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Population and community level indicator in assessment of heavy metal contamination in seagrass ecosystem

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Abstract — Seagrasses reflect the concentration of heavy metals present in the environment and are considered good biomonitors of contaminants. The aim of our study was to determine whether the level of pollution in seagrass areas influences seagrass morphology and abundances of associated epifauna. The following hypotheses were tested: 1, the concentration of heavy metals would be significantly higher in seagrass from the polluted location than in seagrass from relatively clean locations; 2, seagrass morphology (shoot density, leaf density, leaf length, and biomass) would be different between polluted and control locations; 3, abundance and diversity of epifauna associated with seagrass would be different between polluted and non-polluted locations. We measured shoot density, leaf density, leaf length and biomass of seagrass, *Zostera capricorni*, as well as abundance and diversity of associated epifauna. Concentrations of metals (Cd, Cu, Pb, Se, and Zn) in roots and leaves of *Z. capricorni* were significantly higher in samples from polluted compared to control locations. There were no significant difference in shoot and leaf density, leaf length, and biomass of the seagrass between polluted and control locations. Abundance of gastropods was significantly lower in polluted location than in controls, whereas other epifaunal groups displayed no difference. The results are discussed in view of selecting non-costly bioindicators of heavy metal contamination.

Key words: seagrass, *Zostera capricorni*, epifauna, heavy metal, bioindicator

Introduction

Several facts generally indicate that seagrasses provide to overall functioning of coastal zone systems (Hemminga and Duarte 2000). They are commonly perceived as important habitats because they act as a source of food and shelter as well as nursery grounds for numerous ecologically and commercially important species of fish and invertebrates (den Hartog 1977, Kikuchi and Peres 1977).

Seagrasses grow in sheltered coastal areas, which makes them especially vulnerable to human disturbance (Walker et al. 2001). In New South Wales of Australia, seagrasses occur primarily in the protected waters of estuaries and semi-enclosed embayments (West et al. 1989). The evidence that more than 80% of NSW population live near the coast and a high proportion of the State's commercial activity also occurs near estuaries means that estuaries including the seagrasses are subject to a range of direct and indirect impacts due to land use in the catchments and the direct use of estuarine waterways (Department of Land and Water Conservation 2000).

Lake Macquarie of New South Wales is one of the largest estuarine lagoons in Australia. This lake extends approximately 22 km in a north-south direction. It has a maximum width of 10 km and a maximum depth of 11 m, with an average depth of 8 m (Batley 1987, Peters et al. 1999). The total waterway area of Lake Macquarie is 120 km^2 and the catchment area is 700 km^2 (Manly Hydraulics Laboratory 2002). The lake is separated from the ocean by a narrow entrance channel resulting in poor tidal flushing. Despite this poor tidal exchange, the lake has a marine character because of minimal freshwater dilution input (Roy and Crawford 1984). Shallow in the middle part of the lake effectively prevents deepwater movement from the northern to southern part of the lake resulting in a division of the lake into northern and southern components. As such, water circulation in the two portions of the lake is essentially independent (Spencer 1959).

The sediments of Lake Macquarie are contaminated with trace metals such as cadmium, lead and zinc (Roy and Crawford 1984). Toxic heavy metals have been accumulated in Lake Macquarie since 1897 with the start of the operation of Pasminco Metals-Sulphide Smelter. This lead-zinc smelter (1.5 km north east of the lake) discharged heavy metals resulting in the contamination of Cockle Creek and the northern reaches of Lake Macquarie with copper, lead, zinc and cadmium (Environmental Resource Management 2000). The heavy metal concentrations reduce southward through the lake indicating that the lead-zinc smelter was in the past a major source of contaminants (Roy and Crawford 1984).

Zostera capricorni, the most widespread seagrass in this lake had notably declined with around 700 ha of beds disappearing over the past 20–25 years (King and Hodgson 1986). The toxic chemical pollution, primarily heavy metals, are significant environmental contaminants of seagrass systems (Ward 1987, Ward 1989). It is, however, unclear if the heavy metal pollution within the lake had a contribution to the loss of the seagrass.

