RISK ASSESSMENT SCREENING FOR POTENTIALLY INVASIVE FRESHWATER FISHES WITHIN THE WET TROPICS BIOREGION: A REVIEW OF ASSESSMENT APPROACHES, IDENTIFICATION OF KNOWLEDGE GAPS AND FUTURE RECOMMENDATIONS

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Prepared by

Dr. Alan C. Webb,
Australian Centre for Tropical Freshwater Research,
James Cook University,
Townsville, Qld 4811
Phone 07-47814262
Email: actfr@jcu.edu.au
Website www.actfr.jcu.edu.au

1.0 INTRODUCTION

Invasive species, including fishes, are now recognised as a major threat to global biodiversity (Witte *et al*. 1992; Cambray 2003; Njiru *et al*. 2005; Dudgeon *et al.* 2005) and can cause significant economic damage (Perrings *et al.* 2000; Pimental 2002; Cox & Kitchell 2004). Ecological risk assessment (ERA) is the process of predicting or estimating the likelihood and magnitude of adverse ecological effects occurring as a result of one or more threats (stressors) caused by human activities to species (including people), natural communities or ecosystem processes.

The framework for ERA now being implemented by government agencies worldwide is broadly similar and consists of three major phases: problem formulation, analysis, and risk characterisation. ERA applications, however, are numerous and include assessments that range from: screening-level (qualitative) to detailed (quantitative) assessments or a combination of both; predictive to retrospective or local to global in scale, and single threat to multiple threats (US EPA 1992; Burgman 2005; van Dam 2006). While many screening tools were initially developed for risk assessment of plants as weeds (see Pheloung *et al*. 1999; Groves *et al*. 2001; Kolar 2004) there has been increasing application of risk assessment protocols to animals including: insects (van Lenteren *et al.* 2003; Peterson *et al.* 2004; Allen *et al*. 2006); molluscs (Drake & Bossenbroek 2004); mammals (Forsyth *et al*. 2004), birds (Bomford & Sinclair 2002) and fish (Townsend and Winterbourn 1992; Kolar and Lodge 2002; Bomford & Glover 2004; Copp *et al.* 2005). The objective of these ERA processes is to identify high risk species with minimum uncertainty, at one or more stages of the invasion process (establishment, spread and impact) (US, NSTC 1999; Kolar and Lodge 2002). Studies have also focussed on improving the predictive power of screening tools by development of extensive ecological and climatic databases and analytical refinement of species profiling (Peterson and Cohoon 1999; Kolar and Lodge 2002; Anderson *et al*. 2003; Marchetti *et al.* 2004).

The Wet Tropics Bioregion extending between Townsville and Cooktown, northern Queensland, has a very high diversity of fishes and parallels that observed for terrestrial vertebrates. At least 70 species have been reported, approximately 45 % of Australia's freshwater fish fauna, with a very high level of endemism (at least nine species) (Rainforest CRC 2001). Besides natural disturbance (eg. cyclones), the region is experiencing increasing anthropogenic disturbance (Sattler and Williams 1999) with urban and agricultural development impacting on the integrity of waterways and their resident communities (Norris *et al*. 2001; Commonwealth of Australia 2002). The Conservation Strategy of the Wet Tropics Management Authority (WTMA 1998) listed as a priority the prevention of new non-native species (i.e., species originating from beyond continental Australia), including freshwater fishes from establishing in the region. While there are currently only six non-native fishes reported from the Wet Tropics Bioregion, research indicates that such regions of high endemism are susceptible to invasion (Strahm 1999; Clout 1999; Ruesink 2005) and habitat disturbance can facilitate the invasion and survivorship of non-native fishes (Arthington *et al*. 1983; Moyle and Light 1996; Webb 2003). There are at least 13 other non-native fish species reported from areas adjacent to the Wet Tropics Bioregion. The current trend suggests that these introductions may increase and at a faster rate in the future. Between 1994 and 2006, introductions in tropical northern Queensland increased from 10 to 19 species (90%). This latest estimate represents 50% of all species (38) reported from Australian waters, with the Ross River, Townsville, having the highest number of non-native fish introductions (15) reported for any waterway in Australia (Webb 2003 and unpubl. data).

There is also a very large "latent" pool of non-native fishes comprising those species present in home aquaria (and garden ponds). McNee (2002) identified at least 1181 non-native species that have been present in Australia over the past 40 years, the vast majority of which are tropical species, although the number held in captivity in northern Queensland is not known. Only 481 species are on the current permitted import list and other species may be present as the result of illegal imports (Kailola 2002), which are estimated to comprise 5-10 % of fish imported into Australia (AQIS 1999). The majority of these captive species have not been subject to any risk analysis. Burrows (2004) also reported at least 36 native freshwater fishes that have been translocated within the Wet Tropics region and suggested that these fishes can have similar impacts as non-native fishes in waterways where they are stocked but are not indigenous i.e., did not previously occur. None of these species were subject to any comprehensive risk assessment process. Given the number of non-native fishes that are currently present or may be introduced in the near future, and the limited resources available to deal with them, it is imperative that these resources are strategically allocated to those species most likely to become invasive and create negative environmental, economic and social impacts. The process to identify these potential invaders is best done through risk assessment.

