A Comparison of Anthropogenic Discharge Quality and Quantity into Queensland East Coast Catchments.

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Introduction.

Prawn (= shrimp) farming in Australia currently produces approximately 2,400 tonnes of prawns (mostly *Penaeus monodon* - black tiger prawns) from about 500 hectares of ponds annually with a farm-gate value of approximately $45 million (Rosenberry, 2000). Currently, the majority of production comes from Queensland (about 400 hectares of ponds) and is sold on the domestic market. Prawn farming in Asia and the Americas has led to severe environmental impacts (e.g. Lin, 1989; Phillips *et al.*, 1993; Hopkins *et al.*, 1995; Boyd & Clay, 1998; Paez-Osuna *et al.*, 1998). Prawn farming in Australia is expanding and the industry estimates that the area under production is likely to double over the next decade.

There is community and Government concern regarding the potential environmental impacts of an expanding Australian prawn farming industry, particularly the potential impacts of effluent discharged from prawn farms into the World Heritage listed Great Barrier Reef Marine Park or other sensitive marine areas (e.g. Hervey Bay). The major concerns in regard to prawn farm discharges in Australia is the release of nitrogen (N), phosphorous (P), and suspended solids (organic and inorganic).

As well as the discharge of nutrients and suspended solids, concerns overseas have also been raised about the impacts of prawn farming from: the destruction of mangrove forests and wetlands (e.g. Ong, 1982; Parks & Bonifaz, 1994; Primavera, 1995), the addition of chemicals and their release into the wild (e.g. Primavera *et al.*, 1993; Beveridge *et al.*, 1994; Paez-Osuna *et al.*, 1998), and social issues (e.g. Pollnac, 1992; Stonich, 1995; Stonich *et al.*, 1997; Stanley, 1998). Several of these issues (e.g. the destruction of mangroves and wetlands and chemical usage in grow-out ponds) in Australia are addressed by regulation (that is supported by the industry) while the remainder are dealt with in the *Environmental Code of Practice for Australian Prawn Farmers* (Donovan, 1999).

Prawn farming is just one potential source of pollutants to the marine ecosystem, others also arise from agriculture and urban developments. The environmental impacts of prawn farming in Australia have been rigorously examined (e.g Trott & Alongi, 2000; Preston & Rothlisberg, 2000; Preston *et al.*, 2000). This paper compares the environmental impacts of prawn aquaculture in Queensland (currently the major area of production) to 1) other land uses (crop production, grazing) and sewage discharge and 2) the environmental impacts of prawn farming in other countries.

Fundamental considerations when comparing the ecological impacts of prawn farm effluent with agricultural run-off and sewage discharge.

Agricultural run-off in Queensland usually enters river and marine systems as infrequent but intense short lived events associated with heavy rains and/or cyclonic activity (Mitchell *et al.*, 1996). Such high intensity loadings from diffuse sources are considered to give rise to large but short lived impacts (Moss *et al.*, 1996). From observations of flood plumes, the spatial extent of these impacts can be large extending hundreds of kilometres along the coast and up to 20 kilometres seaward of the river mouth (Steven *et al.*, 1996). While flooding is natural, changed land use patterns add additional stress (e.g. increased suspended sediments and nutrients) to habitats such as inshore reef areas at a time when the system is already under heavy stress (e.g. Kinsey, 1987).

In terms of ecological theory, agricultural run-off in Queensland can be considered a 'pulse' disturbance and these are characterised by rapid changes in abundance of animals and possibly long-term disappearances or local/regional extinctions. Such pulse disturbances can alter the structure and function of aquatic ecosystems if they impact on a key component of the ecosystem. For example, severe flooding in the Mary River catchment caused large scale loss of seagrass and an associated decrease in dugong abundance in Hervey Bay and the Great Sandy Straits (Preen *et al.*, 1995). Seagrass was killed because suspended sediments prevented sufficient light from reaching the seagrass over a prolonged period. The development of the Mary River catchment for agriculture was considered by Preen *et al.* (1995) to be an important factor contributing to the elevated level of suspended sediments and ultimately the death of seagrass and dugongs. Although not directly assessed, large changes in the diversity and abundance of a wide range of other organisms including fish, and benthic epifauna and infauna are likely to have resulted from the observed seagrass die-back (see Heck & Orth, 1980; Ferrell & Bell, 1991).

In contrast to agricultural run-off, sewage effluent can be considered a ‘press’ disturbance on the marine environment. That is, a disturbance which has a consistent and sustained impact on the marine ecosystem over time. Importantly, because of it’s point source nature, sewage effluent tends to have localised impacts (e.g to a river or portion of an embayment). In Queensland as a whole, point sources of nutrients form a very small fraction of a catchments total nutrient load (Moss *et al.*, 1992).

Effluent from prawn farming is neither truly a press or a pulse disturbance. Owing to the temporal variability in discharge quantity and quality and the periods when discharges are zero, prawn farm effluent does not show the consistent and sustained characteristics of a press disturbance (Preston *et al.*, 2000; Trott & Alongi, 2000). There appears to be no documented cases of prawn farming causing rapid changes in abundance of animals or long-term disappearances of animals generally associated with a pulse disturbance.

