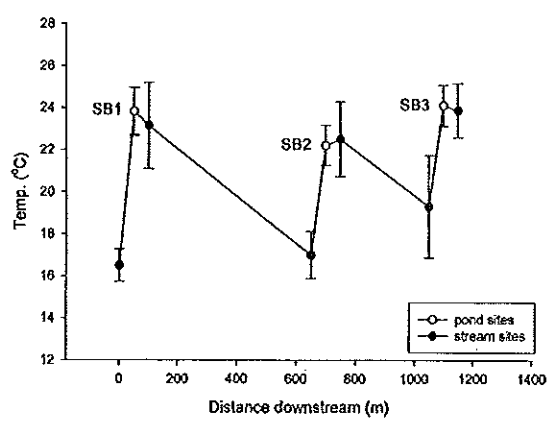

Fig. 6 Stream temperature (°C, mean ± 1 SD) in relation to distance downstream (m), including three sequential ponds (SB1, SB2, SB3), during a 8-day summer period (29 January to 5 February 2002).

also had the lowest minimum DO values recorded (0 mg/litre and 1.9 mg/litre) in streams (Table 3). In rural streams, the slight criterion (6 mg/litre) was exceeded on all days upstream of pond SB1, downstream of pond SB1 (upstream of pond SB2), and downstream of pond SB3 (Table 3). In bush ponds, there were no exceedences of the slight criterion upstream of either of the bush ponds, but this criterion was exceeded on 81% of days downstream of bush pond HB2 (Table 3).

Exceedences of temperature and DO criteria were related to diel fluctuations. Mean diel temperature ranges were higher at rural sites (1.2–5.8°C) than bush sites (0.6–3.0°C), and generally higher at stream sites than pond sites (Table 2). The highest diel temperature ranges were downstream of the rural ponds (3.2–4.5°C), and the lowest were upstream of the bush ponds (0.6–0.8°C). Mean diel DO ranges were higher at pond sites (1.1–3.3 mg/litre) than stream sites (0.2–1.9 mg/litre) (Table 3). The highest diel DO ranges were in rural ponds SB1 (3.3 mg/litre) and SB4 (2.4 mg/litre), and the lowest were upstream of the bush ponds HB1 (0.3 mg/litre) and HB2 (0.2 mg/litre) (Table 3).

Elevated stream temperatures below ponds recovered with distance downstream during an 8-day dry summer period (29 January to 5 February). The mean stream temperature dropped (i.e., recovered) 5.0°C over a 550 m section of stream (c. 1°C/100 m) between ponds SB1 and SB2 (Fig. 6). A similar rate of recovery was found over the 300 m distance between ponds SB2 and SB3 (Fig. 6).

Effects of ponds on macroinvertebrate communities

Differences between sites were shown by abundant taxa (A, VA, or VVA classes; >20 individuals per sample). The number of abundant EPT taxa followed a pattern of disturbance (reference site > upstream site > downstream site) in both native bush (HB) and rural (SB) sites (Table 4). There were 12 abundant EPT taxa at reference site HBR compared with six taxa upstream of pond HB2, and four taxa downstream. There were three abundant EPT taxa (*Arachnocolus, Triplectides, Zephlebia*) at reference

Table 4 Dominant taxa (EPT taxa in bold type) at reference sites, and upstream and downstream of ponds; by stream type and relative abundance. (HB, hard-bottomed; SB, soft-bottomed; VVA, very very abundant; VA, very abundant; A, abundant.)