Risk assessment for non-native fishes in Australian waters is still in an early phase. The development of an effective assessment process for non-native fishes is of increasing urgency. This document reviews ERA models and screening tools applied to introduced, non-native fishes, identifies knowledge gaps and recommends future research priorities to further develop risk assessment processes that are applicable to tropical northern Queensland, including the Wet Tropics Bioregion.

2.0 RISK ASSESSMENT MODELS FOR NON-NATIVE FRESHWATER FISHES: A REVIEW

There are two basic elements in risk assessment: hazard identification and hazard assessment. The first element typically involves qualitative or quantitative analysis of ecological and biological characteristics of existing non-native species in a specific region to identify those attributes that will reliably predict future invasive species from the same donor region (e.g., Kolar and Lodge 2002; Marchetti *et al*. 2004) and can be used as an import screening mechanism to detect and prevent entry of high risk species into a country or region (e.g., Arthington *et al.* 1999; Bomford and Glover 2002). The second element (hazard assessment) provides for consideration of the environmental, social and economic values of a specific region in relation to potential threats posed by one or more target species. Using semi-quantitative or qualitative decision support systems, species are then ranked according to the nature and magnitude of the threat (eg. Clunie *et al*. 2002; Harrison and Congdon 2002). Target species may be present, in areas adjacent to the specified region, or recently detected within it. This phase enables managers prioritise actions including allocation of resources to implement control measures for rapid and effective threat abatement (Harrison and Congdon 2002; Clunie *et al.* 2002; Copp *et al.* 2005).

2.1 Hazard Identification

There have been two broad approaches to hazard identification: the first based on life history characteristics and environmental tolerances of previous invaders, and the second based on ecological niche modelling (climate pattern matching) of potentially invasive species (Kriticos and Randall 2001; Kolar 2004). Besides biological traits (e.g., maximum size and longevity, growth rates, fecundity, parental care, diet breadth, physiological tolerances), the former approach may include stochastic data identified from invasion ecology (e.g., propagule pressure – number of individuals introduced, frequency of introductions and residence time) and extrapolative data (invasive history – i.e., previously documented invasions, and human use) (Rejmanek 2001). Models based on species biological characteristics assume invading species differ from species that tend not to invade and therefore the identity of potential invaders can be predicted. A number of studies have developed models based on comparison of two sets of species (successful and unsuccessful invaders) in a given region (Marchetti *et al.* 2004; Alcarez *et al.* 2005; Vila-Gispert *et al*. 2005), at a given stage in the invasion process (establishment, spread, impact) (Kolar and Lodge 2002; Bomford and Glover 2002), or at a continental or global level (e.g., Ruesink 2005; Jeschke and Strayer 2006). Procedures such as General Rule-set Predictions (GARP), Principal Components Analysis (PCA), Regression Tree Analysis (CART) are then used to identify those attributes which are the strongest, most reliable predictors of invasion success (Kolar and Lodge 2002; Marchetti *et al*. 2004).

There have been numerous attempts to correlate climate with spatial distribution of target species or communities (Kuchler 1967; Box 1981), although these ‘manual’ methods were time consuming until the advent of computer software packages (e.g., CLIMATE, BIOCLIM, CLIMEX, DOMAIN) that automated the process (Sutherst and Maywold

1985; Worner 1988; Kriticos and Randall 2001), to provide a spatial representation of habitat suitability (climatic mapping) of a species in new areas based on their responses to climate in their home range (Baker *et al*. 2000). Climate mapping has been used to predict potential distribution of exotic organisms, predominantly invertebrate poikilotherms including insects (Sutherst *et al.* 1995, parasites (Coakley *et al*. 1999; Sutherst 2001), platyhelminths (Boag *et al*. 1995), but also some vertebrates, including homoeotherms (birds) (Duncan *et al.* 2001) and poikilotherms: amphibians (Sutherst *et al.* 1996) and fish (Bomford and Glover 2002).

Semi-quantitative or qualitative hazard identification approaches, rather than identify specific attributes of successful invaders, use a decision support system consisting of a series of questions that are selected on the basis of expert evaluation of published literature on the target species. Data on target species are incorporated as categorical variables, usually with weighted values associated with each category, and species or taxa are then ranked by expert evaluation as high to low or uncertain risk on the basis of combined responses to each question. These data may be biological or ecological characteristics (Kailola 2000; Copp *et al.* 2005) or taxonomic (Arthington *et al*. 1999).