As well as the issues of comparing the spatial scales of impacts and their nature (pulse and press disturbance), consideration also needs to be given to other factors when comparing prawn farm effluent with sewage effluent and agricultural run-off. With respect to agricultural run-off, caution needs to be exercised in extrapolating measurements of parameters such as N or P concentrations between catchments. Factors such as rainfall patterns, physical and chemical soil parameters, river hydrology and terrestrial vegetation cover that influence run-off characteristics vary between catchments (Eyre & Davies, 1996). Second, while correlations (strong) can be drawn between a specific land use practice (e.g. sugarcane farming) and run-off characteristics and impacts, without appropriate (and very costly) experimentation these do not always imply causation.

Agricultural run-off from cultivated crops in Queensland.

Cropped lands are identified as the source of most suspended sediments and nutrients in Queensland waterways on a per hectare basis (Bramley & Johnson, 1996). Bramley and Johnson (1996) assert that it is the general nature of intensive agriculture, a disturbed and simplified ecosystem with high nutrient inputs (fertilisers) rather than the specific production system employed (e.g. sugarcane compared to maize), which results in downstream impacts. Various crops are grown in Queensland with cereal crops (wheat and sorghum) being the most important in terms of area under cultivation (Figure 1). Sugarcane is third in terms of area under cultivation, however, production of cereal crops is centred in the Darling Downs Statistical Division, whereas sugarcane production is centred in catchments that drain into the Great Barrier Reef (ABS, 1998). Sugarcane farming is responsible for the majority of nutrient (fertiliser) inputs to agricultural land in most east coast catchments (Crossland *et al.*, 1997). For these reasons, this section will focus principally on run-off from canelands.

One of the major concerns regarding the downstream effects of agriculture is the use of fertilisers that results in additional N and P entering the aquatic system. In coral reef environments, additional N and P can increase benthic algal and phytoplankton growth which potentially impacts on corals (e.g. Smith *et al.*, 1981; Bell, 1991; Crossland *et al.*, 1997). Globally, fertiliser inputs (per unit area) to cultivated land shows a clear increase over time with a six fold increase in N fertilisation and a three fold increase in P fertilisation from the 1960s to the present and is also coupled with a global increase in the area under cultivation (Tilman, 1999). Fertiliser usage also increased dramatically in Australia from the 1950s although a levelling out of P fertiliser usage is evident from 1980 to 1995 (Figure 2). There is information to suggest at least a doubling of the rate of fertiliser application to canefields per unit area in the Herbert River between the 1970s and

1994 (Johnson, 1996). Garside *et al.* (1997) reported that N additions to Queensland canelands increased from below 150 kg/ha^{-1} before 1969 to 210 kg/ha^{-1} in the 1970s, then fell slightly in the late 1980s but rates of up to 300 kg/ha^{-1} are thought to have been used by some producers in the late 1990s. Sugar yield has not increased in proportion to increasing rates of fertiliser usage and, in fact, excess N fertiliser usage is implicated in the declining sugar content of sugarcane (Garside *et al.*, 1997).

An estimated 8,800 tonnes of nitrogen (N) and 1,300 tonnes of phosphorous (P) are lost annually from canefields via Queensland's east coast rivers (Rayment,1999). Using a different method of calculation, Bloesch *et al.* (1997) estimated that 1,450 tonnes of P from canefields enter Queensland east coast river systems. The values calculated by Rayment (1999) for N and P loss will be used throughout the remainder of this report when discussing nutrient inputs from canefields. In order to calculate N and P loss from agriculture other than sugarcane farming, the estimates of fertiliser application to agricultural land other than canefields contained in Crossland *et al.* (1997) and an estimate of the percentage of N and P fertiliser lost are used. Rates of N and P loss from canefields differ between catchments but Rayment (unpublished data) suggests a value of 14% for N and 17% for P fertiliser inputs. For the purpose of this report, it is assumed that 15% of N and P applied to agricultural land other than canefields is subsequently lost in run-off. Using the fertiliser application rates in Crossland *et al.* (1997) and the estimate of fertiliser loss it is estimated that fertiliser derived P and N lost from agricultural land other than canelands are 878.1 t/yr^{-1} and 3,502.5 t/yr^{-1} respectively. These values are for cultivated crops other than sugarcane and grazing combined.

Loss of N and P to the aquatic environment is not the only downstream effect from crop production, increased sediment loads from soil erosion is also an important issue. Increased sediment loads can kill corals and seagrass through smothering and increasing turbidity, and accumulation of sediments on coral reefs may change the settlement patterns of larvae (e.g. Preen *et al*., 1995; Crossland *et al.*, 1997). Many chemical contaminants may also enter aquatic systems attached to sediment particles (Crossland *et al.*, 1997). Run-off from cultivated crop contributes 8% of the sediment input into east coast rivers, which is small in comparison to the 66% contributed from grazing lands (Neil & Yu, 1996). However, Neil and Yu (1996) recorded considerable spatial variability in the relative contributions to sediment input from grazing and croplands with grazing sources predominating in the wet-dry tropics and cropland sources predominating in the humid tropics.

Soil loss from canelands is estimated to average 148 $t/ha/yr^{-1}$ under conventional cultivation but can be below 15 $t/ha/yr^{-1}$ with no-tillage and between 1-4 $t/ha/yr^{-1}$ under a no-tillage and a green cane trash blanket (GCTB) regime (Prove *et al.* 1995; Rayment, 1999). Large reductions in soil loss are also possible for pineapple farms by the addition of crop trash mulch (Christensen, 2000). A potential environmental trade-off of reducing sediment loss through using GCTB practices is that it can lead to greater rates of fertiliser loss which is then compensated for by farmers with greater application rates (reviewed in Garside *et al.*, 1997).

