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# Biotransference and biomagnification of selenium copper, cadmium, zinc, arsenic and lead in a temperate seagrass ecosystem from Lake Macquarie Estuary, NSW, Australia

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**Abstract**

In this study the biotransference of selenium copper, cadmium, zinc, arsenic and lead was measured in a contaminated seagrass ecosystem in Lake Macquarie, NSW, Australia, to determine if biomagnification of these trace metals is occurring and if they reach concentrations that pose a threat to the resident organisms or human consumers. Selenium was found to biomagnify, exceeding maximum permitted concentrations for human consumption within carnivorous fish tissue, the highest trophic level examined. Selenium concentrations measured within carnivorous fish were also above those shown to elicit sub-lethal effects in freshwater fish. As comparisons are made to selenium concentrations known to effect freshwater fish, inferences must be made with caution. There was no evidence of copper, cadmium, zinc or lead biomagnification within the food web examined. Copper, cadmium, zinc and lead concentrations were below concentrations shown to elicit adverse responses in biota. Copper concentrations within crustaceans *M. bennettae* and *P. palagicus* were found to exceed maximum permitted concentrations for human consumption. It is likely that copper concentrations within these species were accumulated due to the essential nature of this trace metal for many species of molluscs and crustaceans. Arsenic showed some evidence of biomagnification. Total arsenic concentrations are similar to those found in other uncontaminated marine ecosystems, thus arsenic concentrations are unlikely to cause adverse effects to aquatic organisms. Inorganic arsenic concentrations are below maximum permitted concentrations for human consumption.


*Keywords:* Trace metals; Selenium; Copper; Cadmium; Zinc; Arsenic; Lead; Lake Macquarie Australia; Food web; Biotransference; Biomagnification

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## 1. Introduction

The question of whether trace metals biomagnify through food chains is only partially resolved. Biddinger and Gloss (1984) examined selenium biomagnification and found positive biotransference from phytoplankton (*Scenedesmus dimorphus*) to zooplankton (*Daphnia* and *Cyclops* spp.) to fish (*Puntius arulius*). Patrick and Loutit (1976) reported biomagnification of chromium, copper, manganese, iron, lead and zinc from bacteria to tubificid worms to fish within a food chain, below an effluent output. Several other studies have described the biomagnification of selenium, zinc and mercury (Bowles, Apte, Maher, Kawei, & Smith, 2001; Cabana, Tremblay, Kalff, & Rasmussen, 1994; Harris, Fabris, Statham, & Tawfik, 1979; Martin & Coughtrey, 1982; Peakall & Lovett, 1972; Timmermans, Van Hattum, Kraak, & Davids, 1989). However, other studies have reported that biomagnification does not occur for these and other elements (Amiard, Amiard-Triquet, Metayer, Marchand, & Ferre, 1980; Bargagli, 1993; Liu, Yang, Hu, Harrison, & Price, 1987; Metayer, Amiard, Amiard-Triquet, & Marchand, 1980; Mikac, Branica, & Harrison, 2001; Prahalad & Seenayya, 1986; Ward, Correll, & Anderson, 1986).

There is little information regarding trace metal behaviour within Australian estuarine ecosystems, particularly with respect to biotransference and biomagnification. Biotransference is the transfer of trace metals from a food source to consumer. Biomagnification is when an increase in trace metal concentration occurs through at least two trophic levels in a food chain. We have previously reported that the southern end of Lake Macquarie, NSW is grossly contaminated with selenium and receiving inputs of cadmium, zinc and copper (Kirby, Maher, & Harasti, 2001; Kirby, Maher, & Krikowa, 2001). We also suspect that the lake is receiving arsenic and lead inputs from coal wastes and other sources. Seagrass beds of *Zostera capricornii* are widespread in Lake Macquarie, covering approximately 1220 ha, with a total biomass of approximately 1454 tonnes, forming the basis of the estuarine ecosystem (Furner, 1979). Seagrass ecosystems are of prime ecological importance, as they have high biodiversity and are a nursery to many commercial and recreational fish species (Bell, Burchmore, & Pollard, 1978; Bell & Pollard, 1989; Phillips & McRoy, 1980). If biomagnification of trace metals is occurring, then elevated trace metal concentrations in higher trophic group organisms could pose a threat to the organisms themselves or to human consumers.

In this study we have measured the selenium copper, cadmium, zinc, arsenic and lead concentrations in organisms from a contaminated seagrass ecosystem in Lake Macquarie, to determine if biomagnification of these trace metals is occurring, and if they reach concentrations that pose a threat to the resident organisms or human consumers.

## 2. Study site

Lake Macquarie is located approximately 20 km south of the industrial city of Newcastle at a latitude of 151° 30′ E and longitude of 33° 00′ S (King, 1986). Lake

Macquarie has a catchment area of approximately 622 km$^2$, making it the largest estuary in New south Wales (NSW), extending 22 km in a north–south direction with a maximum width of 9 km, an average depth of 6.7 m, and a maximum depth of approximately 11 m. For a map and further details refer to Kirby, Maher, and Krikowa (2001). Lake Macquarie's narrow shallow mouth, results in poor tidal flushing, with an inter-tidal range of approximately 6 mm–1.25 m. Three coal-fired power stations (two of which are still in operation), several collieries, a sewage treatment works as well as urbanisation occur on the southern shores of the lake (Batley, 1987).

Lake Macquarie supports large commercial and recreational fisheries. There are approximately 50 active commercial fishing licences on Lake Macquarie, and approximately 96 species of fish, molluscs and crustaceans are currently caught commercially (NSW Fisheries, 1995). The annual fish catch in the lake is estimated at 300,000–400,000 kg, with a market value of approximately A$1.0 million dollars per annum (NSW Fisheries, 1995).

## 3. Materials and methods

### 3.1. Sample collection

Sampling occurred on the 13–17 March 1999. Sampling was near Vales Point, in the southeastern end of Lake Macquarie. This site was chosen due to high concentrations of trace metal contamination observed in previous studies by Kirby, Maher, and Krikowa (2001) and Peters et al. (1999). Sampling yielded a total of 157 organisms representing 18 different species (Table 1).

Most fish samples were collected using gill nets (length: 30 mm, drop: 3 m, mesh: 37–50 mm). Nets were set from the bank, extending out into the bay at a 90° angle to shore. *Portunus palagicius* were also caught using these nets. Individuals of similar size within each species were selected to minimize variations in metal concentration due to size of the organism. Fish were then pithed and stored individually in acid washed sealable plastic bags.

*Metapenaeus bennettae* and *Atherinomorus ogilbyi* were caught after dark using a light and a dip net. *Anadara trapezium* were collected from seagrass beds and *Bembicium auratum* were collected from rocks. Both the *A. trapezia* and *B. auratum* were held in a cooled container of clean lake water for 48 h to depurate, before being placed separately in acid washed, sealable plastic bags.

Zooplankton samples were collected by dragging a net (mesh size 76–80 microns) approximately 4 m behind a boat at night. Ten 100-m transects were conducted to gain sufficient tissue for analysis. Zooplankton samples were then placed in acid washed 100 ml screw top plastic bottles (Johns, MC0008, Aust.).

*Zostera capricornii* and *Enteromorpha* sp. samples were collected at a depth of approximately 1 m. Samples were removed from the substrate, rinsed in lake water to remove particulate material and placed in acid washed plastic bags.

Detritus was obtained by collecting all material from ten 0.5×0.5-m plots.

Table 1
Trace metal concentrations in Lake Macquarie seagrass organisms

| Species | Common name | Species code | *n* | Size (mm) | Selenium (μg/g) | Copper (μg/g)[a] | Cadmium (μg/g) | Zinc (μg/g) | Arsenic (μg/g) | Lead (μg/g) |
|---|---|---|---|---|---|---|---|---|---|---|
| **Carnivores** | | | | | | | | | | |
| *Tylosurus gavialoides* | Stout longtom | LT | 10 | 406–544 | 4.4±0.4 | 6.3±0.3 | 0.04±0.01 | 36±4 | 1.7±1.7 | 0.48±0.29 |
| *Platycephalus fuscus* | Dusky flathead | FH | 7 | 310–389 | 9.3±0.5 | 4.1±0.3 | 0.010±0.003 | 19±1 | 1.2±0.2 | 0.82±0.80 |
| *Acanthopagrus australis* | Yellow fin bream | SB | 10 | 113–162 | 5.7±0.5 | 0.57±0.14 | 0.28±0.13 | 21±1 | 6.2±0.4 | 1.5±0.6 |
| *Gerres subfasciatus* | Silver biddy | YB | 10 | 119–155 | 9.2±0.7 | 1.9±0.4 | 0.001±0.005 | 40±4 | 6±1 | 0.54±0.34 |
| **Omnivore** | | | | | | | | | | |
| *Monacanthus chinensus* | Fan-belly leatherjacket | LJ | 9 | 116–198 | 4.1±0.7 | 0.27±0.23 | 0.65±0.07 | 21±3 | 11±2 | 0.42±0.14 |
| **Detritivores** | | | | | | | | | | |
| *Mugil cephalus* | Sea mullet | SM | 10 | 320–390 | 4±1 | 2.8±0.4 | 0.074±0.055 | 26±4 | 3.0±0.3 | 0.10±0.05 |
| *Portunus palagicus* | Blue swimmer crab | BS | 9 | 168–176 | 3.7±0.3 | 63±7 | 1.7±0.4 | 101±17 | 27±4 | 0.54±0.07 |
| *Metapenaeus bennettae* | Greasyback prawn | PR | 10 | 41–58 | 3.6±0.4 | 77±7 | 1.5±0.8 | 80±3 | 8±1 | 4±2 |
| *Eunicida* sp. | Polychaete worm | PW | 10 | 67–113 | 4.8±0.4 | 14±2 | 3.2±0.7 | 86±4 | 5.8±0.3 | 12±2 |
| **Planktivores** | | | | | | | | | | |
| *Anadara trapezia* | Sydney cockle | BV | 10 | 34–41 | 5.2±0.6 | 7±1 | 0.030±0.002 | 126±14 | 4.8±0.9 | 1.1±0.1 |
| *Atherinomorus ogilbyi* | Hardyhead | HH | 9 | 41–56 | 5.7±0.4 | 0.83±0.55 | 0.150±0.017 | 151±8 | 1.2±0.2 | 0.31±0.16 |
| **Herbivores** | | | | | | | | | | |
| *Girella tricuspidata* | Luderick | LU | 10 | 206–299 | 3.4±0.4 | 4.1±0.5 | 0.001±0.001 | 20±1 | 2.3±0.6 | 0.70±0.26 |
| *Bembicium auratum* | Gastropod | GP | 10 | 13–19 | 2.2±0.2 | 88±15 | 14±1 | 87±13 | 8.4±0.6 | 0.53±0.12 |
| **Autotrophs** | | | | | | | | | | |
| *Zostera capricornii* | Seagrass | SG | 10 | | 0.38±0.08 | 9.4±0.5 | 10.0±0.5 | 133±20 | 1.2±0.1 | 1.7±0.2 |
| *Enteromorpha* sp. | Green macroalgae | AL | 10 | | 0.34±0.09 | 3.3±0.3 | 0.68±0.03 | 53±4 | 4.4±0.5 | 1.3±0.1 |
| *Epiphytes* | | EP | 2 | | 1.7±0.4 | 3.1±0.3 | 3.8±0.6 | 55±6 | 2.2±0.1 | 7±5 |
| **Sources** | | | | | | | | | | |
| Zoo plankton | | ZP | | | 6.4 | 46 | 5.1 | 565 | 2.1 | 0.94 |
| Detritus | | DT | 10 | | 0.86±0.12 | 45±12 | 3±1 | 76±11 | 8±3 | 5.9±0.5 |

[a] Mean±standard error dry mass.