Kailola (2000) subjectively ranked exotic fish families present in Australia according to level of risk they posed to native fish and the environment. Risk of establishment was evaluated on the basis of various biological traits (e.g., fecundity, spawning frequency, tolerances to environmental extremes (e.g. high temperature and salinity, low pH and oxygen concentration), genotypic and phenotypic variability, diet breadth, levels of parental care), while potential impacts were based largely on anecdotal evidence of impacts of species in introduced ranges elsewhere. Poeciliids and cyprinids were identified as the highest risk taxa, followed by salmonids, percids, cichlids (moderate risk), cobitids and belontiids (least risk). Copp *et al.* (2005) developed a Fish Invasiveness Screening Kit (FISK) consisting of 2 phases: Hazard Identification (Phase 1) and Hazard Assessment (Phase 2) This kit was based on a Weed Risk Assessment model devised by Pheloung *et al.* (1999) and used similar traits for hazard identification of non-native fishes in Britain: i) biogeography and history of species, ii) the presence of "undesirable traits" and species biology and ecology of non-native fish species in Britain. Within each of these major categories questions (categorical variables), with weighted scores for each response, were further grouped: *Biogeography and History*: domestication, climate and distribution, and invasive history; *Biology and Ecology*: undesirable traits, feeding guild, reproduction, dispersal mechanisms, tolerance attributes. Responses by experts are then totalled and negative scores indicate 'low risk' while positive scores indicate 'high risk' species. Scores could also be allocated to topic subcategories (Aquaculture, Environment and Nuisance) to indicate which 'sector' is most likely to be affected by each species.

Arthington *et al.* (1999) provided a report towards the development of an Ornamental Finfish Import Risk Assessment scheme by AQIS (Kahn *et al*. 1999). The focus of the AQIS report was on the ecological and economic risk of pathogens entering Australia with host fish, the contributory report assessed the probability of non-indigenous fish surviving and establishing self-maintaining populations in Australian waters. A series of qualitative decision "filters" were used to evaluate establishment probability of fish according to records of their establishment of populations outside their natural range, including Australia, and their taxonomic affinities (at the species, generic and family level) (see Table 1):

The first four decision filters were applied to species listed in Schedule 6 of permitted fish imports into Australia while the last filter (residual risk) was applied to and categorised all other fish not categorised.

All of these approaches rely on the generally accepted premise that 'pest' species (i.e. that have negative environmental or socio-economic impacts) in other parts of the world have an increased chance of being invasive in other areas with similar environmental conditions and/or have close taxonomic affinities.

Table 1 Decision filters to identify establishment risk of exotic fishes (from Arthington *et al.* 1999)

Filter	Establishment Probability	Species Category
1	Very high	Species previously present of unknown status, present but not established; established
2	high	Species in same genus as those with successful establishment in Australia or overseas
3	Moderate to high	Species in same family as those with very high or high probability of establishment
4	moderate	Species in families, other than those with established representatives in Australia, with a history of establishment overseas
5	residual	All species not listed on Schedule 6, unless recorded in Australian waters or held in captivity. Species from families with no prior history of introduction or establishment overseas or in Australia

2.2 Hazard Assessment

Williamson (2001) argued that impact of invasions cannot, in general, be predicted due to a variety of factors including a lack of comprehensive information on the biology and ecology of invaders, on residence time and rate of spread of invaders (and therefore interaction time with native species), uncertainties in fish systematics (affecting phylogenetically independent statistical analyses) and stochastic effects such as complexity and non-linearity in population dynamics on outcomes. These outcomes and consequent impacts can be many and varied depending on the strength of the interactions between species. However, Williamson emphasised that, in spite of these difficulties, impacts can still be subject to risk assessment. The hazard assessment approach focuses on environmental, social and economic impacts of target species, and relies on various forms of semi-quantitative or qualitative decision support systems (e.g., Harrison and Congdon 2002; Copp *et al*. 2005). The approach is more subjective and reliant on expert opinion because of the greater difficulty in quantifying impacts of a target species compared with measurement of its ecological or biological attributes.

Phase 2 of the FISK protocol (Copp *et al.* 2005) provides a similar process for hazard assessment as described in the previous section for hazard identification (Phase 1). Questions are grouped into categories and sub-categories: Risk of Introduction (Deliberate, Unintentional) and Establishment (climate matching with donor region), and Impact (economic, environmental, social, dispersal and spread). Total risk is provided by combined scores for all questions. The number of unanswered questions as a percentage of the total number of questions is used as a measure of 'uncertainty' regarding the assessment.

Clunie *et al.* (2002) proposed a hazard assessment procedure for non-native fishes based on that used for potential pest plants from aquatic and riparian habitats in the Murray-Darling Basin. The Decision Support System (DSS) devised for assessing aquatic plants was an analytical, hierarchical process (AHP) based on a generic decision making model developed by Saaty (1995) and used criteria derived from New Zealand's Aquatic Weeds Prioritisation System (see AS/NZS 2004). The DSS assigns numerical values to subjective judgement by experts regarding the relative importance of each variable, and synthesises judgements to determine which variables have the highest priority. While the process appeared to have high potential during a workshop trial, it became apparent to Clunie *et al*. (2002) that further research and collation of expert opinions was required as there was insufficient data on the life history attributes and ecological and socio-economic impacts of all non-native fishes in the MDBC to effectively evaluate emerging and new pest species. To date, an operational risk assessment process for non-native fishes in the Murray Darling Basin has not been developed (M. Lintermans, MDBC, *pers. comm.*).