Run-off from agriculture can contain a variety of agrochemicals. Compared to other agricultural industries, cane farming is a major user of agrochemicals because of high application rates and the relatively large area of production (Rayment, 1999). Agrochemicals (or their derivatives) detected in run-off or ground water in Queensland include Atrazine, DDT, diuron, chloropyrifos, OCDD and dieldrin (Haynes *et al.*, 1998; Rayment, 1999). Crossland *et al.* (1997) justifiably discusses that when determining the effects of agrochemical contaminants, consideration must be given to not only present agrochemical use, but also to the accumulated residues of agricultural products (in agricultural soils or coastal sediments) that have not been used for decades.

Assessing the ecological impacts of agrochemical contaminants in the marine environment is extremely difficult. Potential faunal impacts include: direct mortality, genetic or physiological changes or altered behaviour patterns (e.g. Muirhead-Thompson, 1988). Ecological impacts may not occur for long periods (e.g. decades) and be extremely difficult to detect. Agrochemical products (or their derivatives) are known to occur in the tissue of a variety of fauna including birds, fish and dugong (e.g. Willis & McDowell, 1982; McDougall *et al.*, 1989; Russell *et al.*, 1996; Haynes *et al.*, 1998; Mortimer & Cox, 1999). However from a broader ecological standpoint, caution needs to be exercised in interpreting what the presence of a compound in the tissue of an animal actually means. Beyond the compound being present in animal tissue (or indeed the environment) there may be no further demonstrable ecological impacts. However, because of the uncertainty regarding unequivocally identifying an impact and the potential time frame over which an impact may occur, extreme caution should be exercised in assuming a chemical does not have an ecological impact.

Agricultural run-off from grazing in Queensland.

In the order of 140 million hectares of Queensland are categorised as grazing lands and grazing is the dominant land use in all Queensland catchments except for the Sunshine Coast (Zann, 1995; ABS, 1998). The concerns about inputs to the aquatic system of N and P, suspended sediments and agrochemicals discussed in the previous section are relevant with respect to grazing, however the magnitudes of the inputs vary.

Neil and Yu (1996) estimate that total sediment yield to Great Barrier Reef catchments has increased about 3.8 times since European settlement to 28 Mt/yr^{-1}.

Modelling work clearly points to grazing lands as the major source of suspended sediments to the Queensland east coast. With 66% of sediment input to the Great Barrier Reed lagoon attributable to grazing, 26% to natural processes and 8% from cropping. In the Herbert River catchment, 21% of sediment yield was sourced from areas used for cattle grazing which is considerably larger than the 1% from canelands (Johnson *et al.*, 1998). Total sediment input varies between catchments with a disproportionate amount coming from the Burdekin and Fitzroy rivers (Neil & Yu, 1996). Stocking density affect sediment yield with a four-fold increase in response to a tripling of stocking rate (Prove, 1992). The estimated economic returns for the increased sediment loss due to land use intensification varies between catchments. Neil and Yu (1996) calculate that in the humid tropics the monetary return from agricultural production is more than $1,000 per tonne of production related soil loss, whereas in the Burdekin and Fitzroy catchments the monetary return is as low as $50 per tonne of soil lost.

The amount of sediment associated N and P inputs from grazing can be calculated from the estimates of sediment loss from grazing calculated by Neil and Yu (1996) and by the mean N and P content of soils eroding into the GBRWHA. The latter are calculated by Furnas and Mitchell (2001) to be 0.13% N and 0.04% P. Using these figures gives annual values of sediment associated N and P inputs to the aquatic system from grazing of 18,018 t/yr^{-1} and 5,544 t/yr^{-1} respectively.

Sewage discharge and its potential ecological impacts.

Potential issues of concern for the marine ecosystems (and human health) from sewage discharge include: bacteria and viruses, oil and grease, organic matter, various chemical, heavy metals and nutrients. Queensland has 131 sewage treatment plants discharging effluent into coastal freshwaters, estuaries or inshore marine waters (Zann, 1995). These treatment plants generally treat waste only to a secondary standard which removes most suspended solids and bacteria but not nutrients. The main concern about bacteria and virus in ocean dumped sewage sludge relates directly to public health than their ecological impact. Oil and grease in sewage sludge if not dispersed effectively can accumulate to form floating grease balls. In this form the oil and grease is partly protected from degradation and can survive intact for a considerable period. Bacteria and viruses within the greaseball are also protected and can present a significant human health hazard if washed up on a populated beach. The main problem with organic matter in sewage discharge is its propensity to cause sags in the dissolved oxygen level when it is decomposed by bacteria in the water.

In terms of chemicals and heavy metals in sewage sludge, although elevated heavy metal levels have been detected frequently in organisms adjacent to sewage discharges, only a few cases of acute trace metal toxicity to marine organisms have been documented with the most compelling being the Minimata Bay (Japan) incident (Kudo and Miyahara, 1991), where very large quantities of easily assimilable methyl mercury were released into the Bay. Organochlorides present in sewage sludge have been implicated in sub lethal effects on fish (fin rot) in the U.S. (Sherwood, 1982). Little to nothing appearsto be presently known about the long-term fate of detergent breakdown products (e.g. nonylphenol) contained in sewage discharge.

