Why are we still using a "one size fits all" philosophy for systematic reserve planning in Australia?

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FUNDS available for investment in biodiversity conservation are small in comparison with the resources available to those interested in using the land for other purposes. In response to this disparity, the discipline of systematic conservation planning has developed tools to optimize decision making for investing limited conservation funds in the most effective and transparent manner possible (Possingham *et al.* 2006). Since its origins in the mid-1980s, systematic conservation planning has grown rapidly, spawning hundreds of peer-reviewed papers (Pressey *et al.* 2007). Importantly, it now shapes policy legislation in many terrestrial and marine regions across the globe.

In the mid-1990s, Australia was arguably the first country to depart from an *ad hoc*, site by site approach that had previously dominated conservation decision-making (Pressey *et al.* 1994). The Australian government embraced a process of systematically planning the future additions to the national reserve system (NRS); a vision for a "Comprehensive, Adequate and Representative System of Reserves" (CAR) across terrestrial Australia was endorsed in 1996 (Commonwealth of Australia 1996) and for marine environments in 1998 (ANZECC 1998). Since then, considerable resources have been invested to implement the CAR system and the NRS has grown to cover 11.6% of terrestrial Australia (Sattler and Taylor 2008) and 8% of marine Australia (CAPAD 2004).

There are two key components of creating a CAR system. The first is achieving a comprehensive reserve network, that is, one that contains every feature of biodiversity interest. These biodiversity features can include composition (genetic, species and community diversity), structure (habitats within a region, or aspects of the physical organization of an ecosystem that are important; for example, woody debris in an Australian temperate woodland) and function (key ecological and evolutionary processes). However, comprehensiveness alone will not ensure long term biodiversity persistence, if only small samples of each feature are conserved. Therefore, the second key component to achieving an effective reserve system is ensuring adequacy and representativeness: a CAR reserve system needs to be large enough to ensure the long term persistence of the features contained within it. As a consequence, it must represent a good sample of the variety (i.e., genetic diversity, habitat structural diversity) inherent in the biodiversity features of interest.

While we applaud the Australian state and federal governments' aim to increase the size of the reserve system in both terrestrial and marine environments, and their continued use of systematic, transparent CAR assessments when planning reserve additions, in this essay we outline two major concerns we have with current implementations of these systematic principles in Australia.

Firstly, we are concerned about the use of a fixed threshold for defining adequacy within the current CAR framework. A terrestrial or marine bioregion is often considered adequately protected if 10–15% of its area is reserved (protected areas in IUCN management categories I-VI; Commonwealth of Australia 2005; Environmental Protection Agency 2007). This percentage recommendation is currently 16 years old and was set with no explicit ecological foundation (Soulé and Sanjayan 1998; Svancara *et al.* 2005; Tear *et al.* 2005). There is an emerging consensus that the protection of 10–15% of original habitat extent is too little to sustain populations of most species for the long term, and will not protect the key ecological and evolutionary processes needed to sustain biodiversity (see, for example, analyses conducted by Rodrigues and Gaston 2001; Ward *et al.* 1999; Fahrig 2001; James and Saunders 2001; Possingham and Field 2001; Pressey *et al.* 2003; Desmet and Cowling 2004).

An adequacy threshold must at least reflect the minimum viable needs of the threatened species and ecosystems under consideration, and will consequently vary across space and over time (Desmet and Cowling 2004; Pressey *et al.* 2007). This is particularly true for those species where the matrix is a sink, as their future depends entirely on a well-managed reserve network. Identifying an appropriate local threshold might involve consideration of population viability, ecological processes and the interactions between species, ecosystems and landscape dynamics. While the Commonwealth Government recognize these factors as important (see, for example, ANZECC 1998), implementation has not yet occurred (Commonwealth of Australia 2005; Australian Marine Sciences Association 2006). Achieving true adequacy requires quantification and spatial mapping of all the important ecological and evolutionary processes underpinning current biodiversity patterns (Mackey *et al.* 2008). Data and methods are now becoming available to quantify some of these processes at appropriate spatial scales (e.g., Burgman *et al.* 2001; Pressey *et al.* 2007; Klein *et al.* 2008), but these have yet to be integrated into Australia's conservation planning framework.

Secondly, some reserves within the current NRS contribute little to biodiversity conservation. Arguably the most important goal of a protected area network is to separate samples of biodiversity from processes that threaten their continued existence (Margules and Pressey 2000). Therefore assessments of reserve systems also need to assess the effectiveness with which reserves mitigate such threats. The current CAR assessments include reserves classified under IUCN management categories V and VI. In such reserves, threatening processes such as trawling, mining, forestry and grazing of domestic livestock are legally allowed to continue, despite the fact they are often listed as key threats to species of conservation concern (Boitani *et al.* 2008). For example, trawling is allowed across 34% of the Great Barrier Reef Marine Park (IUCN management category VI; Hopley *et al.* 2007), an activity that has been shown to be detrimental to the benthos (Pitcher *et al.* 2000) and results in bycatch far exceeding that of targeted species (Harris and Poiner 1990; Hill and Wassenberg 2000). In terrestrial reserves, grazing by domestic livestock transforms understorey plant species composition, structure and phenology, increases run-off and proneness to erosion, and results in the trampling of tunnels and nests of ground-dwelling animals (McKenzie and Burbidge 2002; Woinarski *et al.* 2007). Grazing is a major threatening process for more than 40% of Australia's vertebrates listed on the EPBC Act (Australian Government 2008).

We believe that if the "A" in CAR is to be achieved in Australia, there is a need for the incorporation of functioning (or rehabilitating) ecosystems only. Many reserves in IUCN management categories V and VI should not be classed within the framework, as they do not provide adequate protection for ecological processes for the simple reason that threatening processes are explicitly allowed to continue. When only reserves in IUCN categories I–IV (i.e., those designated primarily for biodiversity conservation) are considered, the NRS currently covers only 6% of terrestrial Australia and 5% of the marine environment (CAPAD 2004).