All samples were placed on ice and transported in this manner back to the laboratory.

#### 3.1.1. Categorisation of species

The structure of the estuarine food web was derived from an extensive review of previous ecological studies in Lake Macquarie, that have examined the gut contents, feeding strategies and habitat preferences of organisms residing in the Lake (Anderson, 1958; Bell, Burchmore, & Pollard, 1980; Bell & Pollard, 1989; Burchmore, Pollard, & Bell, 1984; Conacher, Lanzing, & Larkum, 1979; Grant, 1972; Kaiola et al., 1993; Pollard, 1984; Virgona, 1983). Species were divided into six trophic groups: autotrophs, planktivores, herbivores, detritivores, omnivores, carnivores, and sources as shown in Table 1. Detritus was included in the food web and categorised as a source. Zooplankton was also categorised as a source, as samples were comprised of several species exhibiting different feeding habits, for example, copepods being carnivorous, prawns being detritivores.

Most species could be classified with confidence into specific trophic groups. However, an issue in developing the food web, was the difficulty in classifying two fish species, *Girella tricuspidata* and *Mugil cephalus,* into trophic groups. This is because changes in diet occur within these species through different stages of their life cycle, and there is a high frequency of opportunistic omnivory observed within species inhabiting seagrass ecosystems (Livingston, 1982).

The fish species *G. tricuspidata* is known to undergo a change of diet from a carnivorous/omnivorous diet, becoming principally herbivorous when it grows to approximately 40 mm in length (*The ecology of fish in Botany Bay*, 1981). As all individuals sampled were $>100$ mm in length, *G. tricuspidata* has been classified as herbivorous.

The detrivore *M. cephalus* is also known to undergo a dietary shift upon reaching maturity, from carnivory to detritivory (Moriarty, 1976). As only mature *M. cephalus* were caught, these have been classified as detritivores.

### 3.2. Preparation of samples

Samples were stored at $-10$ °C prior to analysis. All samples were washed in deionised water, to remove any particulate matter that may be adsorbed to the sample. Organisms were then thawed at room temperature, weighed and size measured. For fish, total length was measured for species with rounded tails and fork length for species with forked tails. *P. palagicius* were measured across the carapace, prawns were measured from nose to tail, and data on the length, width and height of bivalves were taken.

### 3.3. Dissection of samples

Whole tissues were used for plants, detritus, zooplankton and polychaete analyses. Only fish, mollusc and crustacean muscle tissues were used for analysis, as this was considered to represent the stable pool of trace metals for these organisms.

All dissections were performed using stainless steel implements, on a cutting board covered in disposable plastic sheeting, which was replaced after each dissection. Muscle tissue was taken from the left-hand side of fish, slightly ventral to the dorsal fin, once scales and skin was removed. *P. palagicius* were shelled, and muscle tissue taken from their pincers. Prawns were also shelled, and their tails used. Molluscs were dissected, with their muscle tissues separated from organs. Plant tissues were washed in deionised water, and stripped of epiphytes using a stainless steel razor blade. Plant tissues and epiphytes were analysed separately. As some organisms such as zooplankton were too small to analyse individually, they were combined for analysis. Tissues were then weighed and placed in acid washed 30 ml plastic vials, covered with parafilm to prevent contamination from lids, and stored at −20 °C.

### 3.4. Digestion procedure

All tissues were freeze dried at −80 °C for 48 h (Edwards, Model No. EF2, Sussex, England). Stainless steel scissors cleaned with 70% ethanol were used to homogenise the samples into a fine powder. A low-volume microwave digestion procedure developed by Baldwin, Deaker, and Maher (1994) was used to digest samples. Approximately 0.07 g of freeze-dried material was weighed into a 7 ml polytetrafluroacetate (PFA) closed digestion vessel and 1 ml of concentrated nitric acid (Aristar, BDH, Australia) added. Each 7 ml PFA vessel was then capped and tightened to 2.3 Nm, and placed into larger 120 ml screw top pressure vessels, tightened to 16.3 Nm. A model MDS-81 (CEM, Indian Trail, NC, USA) microwave oven rated at 600 W was used for all digestions, with the microwave procedure consisting of three steps: 2 min at 600 W, 2 min at 0 W; and 45 min at 450 W. After digestion, the 7 ml PFA vessels were allowed to cool at room temperature for 20 mins, and then diluted to 10 ml with de-ionised water. Digests were stored in labelled polyethylene vials in a cool room (∼5 °C) until analysis.

### 3.5. Sample analysis

Samples were analysed for As, Se, Cd and Pb using a Perkin-Elmer 5100 PC atomic absorption spectrometer, equipped with Zeeman background correction, HGA-600 graphite furnace and AS-60 auto-sampler. Trace metal concentrations were determined using a mixture of palladium and magnesium as the matrix modifiers for As and Se (Deaker & Maher, 1995, 1997), and ammonium phosphate and magnesium nitrate for Cd and Pb.

Samples were analysed for Zn and Cu using a Perkin-Elmer 3100 Flame Atomic Absorption Spectrometer equipped with deuterium background correction, and a FIAS 2000 (Tinggi & Maher, 1986).

### 3.6. Quality assurance

Certified Reference materials NIST 1566a Oyster Tissue, NIES No. 9 Sargassum, CCRM DORM Dogfish Muscle No. 1, AGAL Tissue No. 3 Prawn, and AGAL

Tissue No. 2 Shark were used to determine the accuracy of data obtained (Table 2). Recoveries of trace metals were in agreement with certified values (Table 2).

### 3.7. Statistical analysis

The assumptions of normality and homogeneity of variances were checked qualitatively by examining a plot of residuals. As the data was found to be non-normal, and could not be transformed, non parametric equivalent Kruskal–Wallis tests were used to test the significance of differences between trophic groups and species. Statview® statistical package (SAS Institute USA) was used to perform non-parametric analyses.

Classification and ordination techniques were employed to examine groupings of species based on their relative trace metal concentrations. Classification involved the use of cluster analysis. The results of the cluster analysis were then plotted on a Multidimensional Scaling (MDS) ordination, to examine patterns.

## 4. Results

Mean concentrations of trace metals of each species studied are given in Table 1.

### 4.1. Selenium

#### 4.1.1. Trophic group concentrations

Highest mean selenium concentrations were observed in carnivores (6.9±0.4 μg/g), followed by planktivores (6.2±0.4 μg/g), the omnivore (4.1±0.7 μg/g), detritivores (4.0±0.3 μg/g), herbivores (2.8±0.2 μg/g), sources (1.4±0.5 μg/g), and autotrophs (0.49±0.10 μg/g) (Fig. 1). There was an increase in mean selenium concentration corresponding with increasing trophic level.

#### 4.1.2. Species concentrations

Selenium concentrations between species were significantly different ($H = 122.48$, D.F. = 17, $P < 0.0001$). The carnivorous fish species, *Platycephalus fuscus*, had the highest selenium concentrations (9.3±0.5 μg/g), followed by the carnivorous fish species, *Gerres subfasciatus* (9.2±0.7 μg/g) (Table 3). The lowest selenium concentrations were found in the two autotrophs *Z. capricornii* and *Enteromorpha* sp., with mean concentrations of 0.38±0.08 and 0.34±0.09 μg/g, respectively. The trend of selenium biomagnification between trophic groups was maintained in comparisons between species. Twenty-nine of the thirty-five trophic interactions examined demonstrated positive biotransference (Fig. 2).

### 4.2. Copper

#### 4.2.1. Trophic group concentrations

Mean copper concentrations were highest in sources (45±11 μg/g), followed closely by herbivores (44±12 μg/g) (Fig. 1). Detritivores also had elevated mean copper concentrations (39±6 μg/g), in comparison to autotrophs, planktivores, carnivores and the

Table 2
Recoveries of trace metals from certified reference materials

| Element | NIST SRM 1515 Apple leaves | | NIES Sargasso seaweed | | NIST SRM 1566a Oyster tissue | | AGAL-3 Prawn | | NRCC DOLT-1 Dog fish liver | | AGAL-2 Shark | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Measured | Certified | Measured | Certified | Measured | Certified | Measured | Certified | Measured | Certified | Measured | Certified |
| Se | 0.05±0.01 | 0.050±0.009 | 0.06±0.39 | 0.05 | 2.1±0.19 | 2.21±0.24 | 3.1±0.4 | 2.74±0.24 | 1.8±0.3 | 1.62±0.12 | 1.6±0.2 | 1.67±0.19 |
| Cu | 5.9±0.3 | 5.64±0.24 | 4.9±0.2 | 4.9±0.2 | 65±5 | 66.3±4.3 | 14±1 | 14.90±0.63 | 4.9±0.3 | 5.22±0.33 | 5.5±0.6 | 5.65±0.28 |
| Cd | 0.013±0.002 | 0.013±0.002 | 0.15±0.09 | 0.15±0.2 | 4.1±0.2 | 4.15±0.38 | 0.19±0.28 | 0.142±0.011 | 0.08±0.02 | 0.086±0.012 | 0.07±0.06 | 0.069±0.010 |
| Zn | 12±1 | 12.5±0.3 | 15±2 | 15.6±1.2 | 847±30 | 830±57 | 72±3 | 67±3 | 25±21 | 21.3±1.0 | 17±8 | 16.09±0.81 |
| As | 0.037±0.007 | 0.038±0.007 | 114±7 | 115±9 | 14±1 | 14.0±1.2 | 4±1 | 3.85±0.28 | 18.0±0.3 | 17.7±2.1 | 21.2±1.43 | 23.27±1.91 |
| Pb | 0.49±0.03 | 0.470±0.024 | 1.3±0.5 | 1.35±0.05 | 0.4±0.1 | 0.371±0.014 | 0.4±0.3 | 0.378±0.058 | 0.4±0.1 | 0.40±0.12 | 0.9±0.2 | 0.974±0.087 |