Site ID:	(HBR) HB reference	(HB2) Upstream	(HB2) Downstream	(SBR) SB Reference	(SB1) Upstream	(SB3) Downstream
VVA	***Deleatidium***	none	***Orthopsyche***	none	*Potamopyrgus*	none
VA	***Coloburiscus*** ***Orthopsyche*** *Potamopyrgus*	***Austroclima*** ***Coloburiscus*** ***Zephlebia***	***Coloburiscus*** *Potamopyrgus*	***Zephlebia***	Ostracoda *Physa* ***Polyplectropus*** *Xanthocnemis*	Oligochaeta
A	*Archichauliodes* ***Austroclima*** ***Austroperla*** *Chironomus* Elmidae ***Hydrobiosella*** ***Neozephlebia*** ***Nesameletus*** ***Olinga*** ***Stenoperla*** ***Zelandobius*** ***Zephlebia***	Amphipoda *Archichauliodes* ***Austroperla*** ***Deleatidium*** ***Orthopsyche***	***Deleatidium*** ***Zephlebia***	***Arachnocolus*** ***Triplectides***	*Austrosimulium* *Cricotopus* Oligochaeta *Paradixa* *Polypedilum*	Hirudinea Ostracoda

Table 5 Mean values (± 1 SD, $n = 3$) for four macroinvertebrate metrics in streams upstream and downstream of ponds, and for soft-bottomed (SBR) and hard-bottomed (HBR) reference sites. Italicised values differ significantly from the reference site; downstream values in bold type differ significantly from upstream values ($P < 0.05$). (EPT, Ephemeroptera, Plecoptera, Trichoptera; ND, no replicate data, $n = 1$.)

Site ID	Location	Total richness	EPT richness	MCI	SQMCI
SB1	Upstream	18.0 (2.0)	*2.0 (0.0)*	*79.1 (4.3)*	*4.2 (0.0)*
SB3	Downstream	12.3 (3.1)	***0.3 (0.6)***	*69.2 (6.9)*	***2.0 (0.6)***
SBR	SB reference	15.0 (3.6)	6.7 (1.5)	115.0 (3.6)	6.1 (0.7)
HB1	Upstream	23.0 (ND)	14.0 (ND)	128.7 (ND)	7.4 (ND)
HB1	Downstream	*19.7 (3.8)*	*11.7 (2.1)*	128.7 (6.3)	7.8 (0.3)
HB2	Upstream	*28.3 (2.3)*	*14.7 (1.2)*	132.9 (4.1)	7.9 (0.1)
HB2	Downstream	*21.7 (5.5)*	*11.3 (2.5)*	132.0 (4.0)	8.2 (0.1)
HBR	HB reference	34.3 (0.6)	20.0 (1.0)	131.7 (9.9)	7.5 (0.5)

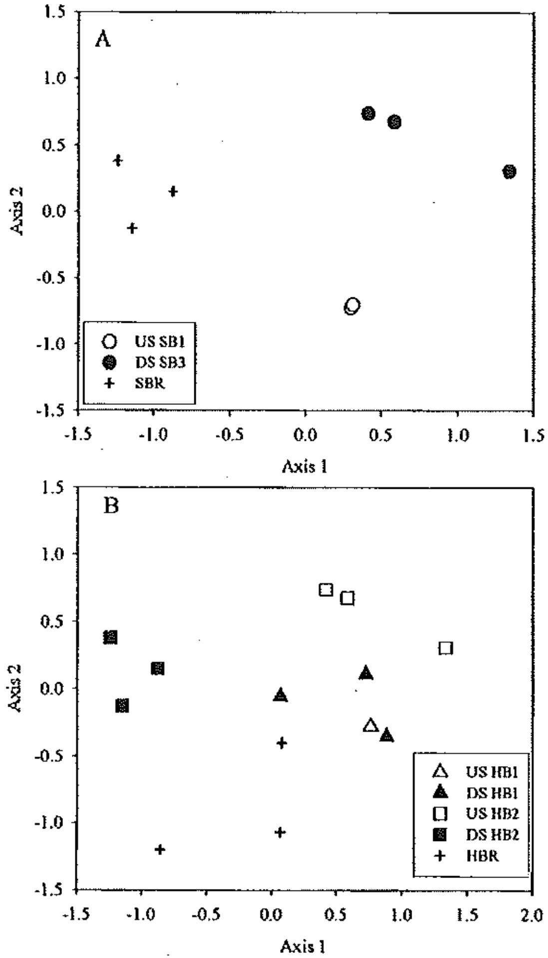

Fig. 7 Non-metric multidimensional scaling ordination plots of relative abundance macroinvertebrate data for stream sites upstream (open symbols) and downstream (filled symbols) of ponds, and reference sites in 100% native bush (crosses). Separate ordinations for sites with: A, soft-bottomed (SB) (stress value = 0.01); and B, hard-bottomed (HB) (stress value = 0.11) substrata.

