Progress towards the eradication of three melastome shrub species from northern Australian rainforests

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Summary

To eradicate a weed incursion, its extent must be delimited and each infestation extirpated. Measures for delimitation and extirpation are utilized to assess the progress of eradication programs currently targeting three melastome shrub species (*Clidemia hirta*, *Miconia nervosa* and *M. racemosa*) in north-eastern Australia. The main infested area for each species was determined during the year after initial detection, but expanded surveys have led to the discovery of isolated, smaller outlying patches. Programs are refining survey methods (including search frequency) to prevent reproduction. Weed incursions that are limited to single infestations represent a prime opportunity for eradication. However, population and dispersal data indicate that eradication will require an ongoing investment for some time for all three species. Highly persistent seed and dispersal by frugivores suggest that eradication may prove extraordinarily difficult should any of these species spread or be discovered at more locations.

Key words: Delimitation, extirpation, Melastomataceae, miconia, clidemia

Introduction

Early intervention during the lag phase of a weed incursion can save some of the millions, or even billions of dollars subsequently incurred as mitigation costs, economic or environmental damage (Hobbs and Humphries 1995). Many serious national weed incursions originated at limited points of introduction and left authors (e.g. Mack and Lonsdale 2002, Simberloff 2003) reflecting on the lost opportunity to eliminate species when incursions commenced. Early intervention and eradication present particularly attractive management options where weed incursions are limited to single sites.

Amongst the invasive Melastomataceae, the control of *Miconia calvescens* D.C. on Tahiti (introduced in 1937) or Hawaii (introduced in 1959) in the couple of decades before larger infestations had developed could have avoided many of the costs later associated with ongoing population suppression (including chemical control operations, helicopter surveys and biological control agent testing), as well as environmental damage (Meyer 1998, Medeiros *et al.* 1997).

A species with a highly restricted distribution presents a prime opportunity to prevent a possible major incursion and eliminate the threat posed to natural and agricultural systems. Larger incursions (in terms of number of infestations and/or total infested area) require greater resources devoted to eradication (Rejmánek and Pitcairn 2002), while smaller infested areas and fewer sites can present a greater chance of achieving eradication (Groves and Panetta 2002, Panetta and Timmins 2004). Early control of small and localized populations aids incursion containment (Zamora *et al.* 1989) by reducing opportunities for seed or propagule spread within or to new infestations. Similarly, the sooner control operations start suppressing a weed population, in particular limiting or reducing the density of the soil seed bank, the earlier it may be extirpated (Drayton and Primack 1999).

Exotic melastome seed can be spread by frugivorous agents (Murphy *et al.* 2008, Meyer 1998), pigs, bushwalkers, vehicles (Smith 1992), water, deliberate cultivation and accidental transport of seed-contaminated materials (such as soil) (Meyer 1998). Clearly, all means of dispersal must be considered when attempting to delimit a melastome incursion. Survey strategies to detect all plants around known infestations need to take into account physical and biotic dispersal mechanisms, especially movement by water and frugivores. Local research has shown that models based on the dispersal by birds of similar sized fruits (functional groups) of native rainforest species within a rainforest landscape (Westcott and Dennis 2006) can be used for estimating search radii from infestation points of origin when applied to a *M. calvescens* infestation (Murphy *et al.* 2008). Such information can be combined with field experience and data to review and refine search strategies to prevent plants from reaching maturity.

Eradication programs may require financial and institutional commitment for many years to reach completion (Mack and Lonsdale 2002) so there is a need for periodic assessment of progress (and/or eradication feasibility) over these time scales (Buddenhagen 2006, Panetta and Lawes 2007). To date indicators of eradication progress have focused on delimiting and extirpating multiple infestations (Panetta and Lawes 2005, Panetta 2007). In this paper we document the progress that has been made towards the eradication of three invasive melastome shrubs from single locations on the north-eastern wet tropical coast of Australia. These species have been subjected to a national cost sharing weed eradication program which commenced in 2001. Progress toward eradication is assessed in relation to conformity to the delimitation and extirpation criteria (Panetta and Lawes 2005, Panetta 2007).