Sewage effluent treated to a secondary level still contains N and P. While estimates of sewage discharge volume by region are given in Zann (1995), state-wide estimates of N and P inputs (t/yr^{-1}or mg/L^{-1}) from sewage discharge were not found by the author in the literature. However, there are estimates for the Brisbane River catchment from the Brisbane River and Moreton Bay Wastewater Management Study (BRMBWMS, 1997) and for Southport (Gold Coast) (Moss & Cox, 1999). Conservative estimates of N and P (t/yr^{-1}) in sewage discharge in each Queensland East Coast catchment were calculated assuming concentrations of 4.0 mg/L of N and 2.0 mg/L of P (Table 4). Using these values, an estimated 1,928 t/yr^{-1} of N and 798 t/yr^{-1} of P are discharged from sewage treatment plants into Queensland east coast waters. There is likely to be regional variation between concentrations of N and P in sewage discharges, hence, the selection of conservative values for the discharge concentrations of N and P. Not surprisingly, the catchments with the greatest sewage discharges of N and P were those supporting large populations (e.g. Brisbane River catchment).

Prawn farming in Queensland and its environmental impacts.

Concerns regarding the environmental management of prawn farming in Australia is a priority issue for the industry itself and the broader community (Fisher, 2000). Prawns farm discharge are a point source of N, P and total suspended solids (TSS) to the marine system. Preston *et al.* (2000) examined levels of N, P and suspended sediments in effluent from several Australian prawn farms with the levels dependent on: rainfall, farm location, season, age of ponds, other activities in the catchment, and prawn farm management practices. The State Government in their review of prawn aquaculture licensing have set the following discharge standards for prawn aquaculture (averaged over the growing season): N= 0.48 kg/ha^{-1}/day^{-1}, P= 0.06 kg/ha^{-1}/day^{-1}, and TSS= 12 kg/ha^{-1}/day^{-1}.

These discharge standards are used to calculate N, P and TSS inputs from prawn farming into Queensland east coast rivers. It is acknowledged that while these values (in particular for N) are not able to be currently met by all existing farms, all new farms and extensions of existing farms will have treatment systems to meet these standards (Preston *et al.*, 2001). The area under production for prawn farming in Queensland is estimated at 400 hectares. The prawn growing season varies with latitude, year round production is possible in the tropics, but in more temperate regions the growing season is restricted to approximately six months and centred on the summer period (Preston *et al.*, 2000). For the purposes of this report, the average length of the growing season throughout the State is assumed to be nine months. Using these figures, estimates of annual discharge from prawn farming in Queensland are 1,314 t/yr^{-1} of TSS, 6.6 t/yr^{-1} of P, and 52.6 t/yr of N.

The ecological impacts of prawn farm effluent on a tropical mangrove estuary in the Port Douglas region was investigated by Trott and Alongi (2000). They found no significant differences between the (putative) impacted (with prawn farm effluent) and control (no prawn farm effluent) estuaries in TSS and dissolved nutrient concentrations. Water quality (dissolved oxygen, biological oxygen demand, pH and salinity) and phytoplankton biomass were at ambient levels within 1 km downstream of the discharge site, based on a comparison with long-term, pre impact data. Within 1-2 months after pond discharge ceased, water quality and phytoplankton biomass at the discharge site returned to levels equivalent to those in control estuaries. Trott and Alongi (2000) concluded that the limited spatial and temporal impact of prawn farm effluent suggests that the effluent was dissipated by tides and assimilated and/or mineralised by the estuarine food web.

Destruction of wetlands and other sensitive habitats is frequently raised as a critical negative issue arising from prawn aquaculture. Destruction of wetlands and other sensitive habitats for prawn aquaculture is minimised or eliminated in Queensland through licence requirements that:

- prohibit pond development below high tide level, in freshwater wetlands or ephemeral swamps;
- allow only minimal disturbance to marine plants and then only for intakes/outlets; and,

- avoid wherever possible disturbance of acid sulphate soils; or if avoidance is not possible, the works must be in accordance with an approved acid sulphate soil management plan.

Donovan (2000) sets out broad principles for chemical usage in Australian prawn aquaculture. All chemicals used in aquaculture need to be registered with the National Registration Authority. Draft guidelines prepared by the Australian and New Zealand Environment and Conservation Council and the Agriculture and Resource Management Council of Australia and New Zealand that set acceptable limits for chemical elements and compounds that can be released into the aquatic environment (by any practice) are currently being finalised. Antibiotics are not permitted to be used in prawn grow-out ponds in Queensland and, even if permitted, would be a significant additional cost to the farmer with a questionable benefit. Limited amounts of antibiotics however, may be prescribed for use in prawn hatcheries to reduce bacteria which can cause mortality in the larval-rearing tanks.

Environmental and social issues in overseas prawn farms.

There is a wealth of literature on potential and realised impacts from prawn farming overseas and these impacts fall into four categories:

1) clearing of wetlands;

2) nutrient, sediment and chemical inputs into the coastal zone;

3) general ecological efficiency of prawn farming; and,

4) social equity/justice issues.

Large scale wetland destruction for the purposes of constructing prawn farms was widespread in many developing countries. The importance of wetlands for aquatic ecosystem function are well established and will not be discussed here. It is suggested by

Hopkins *et al.* (1995) that globally wetland destruction is the most significant adverse environmental impact from prawn farming. An estimated 19% of mangrove forests in Ecuador were destroyed prior to 1990 for the construction of prawn ponds (Aiken, 1990). Large scale destruction of mangroves for aquaculture has occurred in most Asian countries that farm prawns, however, it is not always possible to quantitatively separate mangrove loss from prawn farming and finfish farming (particularly milkfish *Chanos chanos*) (Ong, 1982; Chamberlain, 1991; Primivera, 1991; Chua, 1993).