We conclude by reiterating our support for the Australian government in embracing systematic conservation planning because it facilitates a transparent, inclusive and defensible planning process. However, if a truly comprehensive, adequate and representative reserve system is to be achieved, we must urgently understand and capture the needs of species and ecosystems, so they can persist within the NRS, rather than use simple static targets that have no basis in science. This means mapping and conserving the important ecological and evolutionary processes that underpin current biodiversity patterns, and identifying the minimum requirements for long-term persistence of threatened species and populations. We recognize that such analyses are not easy, they require more long-term data and advanced modelling. We also recognize such analyses will potentially suggest a significant increase in the size of the NRS — an unlikely scenario in an era when funds for acquisition and ongoing management of formal protected areas are becoming increasingly scarce. Therefore, a vital complementary strategy to achieving CAR is to maintain, and restore the condition of the unprotected matrix in order to conserve biodiversity assets outside of protected areas (Recher 2004; Soulé *et al.* 2004).

New planning methods are becoming available to identify thresholds for both (a) the amount of area that needs direct protection via reservation and (b) for conservation management prescriptions such as targeted invasive control, sustainable grazing and fine-scale fire management in the rest of the matrix (Pressey *et al.* 2007; Mackey *et al.* 2008). This type of planning will allow for systematic across-tenure conservation and remove the misconception that once a protected area threshold has been met for a region no other conservation efforts are necessary. Species extinction rates are higher than ever, despite the increased number of protected areas in Australia over the past decade. Therefore, a philosophical shift away from relying solely on protected areas for biodiversity conservation to a more holistic, landscape-wide approach may be the next step forward for saving Australia's threatened biodiversity.

REFERENCES

ANZECC. 1998. Guidelines for Establishing the National Representative System of Marine Protected Areas. Australia and New Zealand Environment and Conservation Council Taskforce on Marine Protected Areas, Environment Australia, Canberra.

Australian Marine Sciences Association. 2006. South East Marine Regional Plan Submission. Viewed 2 September 2008. http://www.amsa.asn.au/PDF-files/Submissions/AMSA_SEMR_Plan_submission_1Mar06.pdf

Boitani, L., Cowling, R. M., Dublin, H. T., Mace, G. M., Parrish, J., Possingham, H. P., Pressey, R. L., Rondinini, C. and Wilson, K. A., 2008. Change the IUCN protected area categories to reflect biodiversity outcomes. *PLoS* Biology **6:** e66.

Burgman, M. A., Possingham, H. P., Lynch, A. J. J., Keith, D. A., Mccarthy, M. A., Hopper, S. D., Drury, W. L., Passioura, J. A. NS Devries, R. J., 2001. A method for setting the size of plant conservation target areas. *Conservation Biology* **15:** 603–616.

CAPAD, 2004. National Summary, Marine and External. Collaborative Australia Protected Area Database. Department of Environment, Water, Heritage and Arts. Viewed 2 September 2008. http://www.environment.gov.au/parks/nrs/science/capad/2004/index.html

Commonwealth of Australia. 1996. The National Strategy for the Conservation of Australia's Biological Diversity, Commonwealth of Australia, Canberra.

Commonwealth of Australia. 2005. Direction for the National Reserve System- A partnership Approach. Canberra.

Commonwealth of Australia. 2008. EPBC Act List of Threatened Fauna. Viewed 5 September 2008. http://www.environment.gov.au/cgi-bin/sprat/public/publicthreatenedlist.pl?wanted=fauna.

Desmet, P. and Cowling, R., 2004. Using the species-area relationship to set baseline targets for conservation. *Ecology and Society* **9:** 11.

Environmental Protection Agency. 2007. Moreton Bay Marine Park draft zoning plan. Environmental Protection Agency, Brisbane.

Fahrig, L., 2001. How much habitat is enough? *Biological Conservation* **100:** 65–74.

Harris, A. N. and Poiner, I. R., 1990. By-catch of the prawn fishery of Torres Strait; composition and partitioning of the discards into components that float or sink. *Australian Journal of Marine and Freshwater Research* **41:** 37–52.

Hill, B. J. and Wassenberg, T. J., 2000. The probable fate of discards from prawn trawlers fishing near coral reefs: A study in the northern Great Barrier Reef, Australia. *Fisheries Research* **48:** 277–286.

Hopley, D., Smithers, S. G. and Parnell, K. E., 2007. The geomorphology of the Great Barrier Reef: Development, diversity, and change. Cambridge University Press, Cambridge.

James, C. and Saunders, D., 2001. A framework for terrestrial biodiversity targets in the Murray-Darling Basin. Murray-Darling Basin Commission, Canberra.

Klein, C. J., Wilson, K., Watts, M., Stein, J., Berry, S., Carwardine, J., Smith, M. S., Mackey, B. and Possingham, H., 2008. In press. Incorporating ecological and evolutionary processes into large-scale conservation planning. Ecological Applications.

Mackey, B., Berry, S. and Brown, T., 2008. Reconciling approaches to biogeographic regionalization: A systematic and generic framework examined with a case study of the Australian continent. *Journal of Biogeography* **35:** 213–229.

Margules, C. R. and Pressey, R. L., 2000. Systematic conservation planning. *Nature* **405:** 243–253.

McKenzie, N. L. and Burbidge, A. A., 2002. Australian Terrestrial Biodiversity Assessment. Chapter 6. National Land and Water Resources Audit, Commonwealth of Australia, Canberra.

Pitcher, C. R., Poiner, I. R., Hill, B. J. and Burridge, C. Y., 2000. Implications of the effects of trawling on sessile megazoobenthos on a tropical shelf in northeastern Australia. *Journal of Marine Science* **57:** 1359–1368.

Possingham, H. P. and Field, S. A., 2001. Regional bird extinctions and their implications for vegetation clearing policy. *Lifeline* **7:** 15–16.