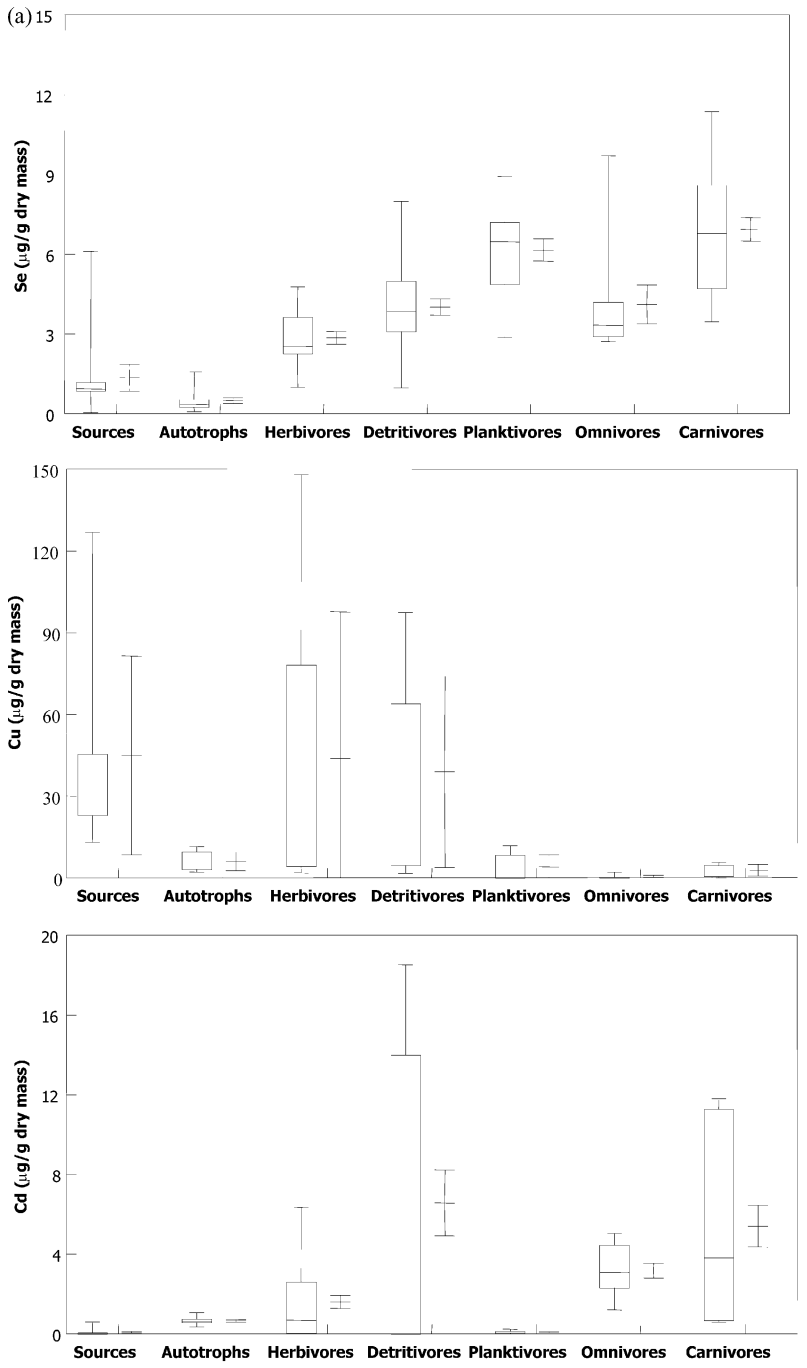


Fig. 1. Trace metal concentrations in Lake Macquarie seagrass ecosystem trophic groups. Box:- 25th and 75th percentile; whiskers:- 5th and 95th percentile. Error Bar (right of box):- mean and standard error.

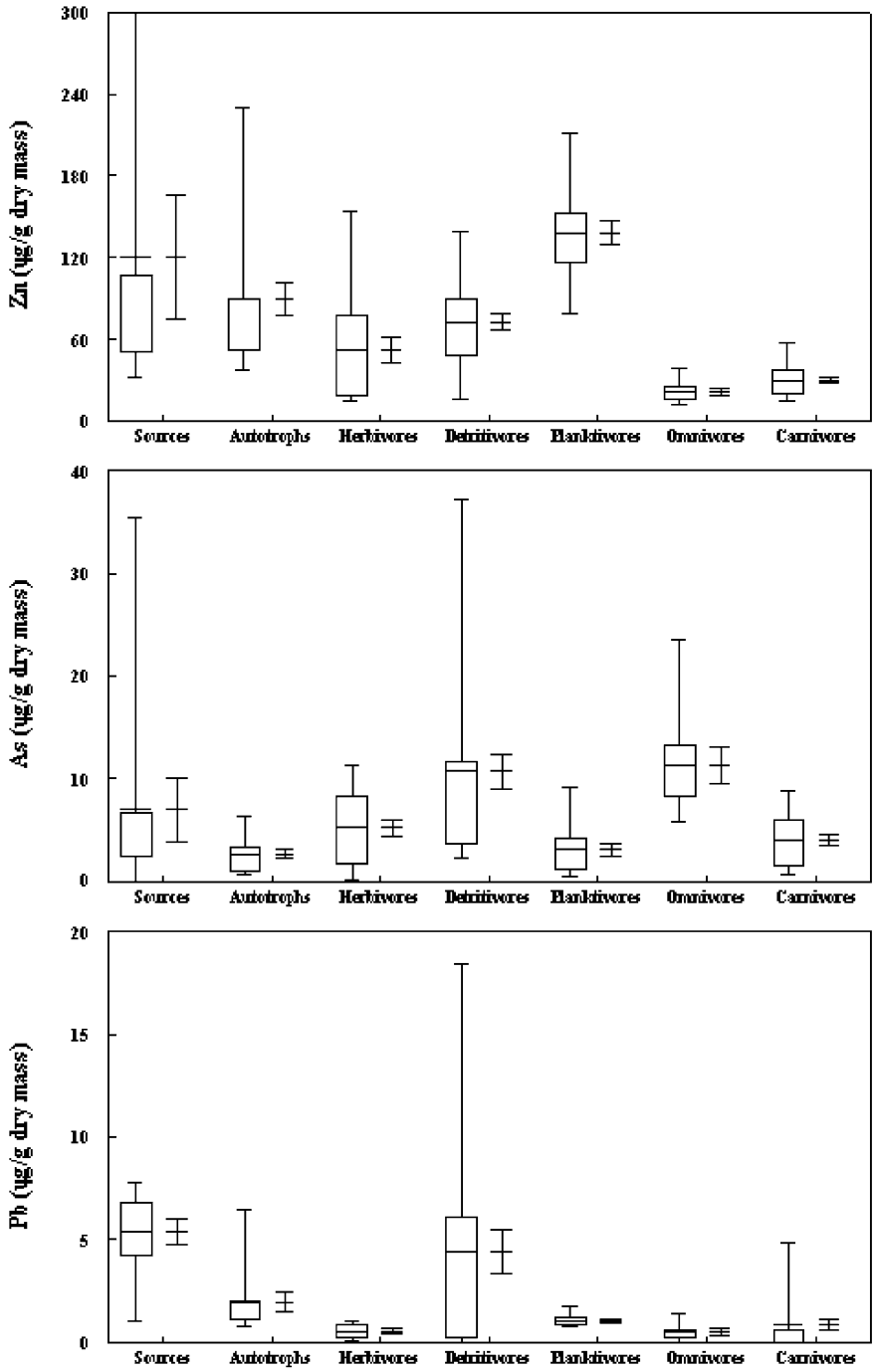


Fig. 1. (*continued*).

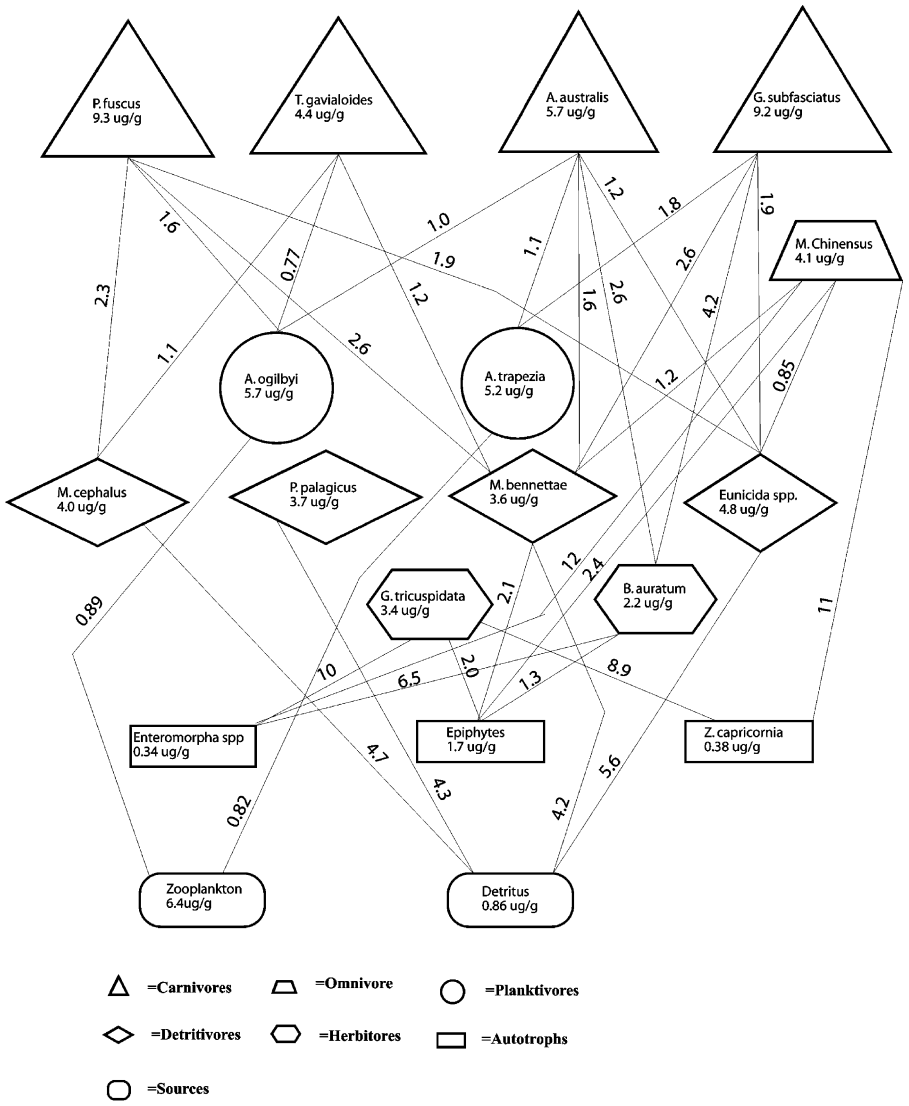


Fig. 2. Selenium concentrations in Lake Macquarie seagrass ecosystem species. Concentrations within symbols are mean concentrations and numbers on lines are biotransference factors.

omnivore (6.1±0.7, 4±1, 2.8±0.3 and 0.26±0.23 μg/g, respectively). Higher trophic groups generally had lower mean copper concentrations than lower trophic groups (Fig. 1).