The species

Clidemia hirta (L.) D.Don (Koster's Curse) is a perennial shrub, native to tropical Central and South America, which has become an entrenched crop and environmental weed throughout southern Asia and on many Pacific Islands (Waterhouse 2003). *C. hirta* forms dense thickets that smother plantations, pastures and native vegetation, causing millions of dollars worth of damage to primary production (Peters 2001). Plants may fruit all year round in wetter areas of Hawaii, with hundreds of seeds in each berry (DeWalt 2006, Smith 1992). Individual plants can produce an average of 3.8 million seeds per year (Medeiros 2004). Each seed is 0.5 mm in diameter and fruit is readily consumed and dispersed by birds (Wagner *et al.* 1999). Plants can produce mature fruit at a height of 45 cm, in a minimum of 365 days from emergence (Graham and Setter 2007). *C. hirta* develops a persistent seed bank (DeWalt 2003) and seedling emergence has been reported up to 10 years after the removal of nearby mature plants (Smith 1992). *C. hirta* was first reported in Hawaii in the 1940s and had spread over 100–300 ha on Oahu by 1952 (Smith 1992). Despite some control efforts (including biological control) commencing in 1954, on Oahu this weed was found over at least 31 350 ha by 1977 and up to 100 000 ha in 1992 (Wester and Wood 1977, Smith 1992).

In 2001 an infestation of a few hundred plants was discovered on a creek bank adjacent to a former palm nursery (Waterhouse 2003) in far north Queensland near Julatten, 426 m above sea level (16°36′S, 145°20′E) (Figure 1). The population was spreading into surrounding *Acacia* spp. and *Alstonia* spp. dominated forest areas via avian and water dispersal (Waterhouse 2003), with the heaviest infestation in a steep gully at the site of the original detection. *C. hirta* is an AQIS alert list species so it is included in Northern Australian Quarantine Strategy (NAQS) surveys of at-risk arrival areas (Waterhouse 2002).

Miconia nervosa (J.E.Smith) Triana. is native to Central and South America (Belize to Brazil), occurring as a widespread, locally abundant rainforest shrub with a relatively high light requirement for growth (Newell *et al.* 1993). *M. nervosa* persists in the rainforest understorey and so has a degree of shade tolerance (Ellison *et al.* 1993), but requires a canopy opening for flower and fruit production. It is common in large gaps and along tracks (Newell *et al.* 1993). It is readily dispersed by birds (Chazdon *et al*. 2003) and year round fruit production was noted by Poulin *et al*. 1999 (citing Croat 1978) in Panama. During the survey and control of a *M. calvescens* infestation in October 2004, 10 mature *M. nervosa* plants were found in a rainforest habitat 250 m above sea level in the Whyanbeel Valley (16°22′S, 145°19′E) near Miallo, north of the township of Mossman (Figure 1) (Sydes and Galway 2008). Prior to identification plants had been noted over the previous two years, and the location of the introduction is known. *M. nervosa* is listed as a weed in Peru (Holm *et al.* 1979) and is considered to have weed potential in Australia (B. Waterhouse personal communication 2004). The plants in the Whyanbeel Valley comprise the only known naturalized occurrence outside of South America.

Miconia racemosa (Aubl.) DC. is a tropical American shrub native to Central and north-eastern South America, including the West Indies, Puerto Rico, Venezuela and Brazil (Francis 2007). It is an evergreen shrub up to 3 m tall in its native range, at altitudes between sea level and 900 m where the annual rainfall is 1600–3000 mm (Francis 2007). In its native range *M. racemosa* is considered weedy though rarely abundant, occurring as scattered individuals or in small clumps and with other early successional species (Francis 2007, Olander *et al.* 1998). It has excellent colonizing abilities in disturbed or degraded sites and occurs on abandoned fields, roadsides, plantations, landslides and secondary forests (Myster 2003, Francis 2007). *M. racemosa* inflorescences are panicles with many tiny white, pink, or purple flowers. The fruits are 3–5 mm diameter berries and contain tiny bird dispersed seed (Wunderle 1995), with flowers and fruit present throughout the year (Francis 2007). Reproduction is favoured by disturbance but is not dependant on it, although partial sunlight is required for flowering and fruiting (Francis 2007). Stems can coppice and re-shoot following damage (particularly landslides) and plants can be multi-stemmed (Francis 2007). In June 2002 a small infestation of *M. racemosa* containing 300–400 plants was found in a semi-urban forested gully near Kuranda (16°48′S, 145°37′E), approximately 330 m above sea level (B. Waterhouse personal communication 2004) (Figure 1). No other records of *M. racemosa* outside its native range have been identified.