Possingham, H., Wilson, K., Andelman, S. and Vynne, C., 2006. Protected Areas: Goals, Limitations, and Design. Pp. 509-533 in Principles of Conservation Biology ed M. J. Groom, G. K. Meffe. and C. R. Carrol. Sinauer Associates Inc., Sunderland.

Pressey, R. L., 1994. Ad hoc reservations: Forward or backward steps in developing representative reserve systems? *Conservation Biology* **8:** 662–668.

Pressey, R. L., Cowling, R. M., and Rouget, M., 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation* **112:** 99–127.

Pressey, R. L., Cabeza, M., Watts, M. E., Cowling, R. M. & Wilson, K. A., 2007. Conservation planning in a changing world. *Trends in Ecology & Evolution* **22:** 583–592.

Recher, H. F., 2004. WildCountry. Pacific *Conservation Biology* **10:** 221–222.

Rodrigues, A. S. L. and Gaston, K. J., 2001. How large do reserve networks need to be? *Ecology Letters* **4:** 602–609.

Sattler, P. S. and Taylor, M., 2008. Building Nature's Safety Net 2008. Progress for the Directions for the National Reserve System. WWF-Australia Report, WWF-Australia. Sydney.

Soulé, M. E. and Sanjayan, M. A., 1998. Ecology — Conservation targets: Do they help? *Science* **279:** 2060–2061.

Soulé, M. E., Mackey, B. G., Recher, H. F., Williams, J. E., Woinarski, J. C. Z., Driscol, D., Dennison, W. C., and Jones, M. E., 2004. The role of connectivity in Australian conservation. *Pacific Conservation Biology* **10:** 266–279.

Svancara L. K., Brannon R., Scott J. M., Groves C. R., Noss R. F. and Pressey R. L., 2005. Policy-driven versus evidence-based conservation: A review of political targets and biological needs. *BioScience* **55:** 989–995.

Tear T. H., Kareiva P., Angermeier P. L., Comer P., Czech B., Kautz R., Landon L., Mehlman D., Murphy K., Ruckelshaus M., Scott J. M. and Wilhere G., 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. *BioScience* **55:** 835–849.

Ward, T. J., Vanderklift, M. A., Nicholls, A. O. and Kenchington, R. A., 1999, Selecting marine reserves using habitats and species assemblages as surrogates for biological diversity. *Ecological Applications* **9:** 691–698

Woinarski, J., Mackey, B. Nix, H. and Traill, B. J., 2007. The Nature of Northern Australia: natural values, ecological processes and future prospects. ANU Press, Canberra.

Thirty years later, should we be more concerned for the ongoing invasion of Mozambique Tilapia in Australia?

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EXOTIC species complicate the management of native biological diversity and their ecosystems, because their effects are pervasive and varied (Cox 2004). For example, they can be beneficial and integral components of global agricultural economies (Sax *et al*. 2007), but when unmanaged and by definition invasive, they can negatively change important variables such as the genetics and population size of individual species, the diversity and structure of ecological communities, disturbance regimes and biogeochemical cycling (Vitousek 1990).

Attempts to explain the ecological impacts of an invader have characteristically focused on assembly theory (see Lodge 1993), of how competitive interactions (*e.g*., broad physiological tolerances, generalist resource usage, phenology, chemical defences) and the comparative distributions and abundances of invader and invaded (*i.e*., shifts in the ecological community structure), ultimately determine the outcome of an invasion (Lovett *et al*. 2006). Any or all of these attributes can change over time due to ecological and/or evolutionary processes, meaning that time since invasion becomes an important consideration when understanding the effects of many invaders. For example, changes in the invasive species themselves, cumulative changes in the invaded community and its physical environment, and interactions between the invader and other variables controlling the invaded environment, are common ecological and evolutionary processes; all occur over time and all potentially modulate the effects of an invasive species (Strayer *et al*. 2006).

The Mozambique Tilapia *Oreochromis mossambicus* belongs to a group of exotic fishes (Pisces: Cichlidae) that are among the most widely distributed in the world (Courtenay 1997). They are an important aquaculture species and food resource, but they have also founded wild populations in every nation (*i.e.*, > 90) in which they have been introduced (De Silva *et al*. 2004; Canonico *et al*. 2005). *O. mossambicus* was first recorded in Australia in 1977 (Arthington *et al*. 1984), and has since established feral populations along much of the Queensland coast and the mid-latitudes of Western Australia (Arthington and Blüdhorn 1994; Maddern *et al*. 2007). *O. mossambicus* continues to invade aquatic habitats in both regions and yet there is virtually no direct evidence of its ecological impact(s) in three decades. We are concerned for this ongoing invasion because experience from other exotic freshwater fish incursions (see for example Crowl *et al*. 1992 or Morgan *et al*. 2004) makes it difficult to accept how this species could not be having a detrimental effect in these novel environments, and especially after all this time. Ironically, we suspect it is this dearth of "hard evidence" which supports a general reluctance by the relevant authorities to invest in targeted research and management strategies for *O. mossambicus*. Or are we just panicking over nothing? In this paper we discuss how temporal ecological and evolutionary processes may have assisted the invasion of this Tilapia species in Australia over the past 30 years. Our review suggests that this incursion might be far from innocuous and action should be taken.

CHANGES IN THE INVADER: PHENOTYPIC PLASTICITY IN *OREOCHROMIS MOSSAMBICUS*

A species can change through time by acclimatization, including shifts in gene expression, resource allocation, or morphology and physiology within the lifespan of an individual. Such changes can influence many community and ecosystem processes (Eviner and Chapin 2003), and might be especially important in invasive organisms because successful invaders often display phenotypic plasticity for many traits that may assist dispersal and persistence (Agrawal 2001; García-Berthou 2007).