Concentrations of cadmium, copper, lead, zinc, and selenium are much higher in sediments of Cockle Bay which is closest to the smelter and the metal concentrations tend to decrease to the southern end of the lake (Batley 1987, Peters et al. 1999). In addition, there was a significant accumulation of heavy metals in the tissues of seagrass, *Z. capricorni* in Cockle Bay (polluted location) (Ambo Rappe et al. 2007).

Heavy metals can be incorporated into seagrass leaves and vascular tissue from either water column or sediment. The presence of the heavy metals has been demonstrated to inhibit the growth of seagrass (Ward 1989) and to affect the structural characteristics of leaves and shoots (Conroy et al. 1991). Toxic concentrations of metals specifically inhibit metabolic activity and interfere with vital biochemical pathway, for example photosynthesis (Ralph and Burchett 1998, Macinnis-Ng and Ralph 2002).

Study of epibenthic seagrass fauna showed that species richness of seagrass fauna was decreased in the highly contaminated site (Ward and Young 1982). Moreover, field acute toxicity experiments indicated that an amphipod crustacean commonly found in seagrasses, *Cymodocea longicaudata* was strongly affected by the metals. This species displayed an extreme sensitivity to the effluent from a smelter with more than 80% dying within the first 7 days of the experiment (Ward 1984).

The main aim of this study was to determine whether there was any detectable effect of pollution on general morphology (shoot density, leaf density, leaf length, and biomass) of *Z. capricorni* in Lake Macquarie and on associated epifaunal assemblages.

The following hypotheses were tested:

1. Shoot density, leaf density, leaf length, and biomass of *Z. capricorni* will be different between polluted and control locations.
2. Diversity and abundance of epifauna associated with the *Z. capricorni* will be different between polluted and control locations.

Materials and Methods

Study Sites

Seven locations were selected along the western side of the lake from north to the south, namely Cockle Bay, Fennel Bay, Killaben Bay, Wangi-Eraring Bay, Myuna Bay, Bonnels Bay, and Wyee Bay (Fig. 1). The sampling locations have been chosen based on the available information on heavy metal pollution within the lake. Cockle Bay, which had the highest concentration of metals both in sediment and seagrass, was treated as a polluted location, whereas the other six locations were treated as controls. Three sites at each location (approximately 200 metres apart) were selected. The sampling was done two times: winter (August) and spring (October).

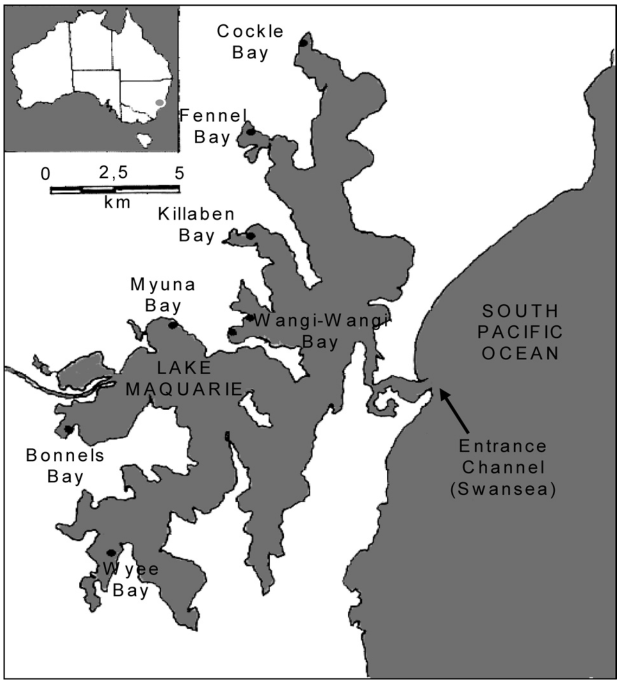

Fig. 1. Study sites in Lake Macquarie, New South Wales, Australia.