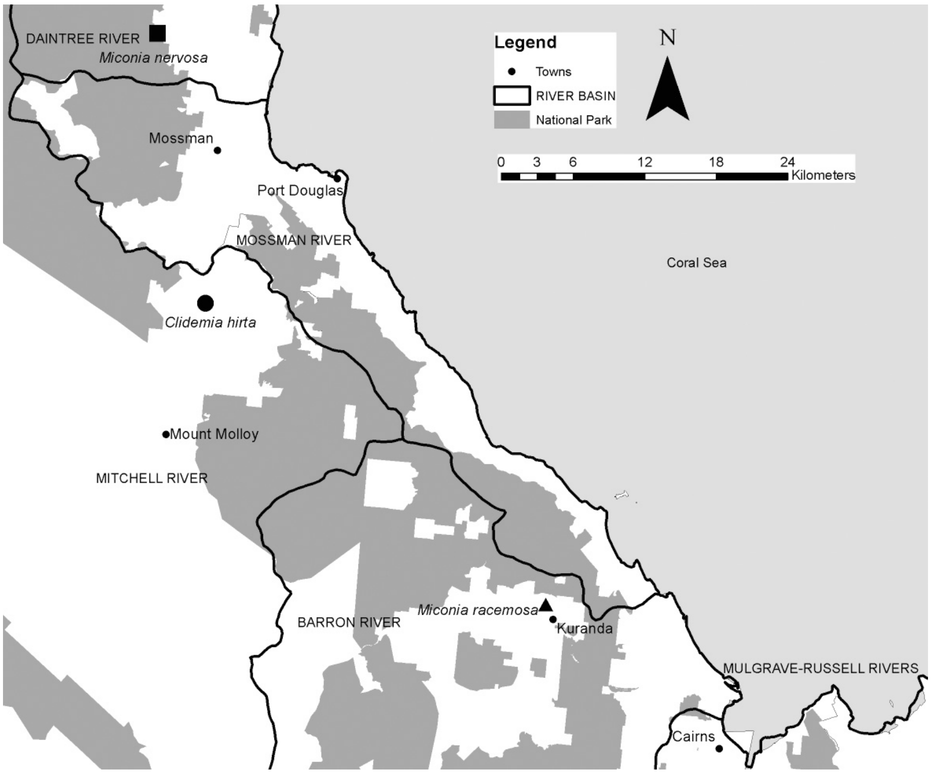

Figure 1. Locations of infestations of *Clidemia hirta*, *Miconia nervosa* and *M. racemosa* in northern Queensland.

In Queensland all *Miconia* spp. and *C. hirta* are declared Class 1 weeds under the *Land Protection (Pest and Stock Route Management) Act 2002*. All three cannot legally be introduced in the Northern Territory or Western Australia. *Miconia* spp. are also banned from introduction into NSW and the three species have no known agricultural use and few horticulturally attractive characteristics. Eradication programs are more likely to succeed where reintroduction is prevented (Simberloff 2002) so legislative barriers and enforcement are important steps to limit the propagation of the species.