Invasion success often depends not so much on filling a vacant niche but on being a better exploiter of resources (Sax *et al*. 2007). Despite being primarily herbivorous or herbivorous/detritivorous, the success of Tilapias in colonizing an extensive range of novel environments has been partially attributed to opportunistic food habits such that food acquisition is rarely a limiting factor for them (Lévêque 2002; Bwanika *et al*. 2006). Moreover, examples of trophic plasticity have raised speculation that exotic Tilapia populations may have evolved to utilize a wider range of food resources (see Bowen and Allanson 1982; McKaye and Marsh 1983; McKaye *et al*. 1995). But what are the implications of trophic plasticity for the wider ecological community being invaded? Optimal foraging theory predicts that consumers will migrate to other habitats when it becomes too energetically expensive in the present patch (Schoener 1971), and Bowen (1979) showed that detritivory in adult *O. mossambicus* can result in protein deficiency and that they will tolerate this shortage only temporally while looking for an alternative form of protein supplementation. Examples of facultative feeding in introduced Tilapia are given by De Silva *et al*. (1984) and Maitipe and De Silva (1985), who describe dietary plasticity in several *O. mossambicus* populations. The literature presently lacks solid evidence of piscivory or obvious predatory behaviours by *O. mossambicus* (but see de Moor *et al*. 1986; Arthington and Blüdhorn 1994). However, we regularly observe cannibalism by this species in captivity regardless of feeding regime or stocking density and suspect that *O. mossambicus* eats a far wider range of organisms than is currently thought. The ability to flexibly exploit food resources through dietary shifts would be of clear benefit on the invasion front (Holway & Suarez 1999) and plastic responses may allow a population to persist under temporarily hard conditions such as during establishment, or allow persistence of the population under novel environmental conditions, providing more opportunity for selection to enhance adaptation (Ehrlich 1989; West-Eberhard 2003; Price *et al*. 2003). Further investigation of the potential for predation on sympatric native fishes in invaded habitats is warranted and especially in confined water bodies such as upstream pools, because it is there that predatory interactions would be most likely.

TEMPORAL CHANGES IN THE INVADED COMMUNITY AND ITS PHYSICAL ENVIRONMENT

Species composition within invaded communities can shift to moderate the impact of the invader or serve to encourage the invasion. There are lots of unknowns for how temporal changes in the invaded community and associated environments may assist dispersal by *O. mossambicus*. Nevertheless, its continued southwards range expansion in Western Australia (Morgan *et al*. 2004; Maddern *et al*. 2007) and westwards invasion of northern Queensland (Burrows unpubl. data) appears to indicate that either the nature of invaded communities and/or environments being encountered is favourable, or that *O. mossambicus* is adapting to them. Whichever, their success is more likely to be determined by factors such as life history strategy (Moyle and Light 1996) or how they change characteristics of the invaded ecosystem through their activities (Strayer *et al*. 2006), than by trophic plasticity alone. As an example, consider the reproductive strategy of *O. mossambicus*; males form dense nest-like aggregations in the substrate called leks to attract females for mating (Bruton and Boltt 1975). Mating success in *O. mossambicus* is highly skewed towards dominant (but not necessarily the largest) territorial males which also build larger leks, produce higher levels of hormones and pheromones, and have higher gonadosomatic indexes (Oliveira *et al*. 1996; Almeida *et al*. 2005). This species is thought to be highly socialized, using pulsed acoustic signals to communicate before, during and after mating (Amorim and Almada 2005), and females are mouth brooders meaning that buccal protection maximizes early survival in the offspring (Merrick and Schmida 1984).

In the seasonal pools of tropical north Queensland, we have often observed areas where male *O. mossambicus* have cleared several square meters of aquatic macrophyte habitat and built substantial (ø 0.5 m) breeding leks; males in full courtship colours aggressively defend this territory in the presence of one or more females and native fishes are conspicuous by their absence. We have also noticed that where the Tilapia population is large, numerous leks can occupy a significant proportion of benthic habitat. Several authors (Arthington and Blüdhorn 1994; Morgan *et al*. 2004; Maddern *et al*. 2007) have asked what effect this courtship behaviour has on native fish species, and particularly on those which require aquatic vegetation and/or intact substrate for reproductive purposes. Mackenzie *et al*. (2001) suggest that numbers of sympatric native fish species in Queensland have declined in areas where *O. mossambicus* proliferates. There is a clear need for experimental work to understand the possibility that with time in these new environments, *O. mossambicus* has developed a much wider capacity to disrupt local fish communities than is currently thought.

INTERACTING FORCES: INVASION OPPORTUNITIES AND CONSTRAINTS

Invading species are not the only factor which controls the ecosystems they invade. For example, stochastic and seasonal disturbances are often important, and can interact with the invading species to variably influence the incursion. The degree to which these other controlling factors might affect invasion success also depends, however, on the physiological attributes of the invader and *O. mossambicus* possesses many of those required for successful radiation (see Ehrlich 1989). It can withstand at least seawater strength salinity (Laundau 1992), high temperatures and low dissolved oxygen levels (Lovell 1998), high nutrient concentrations (Popma and Masser 1999), a variable pH (Krishna Murthy *et al*. 1984), and a range of organic and inorganic pollutants (*e.g.*, Noorjahan *et al*. 2003; Somanath 2003). Considering these then it isn't difficult to understand just how readily Tilapia could invade a wide variety of habitats and/or usurp native sympatric species, which have mostly evolved comparatively more restricted niche requirements. This is particularly so when human-induced disturbance occurs, such as when weed invasions lower dissolved oxygen levels or when waterways become polluted or regulated, diminishing niche space for native fishes and creating opportunities for Tilapia incursions. The broad physiological tolerances of *O. mossambicus* also indicate how difficult it might be to stop this invasion and why more attention should be given to its likely impacts.

CONCLUDING REMARKS

Invasive species also complicate the management of native ecosystems because arguments persist at both the discourse and administrative levels over issues including the relative importance of exotics against other threatening processes (*e.g.*, habitat fragmentation), over which ecosystems are most at threat (*e.g.*, freshwater *v.* terrestrial), and over which species are the most threatening. Of course there is plenty of politics in how all of this is decided, but to raise the profile of *O. mossambicus* to being an invasive species of consequence in tropical and sub-tropical Australia requires a greater emphasis on how ecological effects can change over time. We need to expose the mechanisms by which *O. mossambicus* effects change through time by (1) identifying how those attributes which assist invasion create a temporal feedback between the invader and the invaded communities and ecosystems (Strayer *et al*. 2006), and (2) giving a greater emphasis to experimental designs that yield empirical and therefore evidence-based data (see Reddiex and Forsyth 2006).