Collection of *Z. capricorni* For Measurement of Morphological Characteristics

Four replicate samples of *Z. capricorni* were collected from each site using a 25×25 cm quadrate (0.0625 m^2). The quadrate was pressed into the sediment and all seagrass plants with roots and rhizomes were lifted with the sediment, rinsed and placed in a labelled plastic bag with lake water.

In the laboratory, each sample was washed to remove excess sediment and debris, and then placed in a tray. Total number of shoots and leaves per quadrate were counted. The length of *Z. capricorni* leaf in each quadrate was estimated by measuring the length of 20 leaves to the nearest millimetre. After the measurement, all plant material including root structures was put in the aluminium foil container and dried at 60°C for 48 hr. The weight was recorded to the nearest gram for total biomass estimation (Zieman and Wetzel 1980).

Collection of Epifaunal Organisms Associated with *Z. capricorni*

Collection of epifauna associated with seagrass was done simultaneously with the seagrass collection. Three replicate samples were collected at each site. Epifauna with seagrass were sampled by covering a plant with an epifaunal sampler and cutting the plant at about 1–2 cm above the sediment using a cutting blade, which was inserted into a slit at the bottom of the sampler. The sampler was then turned upside down to wash the contents through the detachable upper part which had 500-micron mesh attached and the samples were collected into a plastic bag.

In the laboratory, samples were sieved through a 500-micron screen to separate the epifauna from seagrass material. The epifauna were preserved in 5% formaldehyde solution for further identification and enumeration. Seagrass was dried at 60°C for 48 hr. The weight was recorded to the nearest gram. Epifaunal abundance was expressed as a number of animals per gram dry-weight of seagrass.

Data Analyses

Asymmetrical ANOVA was used to compare all variables measured between polluted and control locations as we had only one polluted location with 6 controls (Underwood 1994).

Results

Shoot Density, Leaf Density, Leaf Length, and Biomass of *Z. capricorni*

Density of shoots and leaves varied among the sites within locations, and there were no further differences in those variables between polluted (Cockle Bay) and control locations both in winter and spring, even though shoots and leaves were denser in October than in August sampling. Moreover, there was no significant difference in leaf-length and total biomass of seagrass between polluted and control locations at both times. There was more variation in the length of seagrass leaves among the sites within locations in October, but biomass varied more within the sites in August (Fig. 2).

Epifaunal Abundance and Diversity

Analyses were done on the four most abundant epifaunal groups: amphipods, tanaids, gastropods, and polychaetes. The numbers in other groups (e.g. bivalves and cumaceans) were too low to be used in analyses. Abundances of amphipods were higher in October than in August. Other epifaunal groups did not show any clear pattern in abundance between the times (Fig. 3).

Abundances of amphipods, tanaids, and polychaetes did not differ consistently between polluted and control locations at both sampling times. However, the number of gastropods was significantly lower in polluted location (Cockle Bay) than in controls during the second sampling time (October). The difference was not significant during the first sampling time (August), because numbers of gastropods were also low at several control locations (such as at Fennel Bay, Wangi Bay, and Wyee Bay). Number of epifaunal species was higher in spring than in winter. Moreover, there was no significant difference in species diversity (number of species) between polluted and control locations; the number of species varied significantly among all locations in both sampling times (Fig. 3).

Discussion

Shoots and leaves of *Z. capricorni* in Lake Macquarie were denser in spring than in winter. Several studies have shown that seagrasses in estuaries of New South Wales undergo a growth cycle in which the shoot/leaf densities vary depending on the time of a year. In Lake Macquarie, shoots of *Z. capricorni* have a minimum density in autumn and winter. The growth rate is shown to be at a minimum during this time (specifically in March to April) when many leaves of *Z. capricorni* brake off and drift ashore, whereas the growth rate is at a maximum between November and January (Wood 1959). However, there was no detectable difference in leaf-length and total biomass of seagrass between the two sampling times in our study. Although shoots and leaves were denser in spring, their contribution to the total biomass was minimal. Below ground parts (roots and rhizomes) of seagrass seem to contribute more to the total biomass estimation in this study rather than the above ground parts (leaves and shoots).