site SBR compared with one EPT taxon (*Polyplectropus*) upstream of pond SB1 and no EPT taxa downstream (Table 4). There were two EPT taxa (*Austroclima*, *Austroperla*) abundant upstream of bush pond HB2 that were not abundant downstream, and one EPT taxon (*Polyplectropus*) abundant upstream of rural pond SB1 that was not abundant downstream of rural pond SB3 (Table 4).

Differences between sites were also indicated by values of four biological metrics. In general, richness (total and EPT) and organic pollution metric (MCI and SQMCI) values were higher at reference sites than pond sites, and higher upstream than downstream of ponds (Table 5). Mean total richness was 28.3 taxa (±2.3) upstream of bush pond HB2 compared with 21.7 (±5.5) downstream, and 18.0 taxa (±2.0) upstream of rural pond SB1 compared with 12.3 taxa (±3.1) downstream. Mean EPT richness and SQMCI values were significantly lower downstream of pond SB3 than upstream of pond SB1 (Table 5). None of the four metrics were significantly different upstream and downstream of the bush ponds (HB1, HB2).

Separate MDS ordinations were produced for sites with SB (Fig. 7A) and HB (Fig. 7B) substrata. There was separation between reference sites and ponds sites, and between sites upstream and downstream of ponds. SB and HB ordinations had good stress values of 0.01 and 0.11, respectively.

The MDS plot of SB sites showed separation between the reference site (SBR) and pond sites along axis-1 (Fig. 7A). Taxa considered relatively sensitive to poor water quality (e.g., *Zephlebia* and *Triplectides*) were negatively correlated with axis-1 scores ($r_S > -0.76$), indicating that these taxa were less abundant at pond sites (Table 6). Conversely, taxa more tolerant of poor water quality (e.g., Oligochaeta, Hirudinea, Platyhelminthes, and Lymnaeidae) were positively correlated with axis-1 scores ($r_S > 0.74$), indicating that they were poorly represented at the reference site. Axis-2 further separated sites upstream and downstream of the three rural ponds (Fig. 7A). The high negative correlations for *Potamopyrgus* ($r_S = -0.85$) and *Psilochorema* ($r_S = -0.73$) with axis-2 scores suggested that the observed separation on this axis was partly owing to the reduced numbers of these two taxa (Table 6).

The MDS plot of HB sites showed separation between the reference site (HBR) and pond sites along axis-2 (Fig. 7B). Separation was greater for pond HB2 compared with pond HB1. Pollution sensitive taxa (e.g., *Ichthybotus* and *Olinga*) were negatively correlated with axis-2 scores ($r_S > -0.70$), indicating that these taxa were most common at the reference site (Table 6). Amphipoda and Tabanidae were positively correlated with axis-2 scores ($r_S > 0.77$), indicating that they were rare at the reference site. Axis-1 further separated sites upstream and downstream of pond HB2 (Fig. 7B). The positive correlations for *Austroclima* ($r_S = 0.79$) and *Zephlebia* ($r_S = 0.78$) with axis-1 scores suggested that the observed separation was partly owing to the reduced numbers of these two mayfly taxa (Table 6).