### 4.2.2. Species concentrations

The herbivorous gastropod *B. auratum*, had the highest copper concentrations (88±15 μg/g) (Table 1). *M. bennettae* and *P. palagicus*, both detritivores, also had

high copper concentrations (77±7 and 63±7 μg/g, respectively). The lowest copper concentrations were found in the omnivore *Monacanthus chinensus* (0.27±0.23 μg/g). High mean copper concentrations observed within herbivores (Table 1), were due to high concentrations present within *B. auratum*. High mean copper concentrations observed within detritivores, were due to the high copper concentrations found in *M. bennettae* and *P. palagicus*.

The significant difference observed for copper concentrations between species ($H=139.18$, D.F = 17, $P<0.0001$) is due to the high copper concentrations of the herbivore *B. auratum,* and the detritivores *P. palagicus,* and *M. bennettae*, in comparison to other species (Table 1). High copper concentrations were found in crustaceans and molluscs. All fish species had low copper concentrations (Table 1).

Of thirty-five trophic interactions examined, eleven demonstrated positive copper biotransference from food source to consumer (Fig. 3). Increases in mean copper concentration between trophic groups are not systematic. Higher biotransference factors were generally observed between lower trophic groups. There was no evidence of biomagnification of copper.

### 4.3. Cadmium

#### 4.3.1. Trophic group concentrations

Mean cadmium concentrations were highest in herbivores (7±2 μg/g), followed by autotrophs (5±1 μg/g), sources (3.2±0.4 μg/g), and detritivores (1.6±0.3 μg/g) (Fig. 1). Carnivore, omnivore and planktivore cadmium concentrations were found to be below measurable limits. There was no discernible trend in mean cadmium concentrations between trophic groups.

#### 4.3.2. Species concentrations

*B. auratum* had the highest concentrations of cadmium (14±1 μg/g), followed by the autotroph *Z. capricornii* (10.4±0.5 μg/g) These two species had considerably higher cadmium concentrations than other species analysed, with zoo plankton having the next highest concentration (5.1 μg/g). The high mean cadmium concentrations observed in herbivores and autotrophs (Table 1), can be attributed to concentrations observed within *B. auratum* and *Z. capricornii*, respectively. The significant difference in cadmium concentrations between species ($F=143.01$, D.F = 17, $P<0.0001$) is due to the high concentrations measured in *B. auratum* and *Z. capricornii*. The carnivorous and herbivorous fish species, *G. subfasciatus* and *G. tricuspidata,* had the lowest cadmium concentrations, with concentrations below 0.001μg/g (detection limit). Lower cadmium concentrations were observed within species of higher trophic groups (Table 1).

Increases in cadmium concentration between species occurred in five of the 35 trophic interactions examined (Fig. 4). Increases in mean cadmium concentration between food sources and consumers appeared to be non-systematic. There were no trends evident in the magnitude of biotransference factors between lower and higher trophic groups. There was no evidence of biomagnification of cadmium.

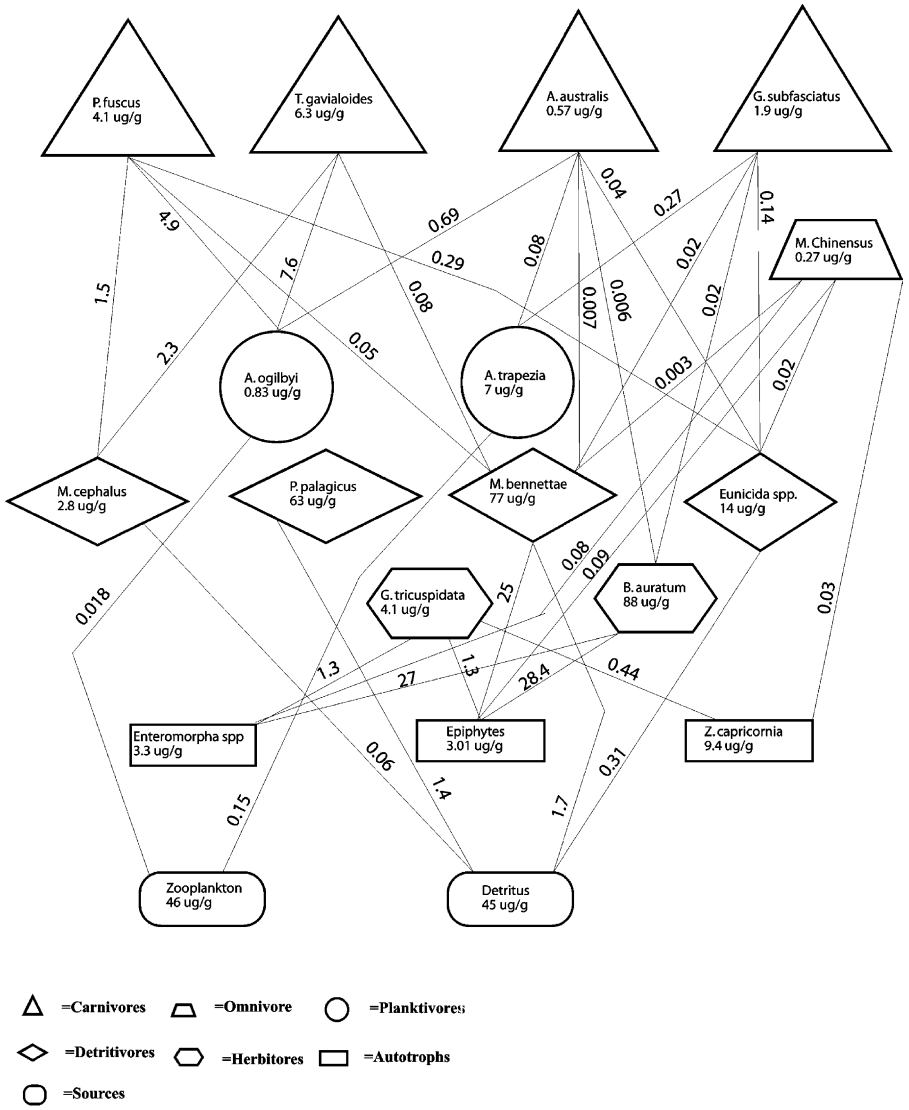


Fig. 3. Copper concentrations in Lake Macquarie seagrass ecosystem species. Concentrations within symbols are mean concentrations and numbers on lines are biotransference factors.

## 4.4. Zinc

### 4.4.1. Trophic group concentrations

Mean zinc concentrations were highest within planktivores (140±9 μg/g), followed by sources (120±46 μg/g). The high mean zinc concentrations measured within sources were attributed to zooplankton concentrations (565 μg/g) (Table 1). Autotrophs also have high zinc concentrations (80±13 μg.g), followed by detritivores (73±6 μg/g),

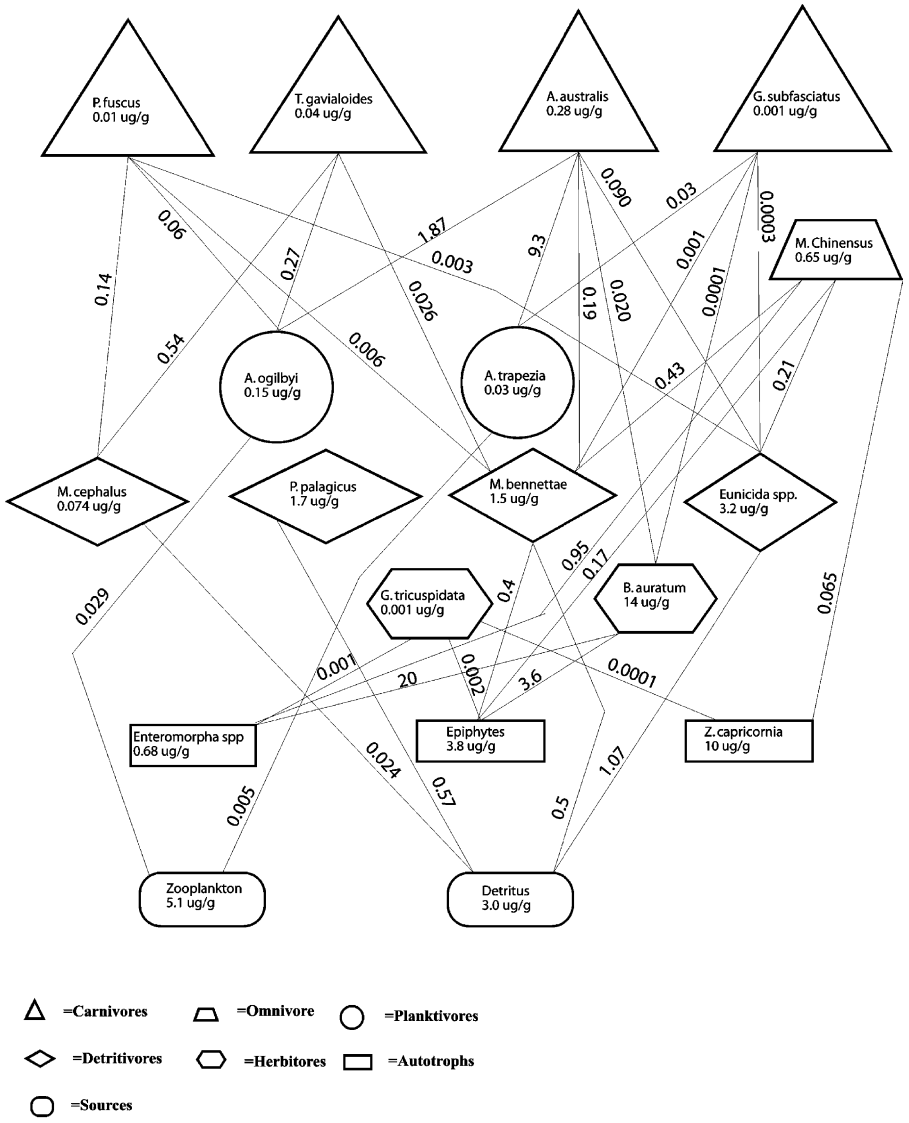


Fig. 4. Cadmium concentrations in Lake Macquarie seagrass ecosystem species. Concentrations within symbols are mean concentrations and numbers on lines are biotransference factors.

herbivores (54 ± 10 μg/g), carnivores (30 ± 2 μg/g), and the omnivore (21 ± 3 μg/g). There was no discernible trend in mean zinc concentrations between trophic groups (Fig. 1).

### 4.4.2. Species concentrations

Zooplankton had a zinc concentration of 565 μg/g, which was four times higher than that found in other organisms. *A. obilbyi* (151 ± 8 μg/g), *Z. capricornii* (133 ± 20 μg/g) and *A. trapezia* (126 ± 14 μg/g) also had high zinc concentrations. The lowest zinc

concentration was found in the carnivore, *Platycephalus fuscus* (19±1 μg/g), and the herbivore, *G. tricuspidata* (20±1 μg/g). All fish species had low zinc concentrations (Table 1). The significant difference in zinc concentrations between species ($H = 133.39$, D.F = 17, $P < 0.0001$ ) is due to the high zinc concentrations of zooplankton (565 μg/g).