Harrison and Congdon (2002) developed the WTVPRAS to establish the relative pest status and potential impact of exotic vertebrates, including fishes, within the Wet Tropics Bioregion. They included assessment of six exotic fish species (Gambusia, Guppy, Platy, Swordtail, Mozambique tilapia and Spotted tilapia) listed as occurring in the Wet Tropics Region and adjacent areas. They provided a series of questions based on five 'generic' pest species characteristics obtained from the literature (previous pest history, reproductive and dispersal potential, ability to capitalise on variation in climatic and/or biological events, ability to act as vectors of diseases or parasites, potential to threaten existing species via predation, competition and/or habitat degradation. Weighted scores for answers to questions (categorical variables) were used to generate four 'impact indices' (current impact, future impact, feasibility of control, and detrimental impacts of control) for each pest species. These indices could then be used separately or in combination to assess the species' pest status and inform management options.

Species were provided with a descriptive risk summary and ranked in one of four categories:

1. High impact/high potential impact & difficult to control
2. ‘Sleepers’ – low to moderate current impact but high future impacts
3. The Moderates
4. Low impact and low to moderate impacts

The six exotic fish species were all ranked as major pest species (‘Sleepers’) and considered they “may constitute the principal unrealised threat to the region” (p.29). Harrison and Congdon also recognised the major problem in providing an effective assessment process was the lack of ecological data on non-native (and native fishes).

2.3 General Summary with Particular Reference to Australian Risk Assessment Protocols

Risk assessment of non-native fishes is a relatively recent process, both overseas and in Australia and there is still no agreed approach, at least in application of assessment methodologies. Ruesink (2005) observed that, beyond human-mediated propagule pressure, which can be difficult to quantify, species’ traits dominate invasion and hazard risk and are more amenable to measurement. Conservation effort should therefore be expended to develop quantitative, ecologically informed screening protocols using traits that can identify successful invaders. Studies indicate that, for non-native fishes, attributes which were best predictors of establishment success varied between regions, and those which were best predictors of success varied between stages of the invasion process (establishment, spread, impact). (Kolar and Lodge 2002; Marchetti *et al*. 2004; Copp *et al*. 2005). Such studies were conducted in cold and warm temperate regions of North America and Europe and no similar study has been done for non-native fishes in tropical regions including Australia. Furthermore, according to Ruesink (2005), as many introduction ‘decisions’ are ultimately made at smaller scales, regional analyses may be more successful at categorising high and low risk species than national or global analyses.

According to Chong and Whittington (2005), while there have been many reviews related to the importation of ornamental fishes into Australia, the country’s biosecurity objective is currently not met with respect to such fish and that current protocols and procedures provided a relatively low level of protection. One of the recommendations proposed by Chong and Whittington was to “review the list of permitted species to reduce risk of pest and disease introduction”, i.e., current import screening procedures for risk assessment of non-natives fishes at a national level were considered inadequate. There is no quantitative, regional risk assessment specifically designed for non-native fishes in Australia. The model proposed by Arthington *et al.* (1999) was initially devised to support development of AQIS import risk protocols for ornamental fish, with a focus on prevention of entry of disease agents. It is essentially a qualitative hazard identification approach based on taxonomic association of a target species but without reference to the species’ biological attributes and does not address hazard risk assessment. The model proposed by Bomford and Glover (2002), used a similar methodology to Kolar and

Lodge (2002) to compare attributes of successful and unsuccessful non-native fishes. Bomford and Glover's model is concerned only with hazard identification and lacks a hazard assessment stage. Furthermore, it is designed only to identify attributes of species which successfully establish, and not those that become high risk, pest species. This model, however, could be improved by addition and testing of additional continuous variables (quantitative biological and ecological attributes) to identify both successfully establishing and nuisance non-native fishes, and could be used within a regional assessment framework. The approach proposed by Clunie *et al.* (2002) is regionally-based and uses a hierarchical, decision support system to rank the level of risk posed by non-native fishes. According to Clunie *et al*. (2002), this approach, initially devised for weedy terrestrial plants, (and similar to the model devised by Copp *et al*. (2004) for non-native fishes in Great Britain), was successful for use with aquatic plants in the Murray-Darling Basin. With modification of the criteria used, these researchers suggested that it could be regionally applied to non-native fishes. Although such an application has not yet been developed (due to lack of ecological data), this methodology is worth further investigation for application to tropical northern Queensland.

While the pest risk assessment scheme devised by Harrison and Congdon (2002) (WTVPRAS) is for the Wet Tropics, there are limitations with respect to assessment of non-native freshwater fishes. It is a 'generic' model and its primary focus is on terrestrial "pest" vertebrates. The assessment is a comparative ranking system between the identified "pest" vertebrates rather than being specifically designed for hazard assessment of fish species. Only six non-native fishes were tested and were designated *a priori* as pest fishes, without reference to any formal hazard identification process, other than to 'expert' opinion. All taxa assessed, including fish, were already present within the Wet Tropics Bioregion and no non-native fishes in adjacent areas were included to test the effectiveness of the procedure to differentiate high and low risk fish species. The system overly relies on 'expert concensus' (Ehrlich 1986), and on the assumption that there are 'generic' characteristics that reliably describe and are common to 'pest' organisms across a wide range of taxa and environments. Recent studies, however, have shown that attributes which best define invasive or 'pest' fish species can differ between regions (Kolar and Lodge 2002; Marchetti *et al.* 2004) and that also can differ from 'generic' characteristics identified for these fishes based on analysis of global fish introductions (Ruesink 2005). Ehrlich's (1986) comments are still relevant in this respect for the need to develop better predictive tools, that include "mathematical models for the behaviour of invaders in some groups (and most successful models will almost certainly be group-specific)."