DISCUSSION

Ponds had severely degraded water quality for temperature and DO, and were more degraded than the streams they replaced. All ponds significantly affected downstream macroinvertebrate communities, irrespective of substrata (HB, SB) and land use (rural, bush). The severity of water quality degradation appeared to be related to land use and pond size. Ponds in rural catchments were more degraded than those in bush catchments, and large ponds were more degraded than small ponds. Ponds in bush catchments had poor water quality despite minimal anthropogenic stress, indicating that the physical characteristics of ponds (e.g., lack of shade, accumulated organic sediments) may be important factors operating in isolation from other factors (e.g., land use, riparian protection).

Our results were similar to those reported in studies of ponds in North America. Rural ponds in Michigan, United States increased stream temperatures to a similar degree (~1–5°C), and were associated with changes in invertebrate community composition (Lessard & Hayes 2003). Smith et al. (1991) found increased stream temperature and reduced DO and benthic invertebrate species richness 1 m downstream of a beaver pond. Alexander (1998) also found significant increases in stream temperatures and effects on the invertebrate community below two beaver ponds. Our results were also similar to those observed in studies of stormwater retention ponds in urban catchments. For example, Lieb & Carline (2000) found increases in temperature downstream of a stormwater retention pond in a Pennsylvania, United States headwater stream.

We expect that temperature is a major stressor in and downstream of ponds, with elevated temperatures persisting for hundreds of metres downstream owing to the slow rate of stream water cooling. A temperature model developed for Auckland streams (NIWA 2003) and applied to the Ngakaroa tributary site estimated a similar rate of cooling (1°C/100 m) below ponds SB1 and SB2. Lessard & Hayes (2003) and Alexander (1998) also found elevated stream temperatures and adverse effects on fish and macroinvertebrates for hundreds of metres downstream of ponds. Our findings indicate that adverse effects of elevated temperatures in ponds extended well beyond the footprint of the pond surface area.

The rate of heating and cooling in streams is effected by numerous physical factors including catchment area, cloud cover, air temperature, wind speed, channel width, water depth, velocity, travel time, and channel/canopy shade (Rutherford et al. 1997, 2004). No attempt was made in this study to determine the relative importance of these factors

Table 6 Spearman rank correlation coefficients (r_s) for associations between relative abundance of individual taxa and MDS axis scores, soft-bottomed (SB) and hard-bottomed (HB) sites listed separately; only taxa with $|r_s|$ >0.70 listed and shown in bold type.

	SB sites			HB sites	
Taxon	Axis 1	Axis 2	Taxon	Axis 1	Axis 2
Ecnomidae	**–0.71**	0.07	Amphipoda	0.19	**0.77**
Harrisius	**–0.71**	0.07	*Aphrophila*	–0.14	**–0.71**
Hirudinea	**0.84**	0.53	*Austroclima*	**0.79**	–0.05
Lymnaea	**0.90**	0.42	*Ichthybotus*	–0.26	**–0.73**
Neozephlebia	**–0.76**	0.05	*Latia*	–0.26	**–0.73**
Oeconesidae	**–0.71**	0.07	*Olinga*	–0.26	**–0.73**
Oligochaeta	**0.95**	0.36	Ptilodactylidae	**–0.81**	0.12
Ostracoda	**0.90**	0.32	Tabanidae	–0.09	**0.77**
Paradixa	–0.09	**–0.80**	*Zephlebia*	**0.77**	0.43
Paranephrops	**–0.78**	0.03			
Platyhelminthes	**0.74**	–0.10			
Potamopyrgus	–0.43	**–0.85**			
Psilochorema	–0.29	**–0.73**			
Pycnocentria	**–0.71**	0.07			
Sciomyzidae	0.02	**–0.78**			
Triplectides	**–0.76**	0.05			
Zephlebia	**–0.84**	0.13			

affecting the heating (on sunny days) and cooling (on shady days) in streams. Direct comparisons between the sites we surveyed were possible (Fig. 6) because the stream reaches between ponds had similar physical characteristics, and the measurements were taken over the same 8-day sunny summer period.