Positive biotransference from food sources to consumers occurred in eight of the 35 trophic interactions examined (Fig. 5). All increases in mean zinc concentration

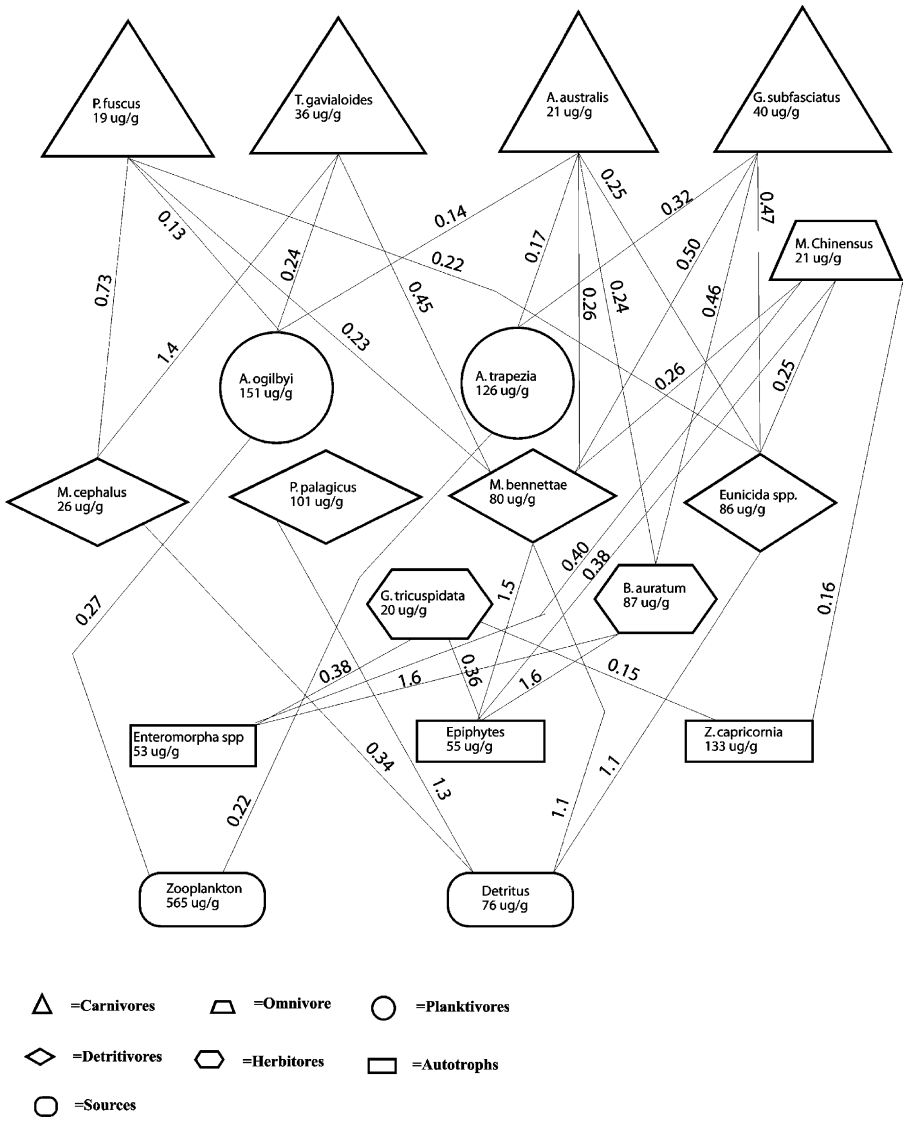


Fig. 5. Zinc concentrations in Lake Macquarie seagrass ecosystem species. Concentrations within symbols are mean concentrations and numbers on lines are biotransference factors.

occurred between lower trophic groups, with the exception of *M. cephalus* and *T. gavialoides* (Fig. 1). There were no trends evident in the magnitude of biotransference factors between lower and higher trophic groups. There was no evidence of biomagnification of zinc.

### 4.5. Arsenic

#### 4.5.1. Trophic group concentrations

The omnivore and detritivores had the highest mean arsenic concentrations (both $11 \pm 2$ μg/g) (Fig. 1). All other trophic groups had lower mean arsenic concentrations, with sources, herbivores, carnivores, planktivores, and autotrophs having concentrations of $7 \pm 3$, $5.2 \pm 0.8$, $4.1 \pm 0.5$, $3.1 \pm 0.6$, and $2.7 \pm 0.4$ μg/g, respectively. There was no discernible trend of mean arsenic concentration between trophic groups.

#### 4.5.2. Species concentrations

The detritivorous crustacean *P. palagicus*, had the highest arsenic concentration ($27 \pm 4$ μg/g), followed by the omnivore, *M. chinensus* ($11 \pm 2$ μg/g) (Table 1). The lowest arsenic concentrations were found in *Zostera capricornii* and *Playcephalus fuscus* ($1.2 \pm 0.1$ and $1.2 \pm 0.2$ μg/g, respectively). The high mean arsenic concentration of detritivores, can be explained by the arsenic concentrations measured in *P. palagicus*. The significant difference observed between species for arsenic concentrations ($F = 117.19$, D.F = 17, $P < 0.0001$), is attributed to the high arsenic concentrations present in the detritivore *P. palagicus* (Table 1).

Increases in arsenic concentrations from food sources to consumers, were observed in 19 of 35 trophic interactions (Fig. 6). There were no consistent pattern of biotransference factors between lower and higher trophic groups (Fig. 6). Of the four carnivorous fish species studied, those with diets mainly consisting of small fish (*P. fuscus* and *T. gavialoides*) had arsenic concentrations closely resembling that of their diet. Similarly, carnivorous species, which mainly consumed invertebrates (*A. australis* and *G. subfasciatus*), had arsenic concentrations closely resembling their prey. Four food links, comprising of three trophic groups, had positive biotransfernce throughout their length, indicating possible biomagnification. They were:

1. Zooplankton ($2.1$ μg/g), to *A. trapezia* ($4.8 \pm 0.9$ μg/g), to *A. australis* ($6.2 \pm 0.4$ μg/g).
2. Zooplankton ($2.1$ μg/g), to *A. trapezia* ($4.8 \pm 0.9$ μg/g), to *G. subfasciatus* ($6 \pm 1$ μg/g).
3. Detritus ($8 \pm 3$ μg/g), to *M. bennettae* ($8 \pm 1$ μg/g), to *M. chinensus* ($11 \pm 2$ μg/g).
4. Epiphytes ($2.2 \pm 0.4$ μg/g), to *M. bennettae* ($8 \pm 1$ μg/g), to *M. chinensus* ($11 \pm 2$ μg/g).

### 4.6. Lead

#### 4.6.1. Trophic group concentrations

Sources had the highest mean lead concentrations ($5.4 \pm 0.6$ μg/g), followed by detritivores ($4 \pm 1$ μg/g), and autotrophs ($2.0 \pm 0.5$ μg/g) (Fig. 1). All other trophic groups had considerably lower lead concentrations, with carnivores, planktivores, herbivores, and

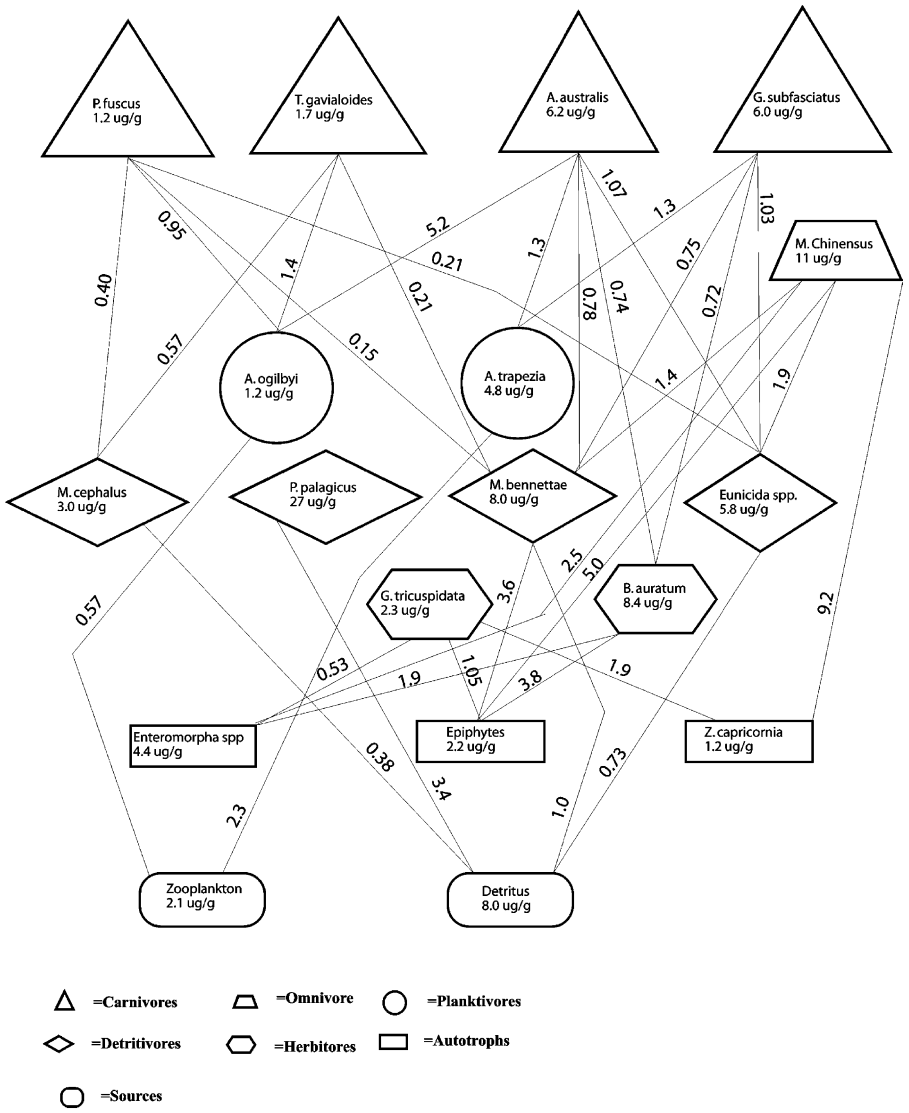


Fig. 6. Arsenic concentrations in Lake Macquarie seagrass ecosystem species. Concentrations within symbols are mean concentrations and numbers on lines are biotransference factors.

the omnivore having lead concentrations of $0.9\pm0.3$, $0.7\pm0.1$, $0.62\pm0.14$, and $0.42\pm0.14$ μg/g, respectively. Highest mean lead concentrations were observed within lower trophic groups.