There was no detectable difference in seagrass morphology between polluted (Cockle Bay) and control locations. The increased concentration of metals, which was reported in seagrass tissue in polluted area (Ambo Rappe et al. 2007) did not cause the seagrass morphology to change in the predicted way. Contamination did not cause a decrease in density of shoots and leaves or a decrease in leaf-length of the seagrass. There was also no apparent effect on total biomass. Similar result was observed in the seagrasses near the Port Pirie lead smelter, South Australia where seagrass leaves were highly contaminated, but the seagrass were growing at their usual depth (Ward 1984, 1987).

Seagrasses seem to be able to accumulate high concentrations of metals, store them in special tissues without affecting their growth rate (Ward 1989). Based on common criteria for choosing bioindicator of heavy metal contamination, namely the ability to exhibit a correlation between metal contents in their tissues and concentrations in the surrounding environment, the ability to accumulate the pollutant without being killed, sedentary life form and abundance in the study region (Phillips 1977), seagrasses may be considered good bioindicators.

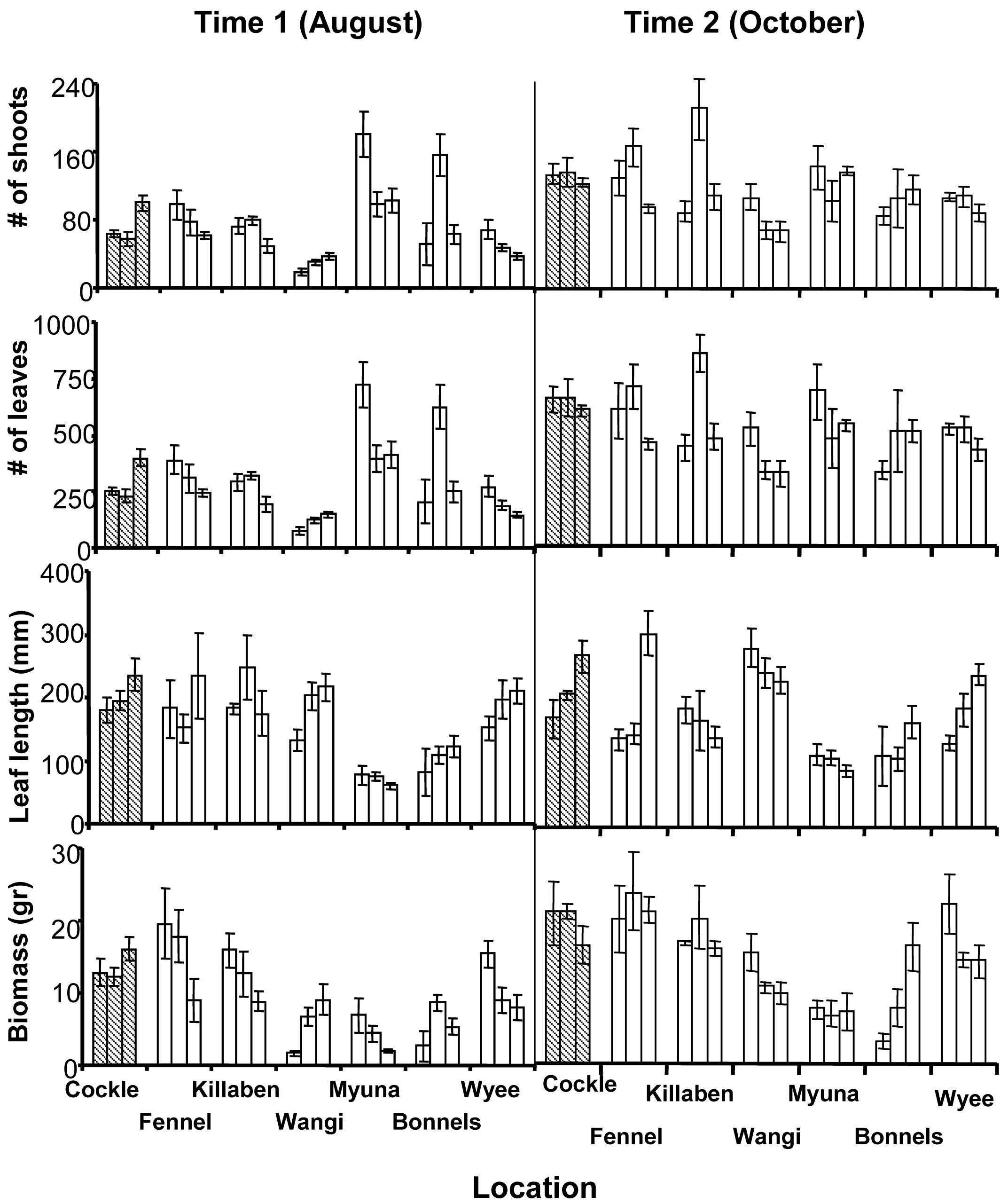

Fig. 2. Mean (±SE; n=4) shoot density, leaf density, leaf length and biomass of seagrass, Z. *capricorni* at polluted (Cockle Bay; shaded columns) and control locations (non-shaded columns) of Lake Macquarie.

However, the lack of easily measured effects on morphological characteristics or biomass means that relatively expensive analyses of seagrass tissues are required in order to detect metals. The alternative way would be to find a species (or group of species), which may change in numbers in response to unfavorable physical or chemical characteristics of their habitat (Simon et al. 2003).

Evidence of impacts of heavy metals on benthic organisms from the field has come mainly from observations of the correlative patterns (Ward 1984, Rygg 1985, Somerfield et al. 1994, Stark 1998, Burton et al. 2001, Campanella et al. 2001, Edwards et al. 2001, Filho et al. 2004). These studies have generally revealed that increasing contamination by heavy metals in sediments is correlated with decreasing numbers of species and changes in abundance of benthic fauna. The establishment of correlative relationships between disturbances and benthic assemblages in the field have been considered as a first and necessary step towards understanding environmental impacts (Underwood and Peterson 1998).

In this study, abundances of gastropods were significantly lower in polluted location. Most gastropods in seagrass beds feed on epiphytic microalgae growing on the leaf surfaces and on detritus particles present on the sediment (Hemminga and Duarte 2000). Since the epiphytic algae can