Quinn et al. (1994) speculated that temperature stress may be an important factor affecting the distribution of New Zealand stream invertebrates, particularly in the North Island. The 30–50% of days exceeding 24°C below ponds would be expected to result in reductions in the relative abundance of temperature sensitive species, particularly mayflies and stoneflies. This is what we observed with reductions in the relative abundance of EPT taxa below ponds in SB (e.g., *Zephlebia, Polyplectropus*) and HB (e.g., *Austroclima, Austroperla*) streams. Although many native fish are more tolerant of high temperatures than invertebrates, we observed maximum temperatures (>26°C) that exceeded the proposed lethal threshold for some invertebrates, and approached the lethal threshold for banded kokopu (*Galaxias fasciatus*, LT_{50} 28.5°C) and common smelt (*Retropinna retropinna*, LT_{50} 28.3°C) (Richardson et al. 1994). DO may also be a major stressor in and downstream of ponds. Low DO may be related to factors we did not measure such as the age of the pond, and the depth and organic content of the sediments.

Sites with the largest number of days exceeding temperature and DO criteria also had the largest diel fluctuations. The lack of shade around ponds and along rural streams leads to higher temperature maxima and greater diel ranges in DO compared to shaded systems (Rutherford et al. 1999), resulting in the more frequent exceedences of criteria we observed. Littoral shade vegetation around ponds was not sufficient to mitigate adverse effects related to temperature and DO because ponds in bush catchments also exhibited frequent exceedences of effect criteria.

Ponds in SB geology in rural catchments affected the macroinvertebrate communities downstream to an extent detected by metrics and MDS ordinations. It was not surprising that the SB site upstream of pond SB1 had a different community than the SB reference site (SBR) because these ponds are in a catchment with intensive rural land use, unshaded channels, and stock access. What was surprising was that the presence of ponds resulted in an additional shift in the community to even more pollution-tolerant groups, including the complete loss of EPT taxa. This further shift was detected by the MDS ordination and two metrics (EPT richness, SQMCI).

Ponds in HB geology in native bush catchments had no significant effect on mean metric values, although there was separation in the MDS ordination between upstream and downstream of pond HB2. This separation may be owing to the lower degree of development in the catchment above bush pond HB2 compared with pond HB1, indicating the sensitivity of New Zealand fauna to subtle changes in land use. Mean metric values for total richness and EPT richness upstream and downstream of both bush ponds were significantly different from the HB reference site (HBR), indicating that even limited development in these bush catchments affected the macroinvertebrate communities. These effects on richness did not translate into significant effects on pollution indices (MCI, SQMCI).

Stream flows and temperatures in 2002 were typical of the summer period in the Auckland region, and therefore water quality and biotic conditions reported in this study are likely to occur frequently. The results should also be viewed with caution as the number of sites represented a limited dataset.

Management implications

Our results indicated that ponds are severely degraded freshwater environments affecting a substantial proportion of the stream network. Policies that encourage the damming of streams need to change, and incentives are needed to encourage the removal (i.e., dam breaching) of unused ponds. Lessard & Hayes (2003) came to a similar conclusion. Breaching should be during dry summer periods to allow sediments to dewater and consolidate, minimising transport downstream. For ponds currently in use (e.g., stock watering, irrigation, stormwater management), alternative sources of water (e.g., groundwater) and their relocation off-line should be evaluated. Shade vegetation along the perimeter of the bush ponds (and along streams in the catchment) was not sufficient to avoid adverse effects. Thus, the planting of shade vegetation along the perimeter of large unshaded ponds may not avoid adverse effects owing possibly to the small percentage of the water surface that would be shaded. The planting of shade vegetation may improve temperature conditions around small ponds but would have little effect on depressed DO levels caused by the accumulation and decomposition of organic sediments. On-line ponds are constructed for a variety of reasons, and are commonly viewed by landowners as an aesthetic amenity. Our results provide new insights into the extent and severity of their ecological effect footprint that we hope will lead to further study, improved policy and management, and enlightened public awareness.

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