### 4.6.2. Species concentrations

The Polychaete *Eunicida* sp. had the highest lead concentrations ($13\pm2$ μg/g), followed by epiphytes ($7\pm5$ μg/g) (Table 1). The lowest lead concentrations

were measured in *M. cephalus* and *Thylosurus gavialoides* ($0.10 \pm 0.05$ and $0.48 \pm 0.29$ μg/g, respectively). High mean lead concentrations observed in sources (Table 1), were largely influenced by detritus concentrations. High mean lead concentrations within detritivores (Fig. 1), can similarly be attributed to high mean concentrations measured in the *Eunicida* sp. (Table 1). The significant difference observed between species for lead concentrations ($F = 98.91$, D.F = 17, $P < 0.0001$) is attributed to the lead concentrations of the *Eunicida* sp.

Positive biotransference of lead was observed in nine of the 35 trophic interactions examined (Fig. 7). There were no trends evident in the magnitude of biotransference factors between lower and higher trophic groups (Fig. 7). Only one food link, comprising three trophic groups, had positive biotransfernce throughout its length, indicating biomagnification: Zooplankton (0.94 μg/g), to *A. trapezia* ($1.1 \pm 0.1$ μg/g), to *A. australis* ($1.5 \pm 0.6$ μg/g), but this was marginal. Therefore there was no evidence of biomagnification of lead.

### 4.7. Multivariate analysis

The MDS shown in Fig. 8 displayed a stress value of 0.09, indicating metal concentrations between individuals of the same species were similar. Species classification (Fig. 9) and MDS ordination (Fig. 8) revealed two primary groups, indicating metal concentrations within fish species (group B), were different from those within all other species examined (group A). The divergence of these two groups represents an interaction in ordinal space. This interaction indicates that metal concentrations within fish, tend to be similar irrespective of species, and are notably different from metal concentrations within invertebrates, plants and sources, with the exception of the bivalve *Anadara trapezia* (Fig. 8).

Species within trophic groups: carnivores (group I), omnivore (group II), planktivores (group III), autotrophs (group VI), and sources (group VII) grouped distinctly; indicating species of the same trophic group shared similar metal concentrations (Fig. 9). Autotroph species (*Z. capricornii, Enteromorpha* sp., epiphytes), were shown to group distinctly from other species at greater that 80% similarity, indicating they shared similar metal concentrations. Detritivorous invertebrate species (*M. bennettae, Eunicida* spp., *P. palagicus*), also separated distinctly from other species, with the exception of *B. auratum* at 80% similarity, indicating invertebrate detritivores and the herbivore *B. auratum* were generally similar in metal concentration. Invertebrate detritivores were further shown to have similar metal concentrations to their food source. The detritivorous fish species *M. cephalus*, was shown to be dissimilar in mean metal concentration to other detritivores, and detritus, and was shown to group more closely to carnivores. There appeared to be large differences in mean metal concentration between the two herbivorous species (group V), with herbivorous fish species *G. tricuspidata* grouping more closely to carnivorous species than with the herbivore *B. auratum*.

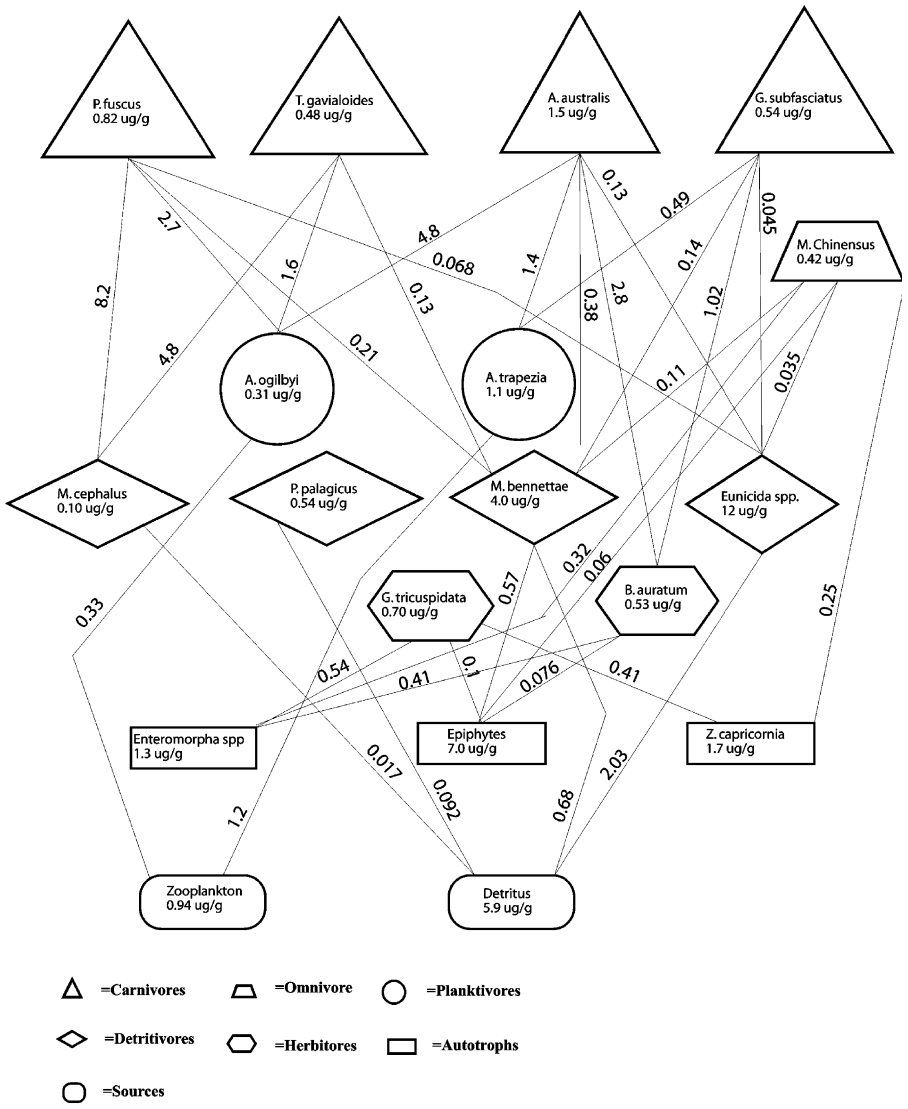


Fig. 7. Lead concentrations in Lake Macquarie seagrass ecosystem species. Concentrations within symbols are mean concentrations and numbers on lines are biotransference factors.

## 5. Discussion

### 5.1. General considerations

With the exception of fish species, *G. tricuspidata* and *M. cephalus*, species of the same trophic level were shown through multivariate analysis to group distinctly based on their measured metal concentrations (Fig. 9). It is thought that

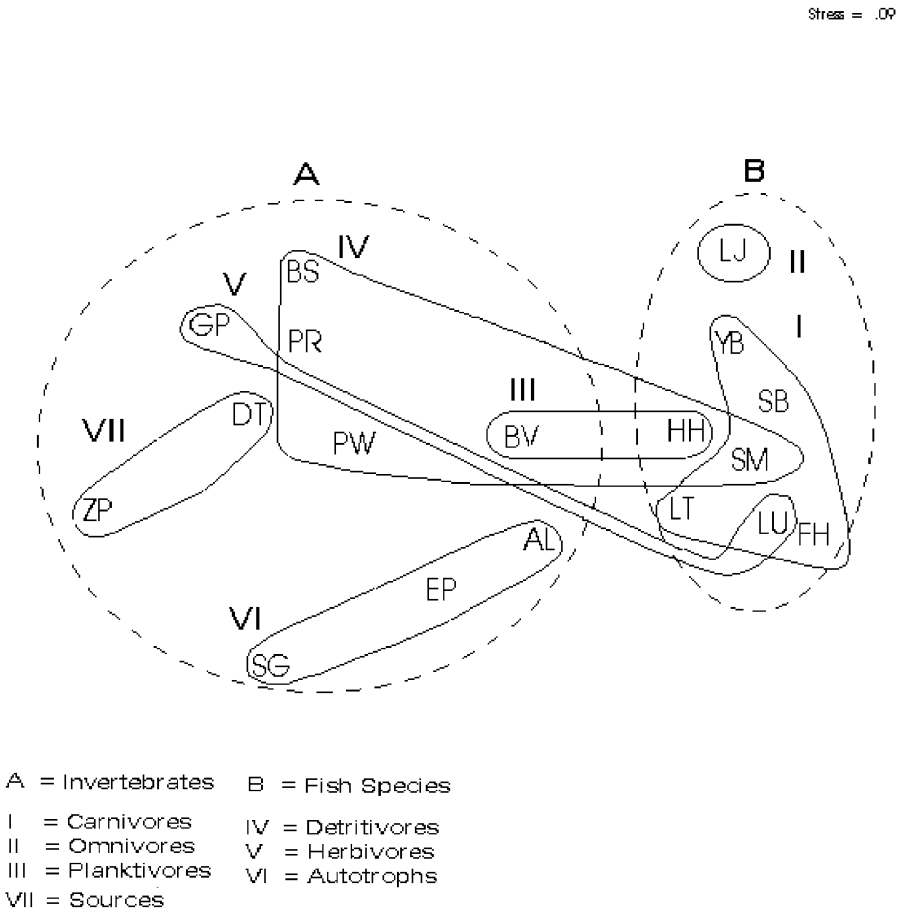


Fig. 8. MDS ordination showing grouping of Lake Macquarie seagrass ecosystem species based on mean concentrations of Cu, Cd, As, Pb, Zn and Se. See table for abbreviations.

*G. tricuspidata* and *M. cephalus* (classified as a herbivore and detritivore, respectively) grouped more closely to the carnivorous species examined, because of a diet shift undertaken by these species upon reaching maturity, i.e. from a principally carnivorous/omnivorous diet, to their classified trophic group (*The ecology of fish in Botany Bay*, 1981; Moriarty, 1976). It is thus possible that individuals of these species analysed, either still maintained omnivorous diets, or reflected metal concentrations accumulated during their juvenile life-stages. Similar metal concentrations were also observed within all fish species examined. Consequently, it is also possible that some physiological or behavioural factors are responsible for similar metal concentrations within fish species, and not food source (Bernhard & Andreae, 1984; Sorensen, 1991). This illustrates the need to measure carbon/nitrogen isotopic ratios, to confirm the proportion various food sources contribute to the overall diet of the species examined.