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REFERENCES

Agrawal, A. A., 2001. Phenotypic plasticity in the interactions and evolution of species. *Science* **294:** 321–326.

Almeida, O. G., Miranda, A., Frade, P., Hubbard, P. C., Barata, E. N. and Canário, A. V. M., 2005. Urine as a social signal in the Mozambique tilapia (*Oreochromis mossambicus*). *Chem. Senses* **30 (Suppl 1):** 309–310.

Amorim, M. C. P. and Almada, V. C., 2005. The outcome of male-male encounters affects subsequent sound production during courtship in the cichlid fish *Oreochromis mossambicus*. *Anim. Behav.* **69:** 595-601.

Arthington, A. H. and Blüdhorn, D. R., 1994. Distribution, genetics, ecology and status of the introduced cichlid, *Oreochromis mossambicus*, in Australia. *Mitt. Internat. Verein. Limnol.* **24:** 53–62.

Arthington, A. H., McKay, R. J., Russell, D. J. and Milton, D. A., 1984. Occurrence of the introduced Cichlid *Oreochromis mossambicus* (Peters) in Queensland. *Aust. J. Mar. Freshwat. Res.* **35:** 267–272.

Bowen, S. H., 1979. A nutritional constraint in detritivory by fishes: the stunted population of *Sarotherodon mossambicus* in Lake Sibaya, South Africa. *Ecol. Monogr.* **49:** 17–31.

Bowen, S. H. and Allanson, B. R., 1982. Behavioural and trophic plasticity of juvenile *Tilapia mossambica* in utilization of the unstable littoral habitat. *Environ. Biol. Fishes* **7:** 357–362.

Bruton, M. N., and Boltt, R. E., 1975. Aspects of the biology of *Tilapia mossambica* Peters (Pisces: Cichlidae) in a natural freshwater lake (Lake Sibaya, South Africa). *J. Fish Biol.* **7:** 423–445.

Bwanika, G. N., Chapman, L. J., Kizito, Y. and Balirwa, J., 2006. Cascading effects of introduced Nile perch (*Lates niloticus*) on the foraging ecology of Nile tilapia (*Oreochromis niloticus*). *Ecol. Freshwat. Fish* **15:** 470–481.

Canonico, G. C., Arthington, A., McCrary, J. K. and Thieme, M. L., 2005. The effects of introduced tilapias on native biodiversity. *Aquat. Conserv.: Mar. Freshwat. Ecosyst.* **15:** 463–483.

Courtenay, W. R., 1997. Tilapias as non-indigenous species in the Americas: environmental, regulatory and legal issues. Pp. 18–33 *in* Tilapia Aquaculture in the Americas, Vol. 1 ed by B. A. Costa-Pierce and J. E. Rakocy. World Aquaculture Society, Los Angeles.

Cox, G. W., 2004. Alien Species and Evolution. Island Press, Washington.

Crowl, T. A., Townsend, C. R. and McIntosh, A. R., 1992. The impact of introduced brown and rainbow trout on native fish: the case of Australasia. *Rev. Fish Biol. Fish.* **2:** 217–241.

de Moor, F. C., Wilkinson, R. C. and Herbst, H. M., 1986. Food and feeding habits of *Oreochromis mossambicus* (Peters) in hypertrophic Hartbeesport Dam, South Africa. *S. Afr. J. Zool.* **21:** 170–176.

De Silva, S. S., Perera, M. K. and Maitipe, P., 1984. The composition, nutritional status and digestibility of the diets of *Sarotherodon mossambicus* from nine man-made lakes in Sri Lanka. *Environ. Biol. Fishes* **11:** 205–219.

De Silva, S S., Subasinghe, R. P., Bartley, D. M. and Lowther, A., 2004. Tilapias as alien aquatics in Asia and the Pacific: a review. FAO Technical Paper No. 453, Rome.

Ehrlich, P. R., 1989. Attributes of invaders and the invading processes: vertebrates. Pp. 315–328 *in* Biological Invasions: A Global Perspective ed by J. Drake, F. di Castri, R. Groves, F. Kruger, H. Mooney, A. Rejmánek and M. Williamson. Wiley, New York.

Eviner, V. T. and Chapin, F. S. III., 2003. Functional matrix: a conceptual framework for predicting multiple plant effects on ecosystem processes. *Annu. Rev. Ecol. Syst.* **34:** 455–485.

García-Berthou, E., 2007. The characteristics of invasive fishes: what has been learned so far? *J. Fish Biol.* **71 (Suppl. D):** 33–55.

Holway, D. A. and Suarez, A. V., 1999. Animal behaviour: an essential component of invasion biology. *TREE* **14:** 328–330.

Krishna Murthy, V., Dhananjaya Reddy, Y., Manohar Reddy, R., Bhaskar, M. and Govindappa, S., 1984. Changes in brain acetylcholinesterase activity and behaviour of freshwater fish in acid polluted environment. *Environ. Ecol.* **2:** 79–82.

Landau, M., 1992. Introduction to Aquaculture. John Wiley and Sons, New York.

Lévêque, C., 2002. Out of Africa: the success story of tilapia. *Environ. Biol. Fishes* **64:** 461–464.

Lodge, D. M., Taylor, C. A., Holdich, D. M. and Skurdal, J., 2000. Nonindigenous crayfishes threaten North American biodiversity. *Fisheries* **25:** 7–20.

Lovell, T., 1998. Nutrition and Feeding of Fish. Kluwer, Boston.

Lovett, G. M., Canham, C. D., Arther, M. A., Weathers, K. C. and Fitzhugh, R.D., 2006. Forest ecosystem responses to exotic pests and pathogens in eastern North America. *Bioscience* **56:** 395–405.