Prawn aquaculture in many Asian countries also relies on the collection of wild post-larval or juvenile prawns and the destruction of wetlands frequently reduces their availability and hence the long-term viability of prawn aquaculture in these countries (Landesmann, 1984). Destruction of mangrove nursery habitat for pond construction in Ecuador, together with over collection of post larvae, are implicated in the decline in the numbers of postlarvae available for capture for pond stocking in that country (Twilley, 1989).

Prawn farms can be abandoned if no longer economically viable (Gujja & Finger-Stich, 1996; Stevenson, 1997) and Kautsky *et al.* (1997) argues that the impacts of wetland destruction from aquaculture (including prawn farming) are largely irreversible. However, Stevenson (1997) outlines that redevelopment of mangroves in disused prawn ponds is a potentially viable. Abandoned prawn ponds in the Mai Po Marshes Nature Reserve (Hong Kong) serve as important feeding areas for piscivorous waterbirds (Cha *et al.* 1997). There is also the potential to maintain mangroves within prawn farms systems, and such an integrated prawn and mangrove farming system can increase economic returns for farmers (Binh *et al.*, 1997).

Concerns regarding nutrients (N and P), suspended sediments and various chemicals in prawn farm effluent are a recurring theme in overseas prawn farming. An important consideration in assessing prawn farm effluent is the quality of the intake water which may already have elevated nutrient levels from non-point sources (e.g. agriculture) (see Chua *et al.*, 1989). Despite being frequently raised as an issue, there are surprisingly few quantitative studies that actually assess the quality of prawn farm effluent. Briggs and Funge-Smith (1994) estimated concentrations of N (2.19-3.45 mg L^{-1}), P (0.27-0.45 mg L^{-1}) and solids (120-165 mg L^{-1}) discharged in effluent from prawn farms in Thailand. Paez-Osuna *et al.* (1999) predicted that P and N loads discharged into the Gulf of California from approximately 26,000 hectares of prawn farms in NW Mexico during 1998 would be 834 and 2,903 tonnes respectively. Ziemann *et al.* (1992) estimated that prawn farm effluent in Hawaii, on average, contains 0.69 mg L^{-1} of N and 0.23 mg L^{-1} of P. These overseas studies appear less comprehensive than the study of Preston *et al.* (2000) conducted in Australia.

A wide range of chemicals can be used by overseas prawn farmers including chemotherapeutants (e.g. antibiotics), vaccines, soil and water treatments, feed additives and disinfectants (reviewed in Primavera *et al.* 1993). The use of some of these chemicals appear to pose more of a threat to the viability of prawn aquaculture and human health than to the environment directly. However, the impact on the environment of these chemicals generally remains poorly studied. Formalin and malachite green which are widely used in Asian prawn ponds, are known to be toxic to prawn nauplii at low concentrations (Castille & Lawrence, 1986), indicating the potential to adversely impact biota in coastal waters when discharged in effluent. Both compounds are also potentially harmful to humans. The indiscriminate use of antibiotics was a major contributing factor to the collapse of prawn crops in Taiwan through the development of antibiotic resistant prawn pathogens (Lin, 1989). Hopkins *et al.* (1995) suggest the use of excessively large amounts of antibiotics was partly driven by unscrupulous agents for drug companies who persuaded poorly educated prawn farmers to buy and use as much of their companies products as possible. There is concern that the antibiotic resistance of prawn pathogens could be transferred to human pathogens (Phillips *et al.* 1993).

Aquaculture consumes natural resources and relies on ecosystem services, as do agriculture and wild capture fisheries (see Boyd & Clay, 1998; Naylor *et al.* 1998). The ecological efficiency of, and "natures subsidy" towards prawn aquaculture has been raised overseas as an environmental issue that needs to be considered and addressed by the industry. The principal resources consumed by prawn aquaculture are habitat (discussed previously), food, and energy. Larsson *et al.* (1994) calculates that a semi-intensive prawn farm in Colombia needs a spatial ecosystem support (or 'ecological footprint') that is between 35-190 times larger than the surface area of the farm. This calculation includes the area of mangrove nursery necessary for postlarvae production, the sea and land area necessary to support production of fish meal and vegetable matter in pelletised food, the area needed to create energy used by the farm, and the area of forest needed to sequester CO_2 released during production. A large component of the ecological footprint was the production of fishmeal to include in pelletised diets; but, there is ongoing research to find fishmeal replacements (e.g. Smith, 1996). Caution should be exercised in directly comparing the ecological footprint of prawn farming in the Columbian example to other countries (e.g. Australia) with different production practices. Importantly, there does not appear to be a comparative analysis of the ecological footprint of prawn aquaculture with agriculture, wild capture commercial fisheries, or recreational fisheries. Such a comparison is necessary to put the impacts of each activity in perspective.

Social conflict is documented in nearly every country where prawn farming is undertaken. The reasons for conflict are principally because of a suite of inter-related issues including: 1) use, modification, or destruction of public resources (e.g. wetlands) that traditionally have been used by many different groups, 2) the potential for public resources to become privately owned with economic and social benefits (e.g. employment) of these resources being (or perceived to be) reduced or taken away from the majority of local community members and benefits accruing to a small number of individuals, companies or corporations, 3) the export orientated nature of prawn farming and the diversion of resources to it and away from practices that produce food for local communities (e.g. crops or artisinal fishing), 4) the rapid growth of the industry, and 5) the general lack of Government policies, planning and management of prawn farming in many countries.