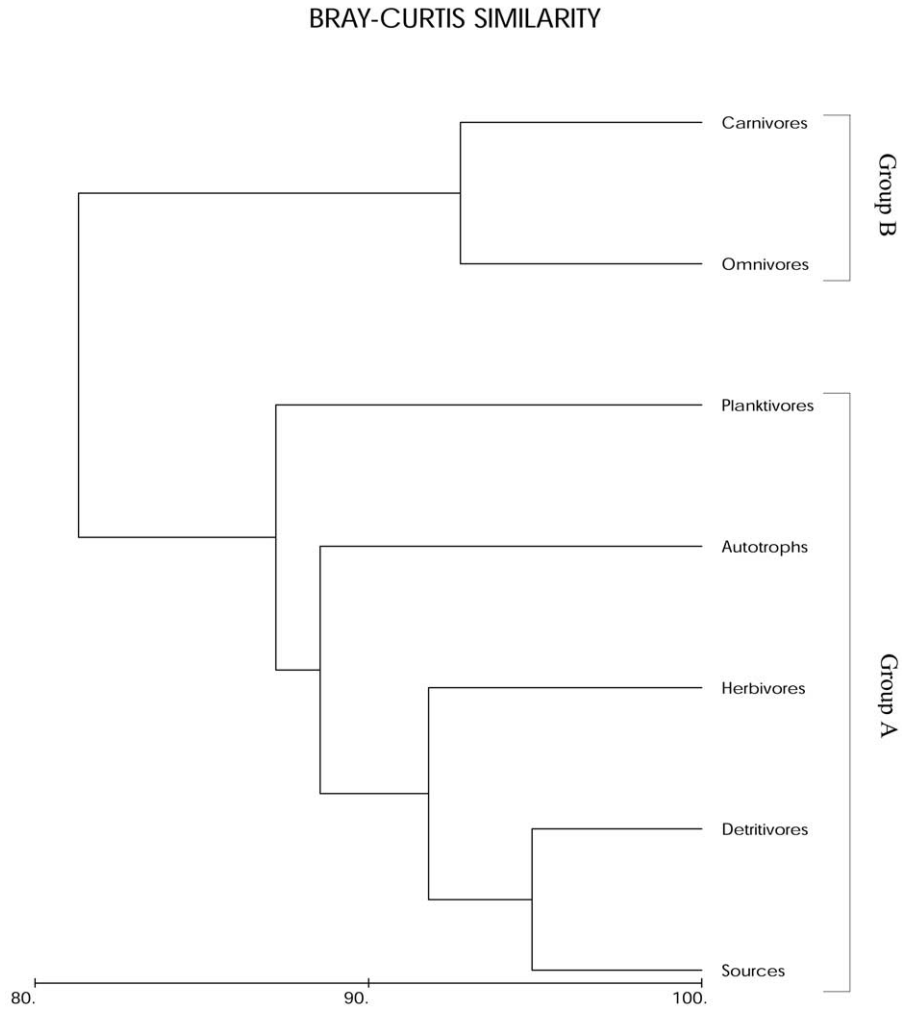


Fig. 9. Hierarchical agglomerative classification of Lake Macquarie seagrass ecosystem species, based on similarities between Cu, Cd, As, Pb, Zn and Se concentrations.

The lack of aqueous point source discharges, low trace metal concentrations in the water column and trace metal accumulation in sediments, indicates trophic transfer is the likely route of metal accumulation. Laboratory experiments have confirmed that Lake Macquarie benthic species take up selenium from contaminated sediments (Peters et al., 1999). Pulse-chase and other experiments, also confirm that marine animals tend to retain trace metals obtained through food sources (Chen & Mayer, 1999; Griscom, Fisher, & Luoma, 2002; Rainbow & Wang, 2001; Wang, Fisher, & Luoma, 1996; Wang, Qui, & Qian, 1999; Wang & Rainbow, 2000; Wang, Stupakoff, & Fisher, 1999; Xu & Wang, 2002). However, controlled experiments partitioning

trace metals from different uptake routes need to be undertaken to confirm trophic transfer as the main transfer route.

In the discussion of each trace metal below, comparisons are made to published data of trace metals in marine organisms from uncontaminated Australian environments (e.g. Maher, Baldwin, Deaker, & Irving 1992). These organisms, which include many of those sampled in this study, are from a range of uncontaminated sites, many with similar geology to Lake Macquarie, and span a number of seasons. This in effect is similar to sampling at a comparison site, and indicates trace metal contamination in Lake Macquarie. Selecting true comparison sites (controls) was not possible, as there are no comparable temperate Australian seagrass systems with all these fish, crustacean and mollusc species.

### 5.2. Selenium

Selenium concentrations measured in organisms (Table 1) are similar to those reported in previous studies of Lake Macquarie (Batley, 1987; Kirby, Maher, & Harasti, 2001; Kirby, Maher, & Krikowa, 2001; Peters et al., 1999; Roberts, 1994; Wlodarczkyk & Beath, 1997). Selenium concentrations were elevated compared to selenium concentrations reported for organisms from uncontaminated environments (0.5–4 μg/g dry mass) (Bebbington et al., 1977; Maher et al., 1992). A number of species consumed by humans (*P. fuscus, A. australis*, *G. subfasciatus*. *A. trapezia*), were found to have selenium concentrations above maximum permitted concentrations for human consumption (5 μg/g dry mass, Australia New Zealand Food Authority (ANFA), 1992). The carnivorous species *P. fuscus* and *G. subfasciatus* had mean selenium concentrations approximately two times the maximum permitted concentrations (Table 1).

These findings are consistent with previous studies (Batley, 1987; Kirby, Maher, & Harasti, 2001; Kirby, Maher, & Krikowa, 2001; Peters et al., 1999), which reported selenium contamination within pore water, sediments and biota of southern Lake Macquarie attributed to fly ash from the coal-fired power station at Vales Point. Selenium present within sediments ($6\pm3$ μg/g dry mass) has been shown to bioaccumulate within benthic infauna, through to fish via trophic transfer (Peters et al., 1999). These findings are supported by other investigations (Hermanutz, Allen, Roush, & Hedtke, 1992; Liu et al., 1987; Rainbow & Wang, 2001; Wang et al., 1996; Wang, Qui, & Qian, 1999; Wang, Stupakoff, & Fisher, 1999; Xu & Wang, 2002; Zhang, Hu, & Huang, 1990), which have demonstrated that dietary uptake is the predominant mode of selenium accumulation by organisms. Selenium concentrations in overlying waters are low (0.3–0.5 μg/l). It is, therefore, likely that selenium concentrations measured in organisms examined in this study, are the result of dietary uptake of selenium through benthic food chains.

Dietary selenium has been described as more harmful than waterborne selenium, with diet concentrations of 10 μg/g eliciting toxic responses in freshwater fish (Hilton & Hodson, 1983; Hilton, Hodson, & Slinger, 1980). No ill effects were observed in fish exposed to waterborne selenium of the same concentrations (Hilton et al., 1980). The selenium concentrations measured in organisms in this study, are above

those reported to impair the growth and survival of freshwater organisms (Bennett, Brooks, & Boraas, 1986; Bertram & Brooks, 1986; Coughlan & Velte, 1989; Hilton & Hodson, 1983; Hilton et al., 1980; Hilton, Hodson, & Slinger, 1982; Ogle & Knight, 1989). As these reports pertain to freshwater organisms, any inferences must be made with caution. High selenium concentrations present within species examined, coupled with the suspected large proportion of dietary selenium within species examined, are a cause for concern.

Selenium concentrations increase up food chains (Fig. 1). Positive biotransference of selenium concentration from source to consumer were observed for 28 of the 35 trophic interactions (Fig. 2). Furthermore, 21 of 26 multi-trophic level food chains examined displayed positive biotransference of selenium between each trophic level (e.g. detritus, to *M. cephalus,* to *P. fuscus*). Thus selenium biomagnification occurred within the food web examined. These findings give credence to those of Biddinger and Gloss (1984), who reported selenium biomagnification within an aquatic ecosystem.

Selenium is an essential element, found in low concentrations within natural aquatic ecosystems (Hodson, 1990). As a result, selenium is taken up by organisms through active transport mechanisms (Zhang et al., 1990). Though excretory processes are also known for selenium, these can be overwhelmed in contaminated environments by high selenium loading, resulting in net accumulation of selenium (Zhang et al., 1990). When an organism is consumed, it is thought that the selenium present within the organism is transferred to its consumer and accumulated through these active transport mechanisms, resulting in biomagnification. Clearly the threshold point at which excretory processes are overloaded is most likely to be reached within contaminated environments such as Lake Macquarie.

### 5.3. Copper

Copper concentrations measured in organisms (Table 1) are similar to those reported in previous studies of Lake Macquarie (Furner, 1979; Kirby, Maher, & Harasti, 2001; Kirby, Maher, & Krikowa, 2001; Roberts, 1994; Taylor, 1998). The invertebrates *B. auratum, M. bennettae* and *P. palagicus* had the highest copper concentrations (Fig. 1b). Furthermore, *M. bennettae* and *P. palagicus,* were the only species examined with copper concentrations exceeding National Food Authority maximum permitted concentrations for human consumption [50 μg/g dry mass for all seafood except molluscs (350 μg/g dry mass), ANFA, 1992].

High copper concentrations found within *M. bennettae, P. palagicus* and *B. auratum,* probably reflect active accumulation of copper by these species for incorporation into the respiratory pigment haemocyanin, a copper-based pigment found in the blood of many species of molluscs and crustacea (Clarke, 1986). The bivalve *A. trapezia* were found to contain comparatively lower copper concentrations to other molluscs and crustacea (Table 1). Unlike most molluscs and crustacea, *A. trapezia* possesses an iron-based respiratory pigment (Bonventura & Bonventura, 1983; Ghiretti & Ghiretti-Magaldi, 1972), consistent with the lower copper concentrations measured in this species relative to other molluscs and crustaceans.

All fish species had comparatively lower mean copper concentrations than invertebrate species (Table 1). Furthermore, copper concentrations in fish species were similar to concentrations reported in fish from uncontaminated sites (0.2–4.75 μg/g dry mass, Brooks & Rumsey, 1974; Denton & Burdon-Jones, 1986). Copper concentrations within muscle tissues of all fish species, were also below those that have been shown to cause a reduction in survivorship of the freshwater fish *Noemacheilus barbatulus* (9.1 μg/g dry mass) (Solbe & Cooper, 1976), when exposed to copper through dietary sources. Copper concentrations may be regulated in fish due to the essential nature of this metal for metabolic processes (Denton & Burdon-Jones, 1986). Copper concentrations are slightly elevated in sediments (17–38 μg/g dry mass; Kirby, Maher, & Krikowa, 2001), but low enough for organisms to maintain self-regulation of this metal.

Positive biotransference of copper concentration from food source to consumer were observed for 11 trophic interactions (Fig. 3). Biotransference factors were found to be higher between lower trophic groups in comparison to higher trophic interactions. Organisms within higher trophic groups, are able to regulate their internal metal concentrations more effectively (Bernhard & Andreae, 1984). Positive biotransference was only evident between two trophic groups (Fig. 3). As positive biotransference must be observed between two or more trophic levels to constitute biomagnification (Connell & Miller, 1984), it can be conclusively stated that copper did not biomagnify through the food web examined. This is consistent with other studies of copper in aquatic ecosystems (Amiard et al., 1980; Anderson, 1958; Metayer et al., 1980; Prahalad & Seenayya, 1986; Timmermans et al., 1989).