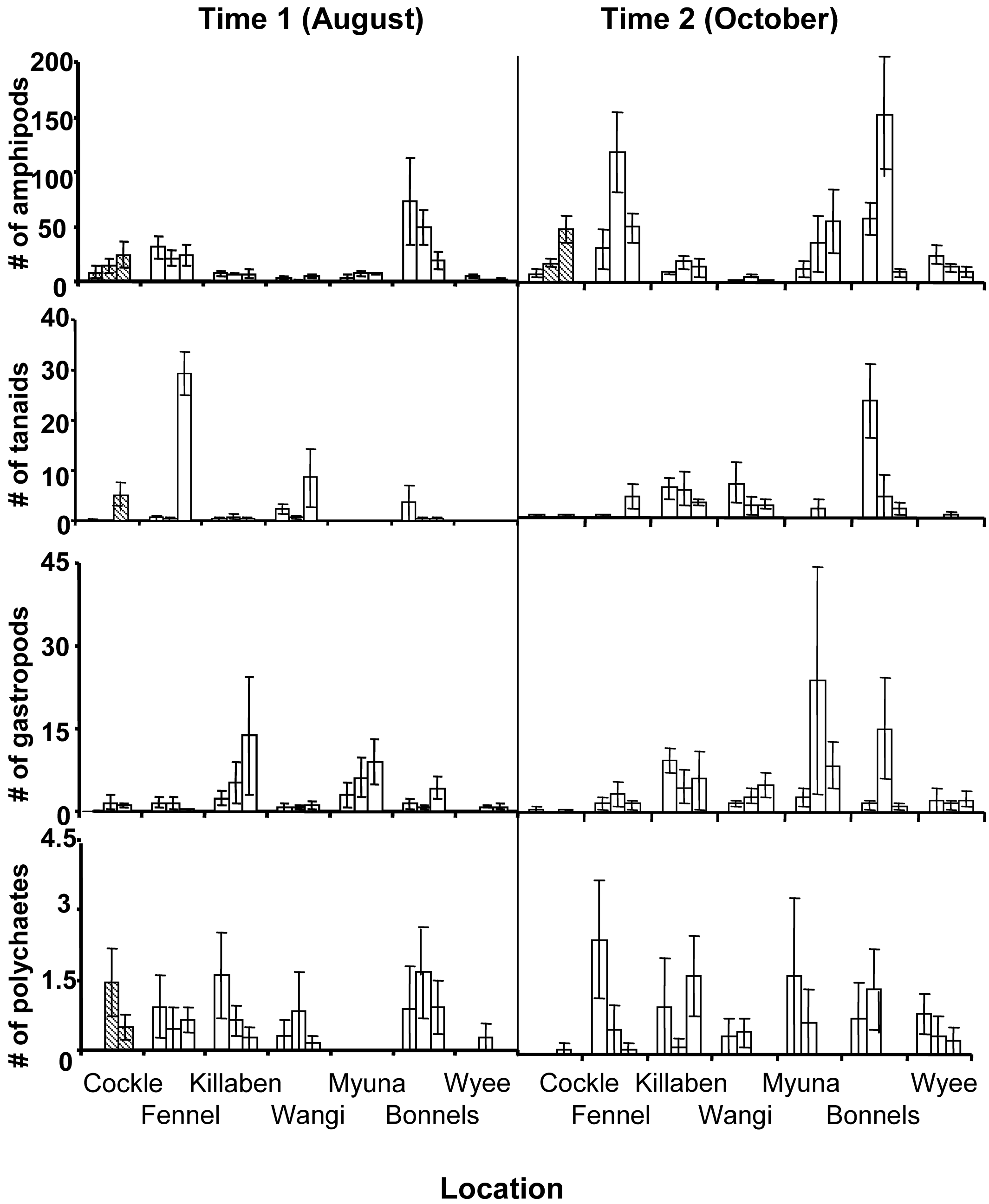

Fig. 3. Mean (±SE; n=3) number of amphipods, tanaids, polychaetes, and gastropods at polluted (Cockle Bay; shaded columns) and control locations (non-shaded columns) of Lake Macquarie.

contain high concentrations of metals (Ward 1984) and the detritus in seagrass bed (which is primarily derived from breakdown of seagrass leaves) can also contain high concentrations of metals, the consumption of contaminated food is likely to be the main cause of the decreased abundance of this group in the polluted location.

There was no significant difference in abundances of other epifaunal organisms (amphipods, tanaids, and polychaetes) between polluted and control locations. This may be due to the higher mobility of small crustaceans, particularly amphipods (Costello and Myers 1996), and particularly patchy nature of their distributions. Epibenthic seagrass fauna (the isopod *Cymodoce longicaudata*) has been found in its normal abundance in the highly contaminated area, even though the acute toxicity tests showed that it was acutely affected by the effluent from a lead smelter (Ward 1984). An experiment on the effects of heavy metal effects on assemblages in soft sediment also found that the mean number of individuals and number of taxa did not decrease more in units contaminated with metals than in the control units (Lindegarth and Underwood 2002).

In summary, the present study discusses the possibility to use biological indicator organisms to define areas of heavy metal pollution. This method is proposed to eliminate analyses of metals from water or sediment which are expensive and laborious. The measurement of metal content in biota (we used seagrass, *Z. capricorni*; Ambo Rappe et al. 2007) demonstrated that the concentrations of metals in seagrass tissues were correlated with the levels in the sediments, but seagrass morphology was not affected. Epifaunal abundance may be a more cost-effective bioindicator of heavy metal pollution. The abundance of gastropods, however, was the only group that could potentially be used as an indicator of metal pollution in this study. Abundances of other groups varied significantly at small spatial scales and displayed no consistent differences in abundance related to the metal pollution.

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