Direct and indirect impacts to existing uses such as artisinal fishing, agriculture and forestry from prawn aquaculture through wetland destruction, post larval/juvenile collection from the wild, or preventing access appears the most frequently raised social issue regarding prawn farming. Gujja and Finger-Stich (1996, p. 35) summarise the Asian situation: '*Historically, many coastal and forest ecosystems have been held in common to satisfy a variety of users and practices. Access was not legally codified but subject to negotiations among the community's various groups. For example, access to the seashore by local fishermen, while not technically a legal right, was certainly customary. But the outside companies and investors who own shrimp farms tend to view as private property what have been seen as commonly held resources. Thus, they frequently bar local residents' passage through the farms to the seashore or the forest* '. Similar sentiments are echoed by Stonich (1995) and Stanley (1998). Flaherty *et al.* (1999) discusses the potential for environmental degradation and flow-on social impacts (e.g. reduced employment) from changing rice paddies to prawn farms but fails to acknowledge that rice paddies themselves represent an extremely modified production system with significant ecological impacts.

The benefits of prawn farming are clearly seen to accrue to a small number of elite individuals, companies or corporations whereas the costs are borne by the whole local community. This is a debate which has lead to considerable emotion in the literature but this emotion does not always appear to be consistently backed up by solid data or analysis. Stonich *et al.* (1997, p.169) describe prawn farm owners in Central America as: '*Wealthy elites* (who)*have capital as well as the connections to seek external investors and obtain expert technical assistance. Also they are politically capable of dealing with the complex property rights laws, permit systems, and relevant import and export regulations* '. Stonich and Bailey (2000) go as far as to claim that prawn farm ownership in developing countries is obtained and maintained through corruption, collusion with government officials and thuggery at the expense of existing rural communities and public resources but provide little direct evidence for these claims. In India, Goss (1998, p.384) reports that opposition to prawn farming: '*is the test case for the poor in the battle against capital and for the rights of people to utilise their own resource* '. Vandergeest *et al.* (1999) discusses that the dominance of large corporate prawn farms in countries such as Honduras, Mexico, parts of Thailand and Bangladesh is due to privileged access to land, water and/or credit which is an outcome of previous or existing agarian relations, government policies and the physical landscape. If this is correct, the social conflicts surrounding prawn farming in some developing countries, may be a symptom of larger social, political and economic problems and inequities- not the driving force as implied by several authors (e.g. Stonich *et al.* 1997; Goss, 1998; Stanley, 1998; Stonich & Bailey, 2000).

Production from prawn farms in developing countries is principally exported to wealthy nations and has replaced products that were previously consumed locally. For instance, production of the domestically consumed milkfish (*Chanos chanos*) in the Philippines fell approximately 35% between 1982 and 1986 with this decrease partly attributed to a shift towards export orientated prawn farming (Primavera, 1991). Flaherty *et al* . (1999) discusses how prawn farming impinged upon traditional rice production areas in central Thailand with rice being a stable of local diets.

However, what is lacking in much of the social science literature (e.g. Stonch, 1995; Stonich *et al.* 1997; Stanley, 1998; Flaherty *et al.,* 1999; Stonich & Bailey, 2000) is rigorous bio-economic modelling to verify the frequent assertions that prawn farming and its environmental impacts are costs borne by the local community with little of the benefits returning to the local community. For instance, a bio-economic model could have been constructed by Flaherty *et al.* (1999) to compare the costs and benefits of prawn farming and rice production in rural Thailand including the impacts on natural capital of both practices. Without such an analysis, the conclusions of Flaherty *et al.* (1999) that prawn farming in rural Thailand may potentially lead to increased marginalization, exclusion and pauperization of the rural community are difficult to unequivocally accept or reject.

The rapid and for the most part ad-hoc growth of prawn aquaculture overseas has also added to the observed social conflicts. Large-scale prawn farming is at a disadvantage because it is not a traditional activity. Compared to a new practice, there appears to be a greater social acceptance of a practice when it is traditional even if it has a clear environmental impact (e.g. agriculture). Certainly the rapid growth, together with the lax regulatory environments of several major prawn farming countries has not helped strategic planning to minimise the social impacts of prawn farming in many countries.

There are genuine attempts in several countries from the industry, regulators and environmentalists to learn from the problems of the past and improve the economic, environmental and social performance of prawn farming. For instance, a large series of case studies involving the World Wildlife Fund for Nature, the Network of Aquaculture Centres in Asia Pacific, FAO and the World Bank have been funded with the aim of suggesting practices for improving farming practices (Boyd, 2000). The Global Aquaculture Alliance has developed a detailed Code of Practice for Responsible Shrimp Farming which addresses the major issues raised in this section. However, on-ground changes to farming practices as a result of this code and other industry initiatives to reduce environmental impacts overseas do not appear to have reached the mainstream scientific literature at the current time.

A comparison of the ecological impacts of prawn farming with agricultural run-off and sewage effluent in Queensland.