### 5.4. Cadmium

Cadmium concentrations measured in organisms (Table 1) are lower than those reported in previous studies of Lake Macquarie (Furner, 1979; Roberts, 1994; Taylor, 1998), with the exception of *A. australis, M. cephalus,* and *Z. capricornii*. Cadmium concentrations in fish were similar to those reported for uncontaminated locations (0.01–0.2 μg/g dry mass, Bebbington et al., 1977; Brooks & Rumsey, 1974). Low cadmium concentrations (1.1–2.3 μg/g) within southern Lake Macquarie sediments (Kirby, Maher, & Krikowa, 2001), indicate no significant sources of cadmium, thereby explaining similarities in cadmium concentrations of organisms to uncontaminated locations.

The gastropod mollusc *B. auratum* has relatively high cadmium concentrations ($14 \pm 1$ μg/g dry mass). Similar concentrations of cadmium have also been measured in *B. auratum* collected from other sites in Lake Macquarie (Taylor, 1998). This supports the findings of Engel and Fowler (1979), who described the active accumulation of cadmium by some molluscs. Conversely, the bivalve *A. trapezia* were found to have low cadmium concentrations (Table 1). *A. trapezia* has been shown to accumulate trace metals from overlying water, rather than from sediments and associated biota (Scanes, 1993). Cadmium concentrations are low within the water column ($< 1$ μg/l), and may subsequently explain the low cadmium concentrations within this species.

Current maximum permitted trace metal concentrations for human consumption do not specify a maximum cadmium concentration for any food types (ANFA,

1992). Under proposed changes by the Australian New Zealand Food Authority, a cadmium concentration for molluscs (2 μg/g dry mass) will be specified, however, no mention is made of other seafoods. None of the species analysed in this study, that are collected for human consumption, were found to contain cadmium concentrations above the proposed maximum permitted concentrations (ANZFA, 1999), indicating that, in terms of cadmium concentration, consumption of organisms from southern Lake Macquarie would not pose a threat to human health.

Cadmium concentrations are also at levels in tissues below those known to affect the growth, reproduction and survival of marine organisms (Canli & Furness, 1995; Jennings & Rainbow, 1979; Olla, Estelle, Swartz, Braun, & Studholme, 1988).

Positive cadmium biotransference occurred in five of the thirty-five trophic interactions examined (Fig. 4). Similar to copper, biotransference factors were greater between lower trophic groups (e.g. autotrophs to herbivores) than between higher trophic groups (e.g. herbivores to carnivores). As a result, positive biotransference was only evident between lower trophic groups of the food chains examined. It can therefore be conclusively stated that cadmium did not biomagnify through the food web examined. These findings are consistent with previous investigations (Amiard et al., 1980; Bargagli, 1993; Bernhard & Andreae, 1984; Metayer et al., 1980; Prahalad & Seenayya, 1986; Ward et al., 1986), which reported variable biotransference of cadmium between lower trophic levels, but found no evidence of biomagnification.

### 5.5. Zinc

Zinc concentrations measured in invertebrate species (Table 1), are similar to those reported in previous studies of Lake Macquarie (Batley, 1987; Furner, 1979; Kirby, Maher, & Krikowa, 2001; Taylor, 1998). Further, zinc concentrations in all fish species are similar to those reported for uncontaminated locations (2.5–180 μg/g dry mass) (Bebbington et al., 1977; Brooks & Rumsey, 1974; Denton & Burdon-Jones, 1986). Zinc concentrations (112–188 μg/g dry mass) within southern Lake Macquarie sediments are elevated relative to background concentrations (50–60 μg/g dry mass) but do not indicate major zinc contamination, thereby explaining similarities in zinc concentrations of organisms to those of uncontaminated sites.

*A. ogilbyi,* had the highest zinc concentrations of all fish species examined ($151 \pm 8$μg/g dry mass) (Table 1). It is probable that the high zinc concentrations found in *A. ogilbyi* are accumulated from its food source (zooplankton), which had the highest zinc concentrations measured (565 μg/g). Active zinc regulation by fish has been suggested in the literature (Denton & Burdon-Jones, 1986), however, regulation was suggested to be incomplete for this metal. It is possible that the dietary uptake of zinc by *A. ogilbyi* exceeded regulatory threshold concentrations, resulting in elevated concentrations of zinc. Only zooplankton was found to have zinc concentrations above current maximum permitted concentrations for human consumption (150 μg/g dry mass) (ANFA, 1992), thus in terms of zinc, consumption of organisms from Lake Macquarie poses no threat to humans.

As zinc concentrations in organisms are similar to those reported for uncontaminated areas, it can be assumed that zinc concentrations are below those likely to cause sub-lethal effects and toxicity to aquatic organisms.

The findings of the present study indicate positive zinc biotransference within eight of the 35 trophic interactions examined (Fig. 4). Similar to copper and cadmium, biotransference factors were higher between lower trophic groups (e.g. autotrophs to detritivores), in comparison to higher trophic interactions (e.g. planktivores to carnivores). This is probably due to internal control of zinc as described for copper (Bernhard & Andreae, 1984).

Positive biotransference was mainly observed between lower trophic groups, and only between two trophic levels within food chains (Fig. 5). It can therefore be concluded that zinc did not biomagnify through the food web examined. These findings are consistent with previous investigations (Metayer et al., 1980; Ward et al., 1986). However, these findings differ to those of Amiard et al. (1980) who reported that zinc biomagnification in an estuarine food chain may occur and Timmermans et al. (1989), who reported biomagnification of zinc within a littoral food web. The food web studied by Timmermans et al. (1989) was comprised entirely of invertebrates. Often invertebrates lack regulatory, excretory and detoxification mechanisms of higher order organisms such as vertebrates (Bernhard & Andreae, 1984).

### 5.6. *Arsenic*

Arsenic concentrations measured in invertebrates and autotrophs (Table 1) are lower than those previously reported for Lake Macquarie (Batley, 1987). Arsenic concentrations within fish species were also similar to those measured in fish from uncontaminated locations (0.5–22 μg/g dry mass) (Bebbington et al., 1977; Maher & Batley, 1990).

Highest arsenic concentrations were found in the crab *P. palagicus* (Table 1). This is consistent with previous studies (Maher & Batley, 1990), which have reported relatively high arsenic concentrations in crab species from uncontaminated areas. Reasons for elevated arsenic concentrations within crabs in comparison to other crustacea are unknown.

Several species consumed by humans (*A. australis*, *G. subfasciatus*, *M. chinensus*, *P. palagicus*, *M. bennettae*) have high arsenic concentrations (Table 1) The maximum permitted concentrations for human consumption is 5μg/g dry mass as inorganic arsenic (ANFA, 1992). All organisms contained $<5$ug/g dry mass as inorganic arsenic (Price, 2001) and thus do not pose a threat to human health. Arsenic accumulated by marine organisms via their diet, has never been shown to be toxic, as most is transferred as arsenobetaine which is not toxic (Maher & Butler, 1988).

Positive biotransference of arsenic occurred for 19 of the 35 trophic interactions examined (Fig. 5). In addition, arsenic biotransference through three successive trophic levels was observed within four food chains examined (Fig. 5), indicating possible arsenic biomagnification. In these few cases positive biotransference factors were low.

On the basis of these findings it is concluded that positive arsenic biotransference was observed within the food web examined and possible biomagnification in some food chains. Previous investigations have reported no evidence of arsenic biomagnification (Bernhard & Andreae, 1984; Klumpp & Peterson, 1979; LeBlanc & Jackson, 1973; Wagemann, Snow, Rosenberg, & Lutz, 1978).

### 5.7. Lead

Lead concentrations measured in species (Table 1) are similar to those previously reported for Lake Macquarie (Batley, 1987; Furner, 1979). Additionally, lead concentrations were also within the ranges reported for organisms from uncontaminated locations (0.4–20.5 μg/g dry mass) (Bebbington et al., 1977; Brooks & Rumsey, 1974). All organisms consumed by humans have lead concentrations below maximum permitted lead concentrations for human consumption (2.5 μg/g dry mass) (ANFA, 1992), suggesting no potential threat from lead to human health.

On the basis of observed similarities in mean lead concentrations to those reported from unpolluted environments, it can be assumed that lead concentrations within organisms are below sub-lethal and toxic thresholds.

Comparatively low lead concentrations found within crustaceans and molluscs (Table 1), are thought to be due to the partitioning of lead to shells and exoskeleton, as lead has been shown to accumulate via the same processes as calcium (Sorenson, 1991). The polychaete *Eunicida* sp. had the highest lead concentrations ($12 \pm 2$ μg/g), although polychaetes have been shown to regulate lead (Phillips, 1991). As the gut of polychaetes are difficult to depurate, it is possible high lead concentrations observed within *Eunicida* spp. were caused by residual sediment particles present within the guts of this species.

Positive lead biotransference occurred for nine of the 35 trophic interactions examined (Fig. 6). Of 26 food chains examined, only one displayed positive lead biotransference through all trophic levels: zooplankton (0.94 μg/g), to *A. trapezia* ($1.1 \pm 0.1$ μg/g), to *A. australis* ($1.5 \pm 0.6$ μg/g). Biotransference factors between species within this food chain were low (Fig. 6). Standard errors for lead concentrations within *A. trapezia* and *A. australis* overlap (Table 1), indicating biotransference between these two species was an artefact of mean concentrations described for these species. On the basis of these findings it is concluded that although positive lead biotransference occurred within the food web examined between some species, biomagnification does not. This is consistent with the findings of previous studies (Amiard et al., 1980; Metayer et al., 1980; Mikac et al., 2001; Timmermans et al., 1989; Vighi, 1981; Ward et al., 1986), that reported no evidence of lead biomagnification within the food webs studied.

## 6. Conclusions

Selenium biomagnification was found in the contaminated seagrass ecosystem, resulting in selenium concentrations above maximum permitted concentrations for

human consumption in three of four carnivorous fish species examined, all of which are consumed by humans. Selenium concentrations were also above those that have been shown to cause adverse effects in aquatic organisms.

There was no evidence of biomagnification of copper, cadmium, zinc, or lead. Copper, cadmium, zinc and lead concentrations were below levels shown to elicit adverse responses in biota. On the basis of these findings, it is concluded that copper, cadmium, zinc and lead concentrations within the Lake Macquarie temperate seagrass ecosystem examined are not present at concentrations hazardous to biota.

Cadmium, zinc and lead concentrations were also below maximum permitted concentrations for human consumption. Copper concentrations within the crustaceans, *P. palagicus* and *M. bennettae,* were found to be above maximum permitted concentrations for human consumption. It is thought copper is accumulated due to the essential role of copper within many species of molluscs and crustacea. As these species support commercial fisheries within Lake Macquarie, these findings identify the need to examine sources of copper contamination within these species, and if present, develop appropriate management strategies to regulate these sources.

Arsenic showed some evidence of biomagnification. Total arsenic concentrations are similar to those found in other marine ecosystems and thus unlikely to cause adverse effects to aquatic organisms. Inorganic arsenic concentrations are below maximum permitted concentrations for human consumption.

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