A major problem concerning development of screening protocols for non-native fishes in Australia is the lack of basic ecological information on life history traits and impacts (Clunie *et al*. 2002; Harris and Congdon 2002; Bomford and Glover 2004). Howard and Congdon (2002) noted that their WTVPRAS using categorical data had to rely on the precautionary principle (for example weighting unknown impact as 'moderate' impact value) where information is limited. While precaution is a prudent approach in risk assessment, over- or under-estimating potential impacts may severely restrict the utility of some assessments and adversely affect resource allocation processes and the subsequent success or otherwise of control measures. The authors noted that this problem was particularly relevant to assessment of exotic fish species in the Wet Tropics Bioregion.

To date, many risk assessment models have used categorical or ordinal measures over continuous measures because of the lack of reliable continuous quantitative data for many species. Assessment requires Kolar (2004) noted that neither qualitative nor quantitative models are error free. However, the benefit of quantitative models is "increased transparency, reliability, and tractability, and decreased opportunity for subjective influence". Kolar and Lodge (2002), demonstrated that, with sufficient quantitative data (on life history characteristics, habitat needs and aspects of invasion history and human use), it was possible to identify a small number (4) of these characteristics which could be used to classify high and low risk non-native fishes for each stage of the invasion process (establishment, spread and impact) with a very high level of accuracy (87-94%). According to *Copp et al*. (2005), a more objective evaluation of invasive potential aids in whether assessment of a target species should continue into a more "comprehensive (and relatively expensive) phase or whether it ceases (and the species placed on a list for monitoring of any change in status)". The greatest benefit using quantitative approaches is they can provide managers and policy makers with more information than reliance on qualitative assessment without knowledge of probable error (Kolar 2004). Uncertainty analysis therefore describes the degree of confidence in the assessment and can help risk managers focus on future research on areas that will lead to the greatest reduction in uncertainty.

3.0 KNOWLEDGE GAPS

There are a number of problems associated with assessment of current and future impacts of invasive non-native fishes in tropical northern Queensland:

- a lack of quantitative biological and ecological data including life history traits, physiological tolerances and impacts on native aquatic communities (and those overseas) to evaluate or develop models of invasion success that define attributes or allow confident discrimination between high and low risk species;
- limited assessment of future climate change predictions on invasive potential; and
- little or no assessment of social and economic impacts on indigenous and non-indigenous communities

3.1 Lack of Basic Biological and Ecological Information on Non-Native Fishes

Any fish, wether exotic or native, when moved into waterways where it did not previously occur may cause significant changes, including local declines or extinction of naïve species by predation or competitive interactions, and genetic impacts through hybridisation (Barlow *et al.* 1987; Burrows 2004; Pusey *et al*. 2005).

Detrimental effects of non-native species can often be difficult to characterise or quantify due to a variety of factors, such as complicating effects of multiple invasions or anthropogenic habitat disturbance, spatial and temporal variations in impacts and complexity of community dynamics (Park 2004; Bomford and Glover 2004). Williamson (2001) warned that impacts of a species in one region cannot necessarily be predicted from impacts in another region, although the frequency and magnitude of adverse impacts elsewhere following introduction can provide some qualified assessment or probability of a species becoming a pest. There is limited ecological information (eg. life history traits, ecological tolerances, rate of spread, potential or realised impacts) available on many non-native fishes present, established or held in captivity within Australia. Also, the majority of the 1181 non-native fish species reported to be held in captivity in Australia (McNee 2002) have not been subjected to any hazard identification process. Burrows (2004) reviewed some of the known adverse impacts of translocated native fish in the Wet Tropics, though information on the ecology and impacts of many of these species needs still to be determined. None of these species were subject to a comprehensive risk assessment prior to release.

There have been very few studies using quantitative data that can be statistically tested to identify attributes that differentiate high-risk ‘nuisance’ (pest) fish from lower-risk ‘non-nuisance’ fishes. For non-native fishes in the Great Lakes region, North America, Kolar and Lodge (2002) compared the attributes of rapidly spreading fish with slow spreading fish, and those of fish considered by experts as ‘nuisance’ and those that were not. Quickly spreading fish had slower growth rates, survived poorly in high water temperatures and tolerated a wider temperature range (discriminant function 94% accurate), while nuisance fish had smaller eggs, wider salinity tolerances and survived in lower water temperatures (discriminant function 89 % accurate). A similar analysis has not been performed for non-native fishes at either the national or regional level in Australia. While it has been argued that collection of such data is time consuming and expensive, these costs are likely to be trivial compared with the ecological or economic costs potentially associated with mis-identified invaders.