Mackenzie, R., Jackson, P. and Cotterell, E., 2001. Control of Exotic Pest Fishes — An Operational Strategy for Queensland Freshwaters 2002–2005. Department of Primary Industries, Queensland.

Maddern, M. G., Morgan, D. L. and Gill, H. S., 2007. Distribution, diet and potential ecological impacts of the introduced Mozambique mouthbrooder *Oreochromis mossambicus* Peters (Pisces: Cichlidae) in Western Australia. *J. Roy. Soc. W.A.* **90:** 203–214.

Maitipe, P. and De Silva, S. S., 1985. Switches between zoophagy, phytophagy and detritivory of *Sarotherodon mossambicus* (Peters) populations in twelve man-made Sri Lankan lakes. *J. Fish Biol.* **26:** 49–61.

McKaye, K. R. and Marsh, A., 1983. Food switching by two specialized algal-scraping Cichlid fishes in Lake Malawi, Africa. *Oecologia* **56:** 245–248.

McKaye, K. R., Ryan, J. D., Stauffer, J. R. Jr., Lorenzo, J., Lopez, P., Vega, G. I. and van den Berghe, E. P., 1995. African tilapia in Lake Nicaragua: ecosystem in transition. *Bioscience* **45:** 406–411.

Merrick, J. R. and Schmida, G. E., 1984. Australian Freshwater Fishes: Biology and Management. Griffin, Adelaide.

Morgan, D. L., Gill, H. S., Maddern, M. G. and Beatty, S. J., 2004. Distribution and impacts of introduced freshwater fishes in Western Australia. *N. Z. J. Mar. Freshwat. Res.* **38:** 511–523.

Moyle, P. B. and Light, T., 1996. Biological invasions of fresh water: empirical rules and assembly theory. *Biol. Conserv.* **78:** 149–161.

Noorjahan, C. M., Dawood, S. S. and Nausheen, D., 2003. Impact of dairy effluent on biochemical constituents of fish *Oreochromis mossambicus*. *J. Ecotox. Environ. Monitor.* **13:** 227–231.

Oliveira, R. F., Almada, V. C. and Canário, A. V. M., 1996. Social modulation of sex steroid concentrations in the urine of male cichlid fish *Oreochromis mossambicus*. *Horm. Behav.* **30:** 2–12.

Popma, T. and Masser, M., 1999. Tilapia Life History and Biology. Southern Regional Aquaculture Centre Publication No. 283, Texas.

Price, T., Qvarnström, A. and Irwin, D. E., 2003. The role of phenotypic plasticity in driving genetic evolution. *Proc. Roy. Soc. Lond. B* **270:** 1433–1440.

Reddiex, B. and Forsyth, D. M., 2006. Control of pest mammals for biodiversity protection in Australia. II. Reliability of knowledge. *Wildl. Res.* **33:** 711–717.

Sax D. F., Stachowicz, J. J., Brown, J. H., Bruno, J. F., Dawson, M. N., Gaines, S. D., Grosberg, R. K., Hastings, A., Holt, R. D., Mayfield, M. M., O'Connor, M. I. and Rice, W. R., 2007. Ecological and evolutionary insights from species invasions. *TREE* **22:** 465–471.

Schoener, T. W., 1971. Theory of feeding strategies. *Annu. Rev. Ecol. Syst.* **2:** 369–404.

Somanath, V., 2003. Impact of tannery effluent on bioenergetics of fishes. *J. Ecotox. Environ. Monitor.* **13:** 161–173.

Strayer, D. L., Eviner, V. T., Jeschke, J. M. and Pace, M. L., 2006. Understanding the long-term effects of species invasions. *TREE* **21:** 645–651.

Vitousek, P. M., 1990. Biological invasions and ecosystem processes: towards and integration of population biology and ecosystem studies. *Oikos* **57:** 7–13.

West-Eberhard, M. J., 2003. Development Plasticity and Evolution. Oxford University Press, Oxford.

Nauru — Opportunity in loss

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THE nation of Nauru lies 72 km south of the equator in the Pacific Ocean. Until the discovery of phosphate deposits on the island at the beginning of the 19th century Nauru was covered in dense tropical rainforest which was tended in a traditional form of agroforestry by the indigenous Nauruans. From the forest came fruits such as pandanus, fibre, and timber such as the Tomano or Pacific Mahogany. Aquaculture was performed in the Buada Lagoon, fish, such as tuna were caught from the steep ring reef, and Noddy birds were caught as they came in from the ocean to roost in the evenings.

Over millennia rich deposits of phosphate formed between the coral pinnacles that form the substrate of the island. The development of superphosphate fertilizer in the late 19th century made phosphate rock a valuable commodity. Nauru, an island which was erstwhile largely ignored by the colonial powers, was thrown into the world marketplace (Weeramantry 1992).

Today the 13,500 inhabitants rely almost entirely on imported foods as most of the trees and birds are gone, as is almost all of the cheaply extractable phosphate (Quanchi 2007). Additionally, around 40% of the fringing reef has died due to sediments from mining and sewerage outflow, affecting local fisheries (NCCC 1999).

The people of Nauru have not given up hope for the future of life on the island. In 1997, the Nauru Rehabilitation Corporation (NRC) was established by the Nauruan government to begin work on rehabilitating the mined areas of the island according to the Nauru Australia Cooperation (NAC) Rehabilitation and Development Feasibility Study (RDFS) of 1994. The process of restoration was not adequately described in the RDFS and as the NRC lacks the ecological expertise necessary to develop the plan further it is unable to fulfill its brief (Clodumar, pers. comm.). There are opportunities for the knowledge and skills of ecologists and conservation biologists to provide real benefits for the people of Nauru, while employed in research which benefits their own disciplines.