There is limited but valuable overseas information on comparing prawn farm effluent (N and P) with sewage and agricultural run-off. Effluent released by prawn farms in Thailand was estimated to contain between at least eight times less total nitrogen (N), at least six times less total phosphorus (P), but at least six times more dissolved solids than secondary treated domestic sewage (Beveridge *et al.*, 1997). Paez-Osuna *et al.* (1999) predicted that P and N loads in the Gulf of California from NW Mexican shrimp farms in 1998 would be 834 and 2,903 tonnes respectively which corresponds to 1.3 % and 3.1 % of the estimated anthropogenic nutrient inputs in that region. Agriculture was the main contributor of P and N with an estimated input of 26,714 (54.9 % of total) and 53,476 (57.7 % of total) tonnes per year respectively; while municipal (sewage) contributions were 1,730 (3.6 % of total) and 3,806 (4.1 % of total) tonnes per year respectively. The major agricultural crops grown in NW Mexico are sugarcane, rice and vegetables (Paez-Osuna *et al.*, 1998). Paez-Osuna *et al.* (1998) estimated that shrimp farming contributed only 1.5 % and 0.9 % of the annual N and P from anthropogenic sources in all Mexican coastal states (Figure 3).

Prawn farming differs from agriculture in Queensland in two very important ways. First, agriculture in Queensland (and Australia) is based almost solely on introduced plants and animals whereas prawn farming is based on culturing native species. The negative ecological impacts of introduced species are well documented (e.g. Atkinson, 1989; Diamond, 1989). Second, prawn farm effluent can and is being treated (Jones & Preston, 1999; Preston *et al.*, 2000); whereas run-off from agriculture is difficult to treat because of diffuse nature. While, changed farming practices such as GCTB may significantly improve some aspects of agricultural run-off (e.g. TSS), it still does not constitute treatment of discharges. Furthermore, there are stringent discharge requirements for effluent from prawn farms, but similar requirements are not in place for agricultural practices. There is clearly a need to focus management on all activities that impact on catchments and water quality.

In Queensland, the area under cropping and grazing is vastly greater than that dedicated to prawn farming (Figure 1). Furthermore, even with the projected growth of prawn aquaculture in Queensland, the area under production would not rival that of major crops such as sugarcane for the foreseeable future. In terms of nutrient inputs (N and P) to aquatic systems, prawn aquaculture makes a very minor contribution relative to agriculture and sewage effluent in Queensland (Figure 4). The contribution is estimated at 0.16% of the anthropogenic N input (Figure 5) and 0.08% of the anthropogenic P input (Figure 6) into Queensland east coast rivers. Even if higher discharge values for prawn farm nutrients are used (e.g. 1.0 kg/ha^{-1}/day^{-1} of N) the percentage contribution from prawn farming to total nutrient input remains low. For instance, increasing the level of N used in calculations by 0.1 kg/ha^{-1}/day^{-1} only increases prawn farm inputs (with respect to total anthropogenic inputs) of N by approximately 0.03%.

Although effluent from prawn farms and sewage effluent are anthropogenic point sources of nutrients to marine systems, sewage effluent differs from prawn farm effluent in four important ways. First, sewage effluent can contain bacteria and viruses potentially harmful to humans. Second, sewage effluent contains a range of chemicals that may have ecological impacts on biota which are currently poorly understood. Third, sewage effluent can contain a range of oils and greases that may have negative ecological or human health impacts. Fourth, sewage discharge is more or less continuous, whereas discharge from prawn farming is intermittent which potentially allows the system to recovery from any impacts from prawn farm discharges (Trott & Alongi, 2000).

Neil and Yu (1996) calculate that in Great Barrier Reef catchments, the total sediment yield is 28 Mt/yr^{-1} which is about 3.8 times that prior to European settlement. The vast majority of this sediment input is from grazing lands. Total suspended solids (TSS) entering the marine system in Queensland from prawn farming is calculated to be 1,314 t/yr^{-1}. Thus, TSS discharged annually from prawn farming in Queensland represents approximately 0.006% of the estimated annual anthropogenic total sediment input since European settlement to the Great Barrier Reef region.

A comparison of Queensland and overseas prawn farming-environmental and social impacts.

Prawn farming in Queensland (and Australia) and it's environmental management differs from that overseas in several ways including: community expectations, spatial scale, the regulatory environment, and farming practices. Community expectations regarding the environmental management of prawn aquaculture in Australia are high. This stems from a generally high (on a world scale) awareness of environmental issues in Australia and the fact that currently most prawn farms in Australia discharge into the World

Heritage listed Great Barrier Reef Marine Park or other sensitive marine habitats (e.g. Hervey Bay).

Prawn aquaculture in Australia occupies an area of approximately 500 hectares. For comparison, prawn aquaculture occupies approximately 350,000 hectares in Indonesia, 100,000 hectares in Ecuador, 80,000 hectares in Thailand and 60,000 hectares in the Philippines (Rosenberry, 2000). Prawn farming was regulated in Australia from its inception and is currently regulated at the local, State and Commonwealth level. Regulations address the potential environmental and social impacts of a proposed prawn aquaculture development and these regulations are enforced. Overseas, prawn farming is generally poorly regulated and if regulations exist they are generally poorly enforced (see Ong, 1982; Aiken, 1990; Chua, 1993; Parks & Bonifaz, 1994).

Prawn aquaculture in many countries still relies on the collection of post-larvae from the wild, whereas the Australian industry relies on hatchery produced post larvae from wild caught spawners for *Penaeus monodon* and domesticated stock for *Penaeus japonicus* and *Penaeus merguiensis* (N.Preston, pers. comm.). The collection of penaeid postlarvae for prawn aquaculture overseas has not generally been managed and the harvest is identified as a cause (along with wetland and mangrove destruction) of declining wild penaeid populations in some countries (Landesmann, 1994). Overseas, the overuse of antibiotics and other chemicals is a significant environmental issue. Antibiotics are not able to be used in prawn grow out ponds in Australia and chemical use, in general, is regulated.