Depending on the suite of variables tested or analysis used, results of studies have varied as to the number and type of variables that provide the strongest predictions of invasion success. These differences may also be region-specific, reflecting historical differences in invasion process, species involved, habitat-species interactions (e.g., pathways of invasion and spread) and propagule pressure (human use). Kolar and Lodge (2002) using discriminant analysis found four key characteristics for establishment success: rapid growth, wide salinity and temperature tolerances, and previous history of establishment (discriminant function), while CART analysis indicated that minimum temperature threshold, diet breadth and relative growth rate provided the strongest predictors. Alcaraz *et al.* (2005) obtained different results for analysis of qualitative and quantitative variables between non-native invasive fishes and native fishes on the Iberian Peninsula. For qualitative variables using univariate analysis, taxonomy (order, family), human use, reproductive guilds and habitat use differed significantly. For quantitative variables, reproductive season span and latitude range had significant differences in variance. These researchers also argued that phylogenetic effects need to be controlled as treating closely related non-native species as independent data may violate assumptions underlying most statistical tests. These researchers found that the taxonomic distribution of invasive species deviated significantly from world freshwater richness. These fishes belonged to a restricted number of taxonomic orders but to a wide spectrum of families not native to the Iberian Peninsula, with some taxa over-represented – i.e., the taxonomic distribution was not random which could affect comparison of life-history traits between native and invasive species. PCA analysis failed to clearly separate the two groups of fishes until the phylogenetic effect was eliminated which showed significant difference in variability of timing of reproductive season and measures of geographic range (mean latitude and latitudinal range). Ruesink (2005), for a global analysis, found small body size omnivorous diet, high endemism in recipient country, and human use were the strongest predictors of establishment success. Marchetti *et al*. (2004) found measures of prior invasion success and propagule pressure were the strongest predictors for Californian non-native fishes along with parental care, physiological tolerance and size of native range, while Jeschke and Strayer (2006) (North America and Europe) found measures of propagule pressure and human use to be the strongest predictors.

For non-native fishes in the Great Lakes region, rapidly spreading species were best characterised by slow growth rate, low upper temperature tolerance but wide temperature tolerance range (Kolar and Lodge 2002), while for continental North America and Europe the strongest predictors were propagule pressure and human use (Jeschke and Strayer 2006). In contrast, Marchetti *et al.* (2004) found that being longer-lived, of regional origin and not a herbivore conferred an advantage during this phase for non-native fishes in Californian waters. Kolar and Lodge (2002) also found that nuisance (pest) fish in the Great Lakes region were best characterised by smaller egg size, wider salinity tolerance and low temperature threshold compared with non-invasive species.

No similar quantitative analyses using biological traits of non-native fishes have been conducted for tropical regions either in Australia or overseas. Since attributes identifying successful invaders appear to differ between regions, similar hazard identification models need to be tested using attributes of non-natives fishes in tropical northern Queensland. The only quantitative hazard identification model developed to date for establishment success of exotic fish in Australia (Bomford and Glover 2004) used a similar methodology as that used by Kolar and Lodge (2002) for non-native fishes in the Great Lakes region, but did not use biological traits. Their analysis had relatively low prediction accuracy for classifying successful invaders. Using PCA and CART analyses to compare scores of established species and those present but not established, these researchers obtained an accuracy of 63.3% and 77.6% respectively. Kolar and Lodge (2002), with a more comprehensive set of quantitative ecological data obtained much greater predictive accuracy. A discriminant analysis using four key characteristics (rapid growth, wide salinity and temperature tolerances and previous history of establishment) discriminated between failed and successfully established fish with 87% accuracy. CART analysis, using minimum temperature threshold, diet breadth, and two measures of relative growth, classified failed and successfully established fishes with 94% accuracy (only 2 out of 45 species misclassified). Differences may also reflect that Kolar and Lodge’s analysis was restricted to a single region, while that of Bomford and Glover was for non-native fishes throughout all of continental Australia.

Some data on ecological attributes and life history traits published in the literature may also be of limited usefulness for model testing due to a lack of standardisation in parameters and methodologies used to obtain them, and in their units of measurement. For example, lower thermal tolerances have been variously reported as critical thermal minima, incipient lethal temperature and ultimate lethal temperature (Elliott 1981) and cooling-degree days (Cnaani *et al*. 2000). These values are obtained by different methodologies and are not equivalent (one cannot be derived from any of the other values). Similarly, fish body measurements have been given as Standard length or as Total length and are not derivative unless regression equations are available. Non-equivalent measures of reproductive output (fecundity) also have been used and include brood size (Herbert and Graham 2004), total seasonal fecundity (seasonal brood numbers) (McDowaland & Eldon 1997), total or mean fecundity (number of oocytes/female), relative fecundity (number of oocytes/kg female bodyweight (Minto and Nolan 2006), absolute fecundity (number of oocytes produced over reproductive life of female) (Tarkan 2006).

This lack of quantitative information hampers further development and testing of models to identify variables (biological traits) that can be incorporated into a risk assessment framework for more accurate prediction of establishment success of non-native fishes and high risk (pest) species. At present, only about 54% of information needed to conduct a similar analysis to that done by Kolar and Lodge (2002) (see above) is readily available in the literature (including databases, such as Fishbase, journal articles, reports, theses, and texts) for the 28 species of non-native fishes reported in Queensland (see Table 1A, Appendix A). If only the small group of variables, identified by Kolar and Lodge (2002) as major determinants of establishment success or invasiveness of non-native fishes, apply to non-native fishes in northern Queensland, only 35% of the data for non-native fishes in Queensland is available for analysis. Furthermore, some variables may be more important in determining outcomes in the tropics than in higher latitudes, e.g., higher temperature thresholds and dissolved oxygen concentrations, and these, therefore, need to be assessed and measured.