The denuded karrenfield of coralline rock which makes up the interior of the island post mining provides many examples for ecological research. For instance, the coral pinnacles create a landscape of islands of different sizes at greater or lesser distances from remnant areas of vegetation. Different areas of pinnacles have been exposed during different periods of mining allowing different time periods for colonization (Weeramantry 1992). Colonization is occurring; the oldest areas are covered with well developed vegetation, while the newest are stripped to bare rock. Despite regenerating for decades, the two topsoil deposits accumulated during mining are species poor and sparsely covered with acacia and little else.

Of particular interest are the issues of cadmium contamination and climate alteration consequent to mining. Cadmium once associated with the phosphate has been distributed throughout the environment accumulating in soils and vegetation (Blake 1992; Kirk 2000; Manner, Thaman and Hassal 1984; NCCC 1999). The long term effects of this ecologically and sociologically have not been explored. Neither have the effects of the removal of vegetation on the local climate. The island was once covered in dense rainforest, and rain was a regular and common phenomenon. Today, the interior of the island bakes under the equatorial sun and receives less rainfall than is necessary to support rainforest communities (Department of Economic Development and Environment 2003; Kirk 2000; Gowdy and McDaniel 1999). While I was on the island earlier this year, I was told that until a week before my arrival the island had not received substantial rain for a period of three years (Barker, pers. comm.). The consequences of these kinds of changes for the ecology of the island are considerable.

Also of interest are the sociological aspects of conservation. Nauru was rapidly transformed environmentally and socially. Today, barely anyone enters the interior of the island except the Noddy bird hunters and those employed to extract the remaining phosphate. From a society with highly evolved traditional systems of ecological knowledge which guided their agroforestry and aquaculture practices (to give two examples), Nauruan society is now highly urbanized. Nauruans have become reliant on external resources which are becoming less affordable as product and transportation costs increase. Nauru presents a opportunity to successfully employ the principles of Integrated Conservation and Development Projects (ICDP's) as conflicts between biodiversity conservation goals and development potential are minimal. This is due to indigenous recognition of the need for re-establishment of rainforest to provide ecosystem services, and because levels of endemism on the island are low (NCCC 1999).

Akin to Easter Island/Rapanui, Nauru provides a warning about the societal effects of unmitigated environmental degradation. However, Nauru also provides an opportunity to rectify environmental abuse, to develop skills and knowledge of how to reconfigure the relationship between humanity and the environment in a manner that is sustainable and conserves the biological and cultural values of our world.

REFERENCES

Barker, X. Acting Director of the University of the South Pacific in Nauru. Personal Communication. 9th September, 2008

Blake, C. 1992. Environmental distribution of heavy metals on Nauru, Central Pacific, and possible relationships to human health. B.App.Sci. Honours Report, University of New England, Northern Rivers, Lismore, New South Wales, Australia

Clodumar, V. CEO, Nauru Rehabilitation Corporation. Pers. Comm. 12th September, 2008.

Department of Economic Development and Environment. 2003. First National Report of the Republic of Nauru to the United Nations Convention to Combat Desertification. Republic of Nauru.

Gowdy, J. M., and C. N. McDaniel. 1999. The physical destruction of Nauru: An example of weak sustainability. Land Economics **75:** 6.

Kirk, A. R. 2000. The Republic of Nauru. The Blue Pacific Series. Capital Letters Publishing, Masterton, NZ.

Manner, H. I., Thaman, R. R., and D. C. Hassall. 1984. Phosphate mining induced vegetation changes on Nauru Island. Ecology **65:** 11.

NCCC. 1999. Climate Change Response. Nauru National Committee on Climate Change. Nauru.

Quanchi, M. 2007. Troubled times: Development and economic crisis in Nauru. Pp. 249–64 in Security and development in the Pacific Islands: Social resilience in emerging States. ed by M. A. Brown. Lynne Rienner Publishers, Boulder, CO, USA.

Eucalypt decline and dead trees: if it's not sexy few seem to care

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WHETHER its Bell Miner Associated Dieback (BMAD), Eucalypt Decline or as the wording of the Key Threatening Process nomination states "Forest eucalypt dieback associated with over-abundant Bell Miners *Manorina melanophrys*, and psyllids, the phenomena of eucalypt decline on the east coast of Australia is serious. Approximately 781,000 hectares of east coast forests are currently predisposed to decline (Vic Jurskis unpub. data 2008). Among all the forms of dieback affecting east coast forest types, BMAD is one of the most serious and over $700,000 has been spent in the last few years trying to work out the what, why and where of managing this threat to forest ecosystems. Despite a 'task force' (the BMAD Working Group) having been set up to report on Bell Miner Associated Dieback, to the majority of Australian's, it either doesn't exist, they cannot agree on an acceptable term to describe it or it is not on their "peril radar". Substantial effort has been invested trying to unravel the mystery of BMAD and mitigating its impacts. Yet, to some decision makers, BMAD is not important and they see little association between this phenomena and drought, poor land management, weeds, fire and climate change; issues which do seem to capture their attention.

In northern New South Wales alone, it was estimated in 2004 using aerial surveys that 200,000 hectares of forest in the Kyogle area were affected by this form of dieback (FNSW unpub. data. 2004). Over the whole State there are 2.5 million hectares of forest that are suitable forest types where eucalypt decline could occur (after Keith 2004). While factors such as exclusion of fire (Jurskis and Turner 2002; Jurskis 2005), poor forest restoration following timber harvesting (Florence 2005), forest structure and canopy degradation (Stone 1998; Stone *et al*. 2008), drought, weed invasion and a multitude of other factors are listed as potential culprits (Wardell-Johnson 2006), we may never be able to tease out a cause. What is certain is that large areas of forests are dying and Lantana *Lantana camara* is often abundant, fire frequency has decreased, Bell Miners are often in large numbers, as are psyllid insects, forest structure has been simplified and tree canopy cover has been reduced. The consequences of this is an increasing area of forests with dead trees, poor forest recruitment, serious decline of habitat health and an economic loss to the forest based industries that rely on flowers, seeds and wood.