Destruction of marine plants (e.g. mangroves) for pond construction in Australia is prohibited and pond construction is also prohibited in intertidal areas and sensitive habitats (e.g. ephemeral wetlands). In countries where clearing of mangroves and wetlands for prawn pond construction is a significant environmental issue, basic legislation to protect these habitats is generally lacking, or if present not effectively enforced (see Ong, 1982; Aiken, 1990; Chua, 1993; Parks & Bonifaz, 1994). Prawn ponds overseas have frequently become disused as ponds become unviable due to poor farm management and farmers move onto new ponds without rehabilitating their previous ones. Such a scenario is highly unlikely to arise in Queensland given that if a farmer has already been unsuccessful due to poor farm management and has not adequately rehabilitated his previous farm, he would be unlikely to obtain the licensing necessary to develop a new farm. Furthermore, there is industry recognition that mangrove areas are generally poor sites for prawn farming (Preston *et al.* 2001).

In terms of social impacts from prawn farming in Australia, there has been no detailed study. However, there are four (interrelated) reasons why negative social impacts from prawn farming in Australia are likely to be less than those reported from overseas: 1) Australia is not a developing country, some of the social conflict overseas is because of existing and entrenched social and economic inequities that prawn farming may have exacerbated in some instances but not necessarily caused. 2) Prawn farming in Australia was regulated from its inception, is still heavily regulated, and regulations are enforced. This is related to the first point but is sufficiently important to be mentioned separately. It has allowed for the development of aquaculture to be planned and for local communities to have input into planning procedures 3) Large areas of wetland are not destroyed for pond construction and the associated disenfranchisement of local communities from community resources is unlikely to be significant. 4) Prawn farming in Australia has not displaced practices that produced the bulk of the food for local communities.

Conclusions.

Compared to other anthropogenic sources, inputs to the aquatic environment of N and P from prawn farming in Australia are small and treatable. Prawn farming contributes just 0.16 % of the anthropogenic N input and 0.08 % of the anthropogenic P input into Queensland east coast rivers. Most anthropogenic inputs of N and P into Queensland east coast rivers are from N and P associated with sediment loss from grazing. Total sediment yield to the Great Barrier Reef has increased about 3.8 times since European settlement and this increase represents approximately 21 Mt/yr^{-1} of anthropogenic sediment input to the Great Barrier Reef region. TSS discharged from Queensland prawn farms is calculated to be 1,314 t/yr^{-1} which represents just 0.006 % of the total annual anthropogenic sediment input to the Great Barrier Reef region. A wide range of chemicals (herbicides and pesticides) are applied to agricultural land and enter the aquatic system. Many of these chemicals (or their residues) are now found in animal tissues, although the long-term ecological effects of these chemicals on aquatic biota are poorly understood.

Sewage effluent and its potential impacts differs from prawn farm effluent in that it potentially contains: bacteria and viruses harmful to humans and other organisms, oils, greases and detergents, and a wide range of chemicals and heavy metals. Most sewage treatment plants in Queensland treat sewage to a secondary level which does not remove N and P. Sewage treatment plants are estimated to discharge approximately 1,928 t/yr^{-1} of N and 798 t/yr^{-1} of P into Queensland waters.

Prawn farming initially overseas has caused environmental damage and social conflict. Prawn farming in Australia differs from prawn farming overseas in several ways, not the least of which is that it is regulated and that regulations are enforced. Large amounts of mangrove forests and wetlands have been destroyed for pond construction in Asia and South America, however, removing mangroves for pond construction is prohibited in Australia. Prawn farming in Australia does not rely on the wild capture of post larvae or juvenile prawns. Chemical usage in Australian prawn farming is regulated and antibiotics are not permitted in prawn grow-out ponds, however, limited amounts are used in hatcheries. Although not yet studied in detail, it is possible that the negative social impacts reported from prawn farming overseas may not occur because of the regulatory environment and different farming practices.

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Table 1. Annual Discharge Volume and Estimates of Nutrient (N and P) Inputs from Sewage Discharge in East Coast Queensland Catchments.

Region	DischargeVolume (ML/yr^{-1})	Total N (t/yr^{-1})	Total P (t/yr^{-1})
Southport *	32,525	279	186
Brisbane *	117,000	1,100	350
Sunshine Coast	48,950	195	97
Mary	9,563	38	19
Burnett- Kolan	10,320	41	10
Curtis Coast	4,380	17	9
Fitzroy	17,030	68	34
Pioneer- O'Connell	7,701	30	15
Proserpine	2,266	9	4
Don	657	2	1
Burdekin- Haughton	2,701	10	5
Ross- Black	15,163	60	30
Herbert	2,018	8	4
Tully- Murray	584	2	1
Johnstone	2,409	9	4
Mulgrave- Russell	1,945	7	3
Barron	13,632	53	26
TOTAL	288,844	1,928	798

NOTE. Estimates of discharge volume were calculated from Zann (1995) and N and P discharge by assuming discharge concentrations of 4.0 mg/L^{-1} N mg/L^{-1} and 2.0 mg/L^{-1} P. Exceptions are (marked with *) where estimates of N and P input were found in the literature.

Figures