3.2 Lack of Information on the Potential Impact of Climate Change on Establishment Success and Spread of Non-native Fishes

Freshwater ecosystems are very vulnerable to climate change due to global warming where changes in temperature, precipitation and run-off are likely to disrupt patterns of plant and animal distributions (Shuter and Meisner 1992; Allan *et al*. 2005). A number of studies have investigated the potential effects of climate change on freshwater fish communities (e.g., Chang *et al.* 1992; De Stasio *et al.* 1996; Jackson and Mandrak 2002; Shuter *et al*. 2002; Daufresne *et al.* 2004; Chu *et al*. 2005), although virtually all are for temperate northern hemisphere communities. Few studies have considered potential impacts of climate change on warmwater or tropical freshwater fishes (Meisner and Shuter 1992; Fang *et al.* 2004).

Krockenberger *et al*. (2003) outlined possible impacts to the Wet Tropics Bioregion due to global warming and sea-level rise which, even under the least extreme of scenarios, included deterioration in ecosystem function associated with elevated temperatures (between, on average, 1.4 and 5.8°C), extreme heat waves and more variable rainfall, including prolonged droughts and more frequent, intense cyclones. Wetlands, rivers and associated riparian systems in the region are at relatively high risk due to changes in water quality (increased turbidity, temperature and salinity), water flows (increased flooding) and availability (loss of dry season refugia) (Meyer *et al.* 1999). These changes will directly impact on fauna and flora with potentially a catastrophic loss of biodiversity, particularly of endemic species unable to adapt to changing conditions (Williams *et al.* 2003).

While the focus has been on impacts to terrestrial fauna, the Wet Tropics also has a large number of endemic freshwater fishes (Rainforest CRC 2001). A global analysis of freshwater fish introductions by Ruesink (2005) suggested that establishment success of non-native fishes increased with fish endemism of the recipient country, i.e., regions such as the Wet Tropics are particularly vulnerable to invasion by non-native fishes. Dukes (2003) predicted an increase in weedy or invasive species able to exploit opportunities opened up as a consequence of ecosystem disturbance associated with global warming. In the Wet Tropics, even in the absence of human translocation, natural dispersal of exotic fishes is likely to be facilitated by an increase in severe flooding events. Combined with a continued upward trend in release of aquarium fishes in northern Queensland (Webb 2003), this will increase the likelihood that some of these species will possess attributes that allow them to establish, spread rapidly and negatively interact with resident aquatic communities. Future climate change and its potential affect on the invasiveness of non-native fish species need therefore to be considered as a component of risk assessment, both at the national level and at the regional level such as the Wet Tropics Bioregion.

3.3 Lack of Social and Economic Impact Assessment of Non-Native Fish Species

According to a recent report on the regulation, control and management of invasive species in Australia (Commonwealth of Australia 2004), the economic impact of invasive species, while difficult to quantify, is very high. The cost of weeds and 11 key vertebrate pest animals has been calculated at $4 billion and $720 million per annum respectively (McLeod 2004). Only one fish species, the European carp, was included in the assessment. The annual economic cost for carp has been estimated at $15.8 million (approximately $4 million in annual research and management costs and $11.8 million in environmental costs) (Bomford and Hart 2002; McLeod 2004). This is in contrast to the USA where the problem of invasive species is on a greater scale and the estimated annual costs of management range from 100's of millions to trillions of US dollars (Simberloff 1996; Pimental *et al.* 1999; Berenbaum 2001). According to McNeely (2005), the annual cost in the USA associated with invasive fishes is US$1 billion, while Shine (2005) stated that the US Department of State spends more than US$10 million annually on management of lampreys in the Great Lakes.

As yet no detailed socio-economic impact assessment has been undertaken for other non-native fishes in Australia. The economic costs associated with the management of other invasive species, such as tilapia may be comparable, if not exceed, those for carp, since tilapia occupy a wider range of habitats other than fresh water (i.e., estuarine and marine). Impacts may include those that can be readily identified and assigned a monetary value, e.g., downturn in local economies, including ecotourism, recreational fishing with loss of amenity, revenue and employment opportunities. Other social impacts may be less easily quantified but important for effective wetland management. Community motivation to participate or cooperate with conservation strategies may be affected by: adverse perceptions of fisheries management either doing "too little", or intervening too late to resolve an important environmental problem, the use of control measures which conflict with community values or have adverse impacts on other aquatic flora and fauna (e.g., use of fish poisons) and frustration of recreational fishers prevented access to waters undergoing control measures (Henry and Lyle 2003; McCleod 2004).

Maintaining connection with traditional country continues to be fundamental in indigenous people's identity and well-being. As part of this, fisheries resources are of critical importance to indigenous communities, not only as food but also for purposes of culture, spirituality, trade, health and education. This connection has been documented for several regions in Australia (Rowland 1989,2004; Morgan and Sefton 2005). At a recent Wet Tropics Regional Environmental and Natural Resource Management workshop, Traditional Owners emphasised that clean and healthy waterways are essential for the maintenance of cultural resources (including fish) and practices. They also expressed concern about the impacts of introduced fish species (e.g., tilapia) and how this impacts other important native species. They noted that there has been limited documentation of relevant cultural knowledge and the value of such resources to indigenous communities within the Wet Tropics region (Johnson *et al.* 2003).