The role of the native bird, the Bell Miner, in some tree decline is stark. I have no doubt that the birds and their sap sucking psyllid insect mates (*Glycaspis* spp. *and Cardiaspina* spp.), are involved in the cycle of BMAD. Though I dare not ridicule this clever bird merely capitalizing on an opportunity to do what they were designed to do . . . eat insects and procreate. Bell Miners are adored by the general public as those lovely green birds that "remind me of my childhood". Although to those who live with them each day and see their trees dying and the diversity of other resident native birds dwindle, they are a menace. Henry Kendall was more insightful than he realized in his poem "Bellbirds" when he wrote;

The beauty and strength of the deep mountain valleys:
Charming to slumber the pain of my losses
With glimpses of creeks and a vision of mosses

Little did he realize that the "*losses*" attributed to Bell Miners and associated factors would potentially be tens of thousands of hectares of trees in those deep mountain valleys and the *glimpses* of creeks are just that, glimpses, because Lantana now has exclusive rights to the views. No longer do the Bell Miners *direct him* (the wayfarer) *to spring and to river*. . . *when fiery December sets foot in the forest*, now they fill our hearts with dread and direct us to graveyards of trees. Now we know what the consequences of that bell-like call means — more forest devastation.

On a weekly basis calls are received through the BMAD website (www.bmad.com.au) from people pleading for help to save their trees from the BMAD that has been cast upon them, soon after Bell Miners have taken over the surrounding tree canopy. Our efforts to raise awareness of BMAD and to invite participation in our trials and research are taking effect, as more and more records of outbreaks are reported. However, there are still those who are in denial that BMAD exists. I have heard of land managers rejecting BMAD as a phenomena and not supporting opportunities to carry out surveys to assess reports of dieback. There are certainly a range of factors involved in the initiation of what we call BMAD and the birds have their role in tree decline. I would have thought that the more investigations we carry out the more knowledge we collect. In some forums statements have been made that linking BMAD to global warming and carbon sequestration is "drawing a long bow", when to me the link seems rock-solid. Calculations by the BMAD Working Group estimate that 15 million tonnes of CO^2 storage from native coastal forests could be lost per annum due to dieback in NSW. This figure is based on annual new growth and does not include the stored CO^2 in standing biomass, so the correlation between 2.5 million hectares of healthy forests and their role in sequestration of carbon is significant.

The problem we face is that dying trees don't bleed and don't have eyes and a brain, and most voting city dwellers never see the devastation of a dying forest. In the absence of a champion to capture the economic intrigue of politicians and coerce the community, there is little likelihood of having the issue recognized for its potential. The BMAD Working Group have made numerous inquiries to enlist a champion, someone who is viewed by the public as an important personality, but sadly our topic does not stimulate the anthropomorphic nerve like whale hunting or eating our national emblem. If it is not seen as sexy and tangible, it not important and recognition is not forthcoming.

The BMAD Working Group, a committee of 12 stakeholders has worked tirelessly since 2001 to be at the forefront of this ecological catastrophe, and with a short term budget. Their united efforts to lead research and adaptive management trials to mitigate dieback have largely gone unrecognized and in some forums they have been criticized. The failure to think in dollars and votes like politicians and bureaucrats, and more like concerned citizens, alarmed land managers and conservationists has been a weakness of the campaign. In response, the group has now began to compile an economic impact assessment that should hit the political and economic alarm buttons — hopefully it is not all too late.

While it is challenging to quantify the area of impact of BMAD, the costs to industry, and the effects on tourism of dying forests, the implications of not managing the spread of BMAD is staggering. Based on rough estimates, our initial calculations report significant losses to the $21 million New South Wales honey industry, with one apiarist reporting losses of $40,000 from one forest apiary site each year (Geoff Manning pers. comm.). Costs to the timber industry are almost impossible to quantify, but restoration of dying forests decimated by eucalypt decline range from $800–3000 per hectare (Peter StClair pers. comm.) depending on the level of impact. The double edge sword in the forest is the potential loss of CO^2 storage from dying and dead trees. With large areas of forest affected each year by dieback and a loss of around 15 million tonnes/year of CO2 stored, it is likely that achieving climate change targets will be significantly crippled by dieback. The link between BMAD and climate change seems quite obvious to us and we don't understand why it's not on the political agenda.

It was not until Warragamba Dam water levels dropped to dangerous levels and Sydney-siders felt the pinch because they could no longer wash their cars and water their gardens, that the populous realized we were in severe drought. When fuel costs continued to increase over the $1.00/litre most Australian's started to listen and they realized that fossil fuels are not renewable and that alternative energies need investigation. I just hope that we don't have to wait until the costs of eucalypt hardwood and a jar of honey increase by 500%, or until a famous tourism site like the Three Sisters in the World Heritage Blue Mountains has a back drop that resembles a tree cemetery, or Australian native mammals are better represented in zoos than in the wild before BMAD and eucalypt tree decline becomes sexy.

REFERENCES

Florence, R. 2005. Social responsibility for the New South Wales forests. *Australian Forestry* **68**: 1–2.

Jurskis, V. and Turner, J. 2002. Eucalypt dieback in eastern Australia: a simple model. *Australian Forestry* **65**: 87–98.

Jurskis, V. 2005. Decline of eucalypt forests as a consequence of unnatural fire regimes. *Australian Forestry* **68**: 257–262.

Keith, D. 2004. Ocean Shores to Desert Dunes: The Native Vegetation of NSW and ACT. NSW Department of Environment and Conservation, Hurstville.

Stone, C. 1998. Assessment and monitoring of decline and dieback of forest eucalypts in relation to ecologically sustainable forest management: a review with a case study. *Australian Forestry* **62**: 51–58.

Stone, C., Kathuria, A., Carney, C. and Hunter, J. in press. Forest canopy condition and stand structure associated with bell miners (*Manorina melanophrys*) on the central coast of NSW.

Wardell-Johnson, G., Stone, C., Recher, H. and Lynch, A. J. 2006. *Bell Miner Associated Dieback (BMAD): Independent Scientific Literature Review*. Occasional paper DEC 2006/116.