The cost of invasive species to the environment and biodiversity – and therefore to the cultural heritage of indigenous and non-indigenous Australians is virtually incalculable. Such non-monetary 'costs', however, still need to be identified and incorporated into the impact assessment framework to assist in evaluating threats and allocation of often-scarce resources (money and personnel) to prevent or eradicate new introductions, and maximize strategic use of control measures.

4.0 RECOMMENDATIONS

Key recommendations for further research on risk assessment of non-native fishes in northern Queensland, including the Wet Tropics Bioregion are:

4.1. the collection of basic ecological data from laboratory and field studies using standardised protocols to fill identified gaps in the literature (see Table 1A, Appendix A), including life history traits and physiological tolerances (T, S, DO_2);

4.2. development of a broad assessment framework consisting of:

a) a quantitative hazard identification process based on a similar model to that used by Kolar and Lodge (2002) and Bomford and Glover (2002) to identify

attributes that maximise predictive power to discriminate successful from failed invaders in tropical northern Queensland fresh waters; and
b) a risk assessment process to assist managers in assessing the nature and magnitude of threats to environmental, social and economic values identified within the Wet Tropics Bioregion. This requires the collection of necessary data and evaluation of decision support systems (DSS) that can provide the most effective and efficient assessment for the region.
c) a regional risk assessment process using a DSS based on an existing model (or modified version) to assess environmental, social and economic impacts.

It is recommended that, for 4.2b, three models should be further evaluated to assess their effectiveness for regional risk assessment – i.e., *specifically* for non-native fishes in the Wet Tropics. The models to be tested are:

- the FISK protocol for non-native fishes in Great Britain (Copp *et al*. 2005) ;
- the WTVPRA protocol for pest vertebrates in the Wet Tropics (Harrison and Congdon 2002); and
- the AHP protocol for non-native fishes in the Murray-Darling Basin (Clunie *et al.* 2002).

4.3. using computer-based climate matching models (e.g., CLIMEX), investigate potential changes in current range limits estimated for non-native fishes in tropical northern Queensland in relation to projected climate regimes associated with global warming.

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6.0 APPENDIX A

Table 1A. Quantitative information available on a range of species characteristics for non-native fishes reported from Queensland fresh waters (* key attributes identified for successful invaders in studies by Kolar and Lodge 2002; Marchetti 2004))(Con = continuous; Propn = proportional; Cat. = Categorical) (Species characteristics list from Kolar and Lodge 2002).

Species characteristic	Type of variable	Family/Species																											
		Cichlidae																	Poeciliidae					Cyprinidae				Cd	Os
		a	b	c	d	e	f	g	h	i	j	k	l	m	n	o	p	q	r	s	t	u	v	w	x	y	z	aa	bb
Average Adult total length	Con.	√		√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√		√	√
Average larval length*	Con.	√																							√				
Age at maturity	Con.																				√				√				
% mature length at 1yr	Propn.																												
% mature length at 2yr*	Propn.																												
Longevity*	Con.	√			√																			√	√				
Annual fecundity	Con.																												
Lifetime reproductive potential	Con.																												
Average incubation period	Con.	√			√																√								
Maximum lifetime spawns	Con.																												
Egg diameter*	Con.	√			√	√	√	√	√	√	√	√	√	√		√	√				√				√	√	√	√	√
Parental Care	Cat.	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√
Diet (breadth/diversity)	Cat.	√		√		√	√	√	√	√	√			√	√	√	√		√	√	√	√	√	√	√	√	√	√	√
Maximum temp*	Con.	√		√				√		√										√	√		√	√		√	√		
Minimum temp*.	Con.	√		√						√										√			√	√		√	√		
Temp tolerance range*	Con.	√		√						√										√			√	√		√	√		
Salinity tolerance range*	Cat.	√																			√	√	√			√	√	√	
Native range area	Con.	√		√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√
Human use*	Cat.	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√
Year of introduction	Year	√		√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√
Species history establishment*	Cat.	√		√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√
Species history invasion*	Cat.	√		√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√
Species history introduction	Cat.	√		√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√	√

a. Oreochromis mossambicus; b. O. mossambicus (hybrid); c. Tilapia mariae; d. Tilapia zillii; e. Haplochromis burtoni; f. Thorichthys meeki; g. Archocentrus nigrofasciatum; h.Cryptoheros (A.) spilurus; i. Geophagus brasiliensis; j. Aequidens pulcher; k. Aequidens rivulatus; l. Hemichromis guttatus; m. Heros severus; n. Heros trimaculatus; o. Astronotus ocellatus; p. Amphilophus citrinellum; q. Vieja synspila; r. *Gambusia holbrooki; s. Poecilia reticulata; t. Poecilia latipinna; u. Xiphophoris maculates; v. X. helleri; w. Puntius conchonius; x. P. tetrazona; y. Cyprinus carpio; z. Carassius auratus; aa. Jordanella floridae; bb. Trichogaster trichopterus*