These species are eradication targets under the nationally cost shared 'Four Tropical Weeds' Eradication Program which commenced in 2001 and is currently managed by Biosecurity Queensland (Department of Primary Industries and Fisheries). Under this program expenditure, including field resources, management and extension activities, is spread across six species from four genera, with field assistance from local government officers, National Parks Rangers, researchers and other stakeholders (Erbacher *et al.* 2008). In-kind (37% of person days) and program field resources (63% of person days) were combined to calculate the total field effort in Table 1, but the melastome shrubs accounted for approximately 20% of the eradication program's resources or $350 000 in total from June 2002 to June 2007. The shrubs have been found predominantly in forested situations that are not regularly managed by landowners. While the three species are known only at single locations in Australia, there have been continuing efforts to prevent or detect any other infestations. However, as the precise method of entry of the three species into Australia has not been verified, the programs are continuing to address the risk of undetected infestations through maintaining public and professional awareness and trace back and forward activities (Brooks and Galway 2008).

Materials and methods

Data acquisition

Information on the date of discovery, location, area and monitoring of each infestation of each species was sourced from notes, field records and reports from the Department of Primary Industries and Fisheries (DPIF), Department of Natural Resources and Water (NRW), Commonwealth Scientific Industry Research Organisation (CSIRO), Australian Quarantine Inspection Service (AQIS) and local government staff. Records on infestation management were acquired for each infestation for between 2001 and 2007. For each year the total cumulative infested area and total search area were calculated for each species. Some portions of the total search area were visited two to four times per year. The infested area was defined by all recorded GPS points plus a 5 m buffer, with the overlap removed to determine a net infested area (Rejmánek and Pitcairn 2002). The 5 m buffer accounted for variation in GPS coverage under tropical forests and high recruitment densities around single points. From 2004 onward the numbers of juvenile and reproductive plants were recorded. All plants that were suspected as having seeded (on the basis of plant size) or were observed flowering or seeding were recorded as reproductive.

Search strategies

Survey strategies utilized for these species were defined as either 'core' or 'outlier'. From 2005 onward core surveys covered the location of reproductive individuals plus a dispersal buffer with a 500 m radius. This encompassed the net area plus the dispersal buffer, to form a gross infested area (Rejmánek and Pitcairn 2002), where most of the suitable habitat was searched at least once each year. Within the core, dense areas of recruitment were visited every three months (Graham *et al.* 2008), while it took several years to survey all potential habitats in the dispersal buffer. This introduced a risk that undetected plants could reproduce in the interim. Unsuitable habitats included annually cultivated crops and mown lawn. Surveys have shown forested areas near creek lines are a more common habitat for these shrubs, so some additional outlier surveys beyond the 500 m buffer were also conducted, concentrating on streamside habitats.

During October 2006 and 2007 additional staff from CSIRO, Queensland Parks and Wildlife Service (QPWS) and local government collaborators spent a week with eradication program staff surveying and controlling core and satellite areas of the three melastome shrub infestations. These 'taskforce' events brought additional search and mapping resources to bear while genetic, demographic and dispersal studies were being undertaken. This resulted in a greater accounting and physical control of small seedlings in the last two years, particularly in 2007. In previous control operations, small dense carpets of minute seedlings were either foliar sprayed or left for three months to self thin. However, in 2007, an additional 732 *C. hirta*, 17 095 *M. racemosa*, and 585 *M. nervosa* seedlings with 1 or 2 mm basal diameters were recorded, compared with numbers in 2006 (Table 1). The greater intensity of sampling of small seedlings in 2006 and 2007 was undertaken to gather material for DNA extraction and analysis of within- and between-site genetic diversity, and to estimate effective dispersal (Hardesty, in press). Focussing larger groups of people on extended survey activities increased the search efficiency and total areas covered in 2006 and 2007 (Table 1).

Field surveys were conducted on foot, with new survey areas first accessed along creeks and ravines, followed by systematic parallel field surveys with operators evenly spaced along contours. Operators at either end of the line steered the survey using flagged tracks, creek lines or compass bearings as boundaries, with string lines used in dense vegetation to mark boundaries of consecutive survey runs. Search areas were clearly defined by access tracks, ridgelines and watercourses, with superficial tracks established to access all of the gross area. Resources used to establish landmarks and other survey infrastructure are included in the work effort (Table 1). Search speed could be as low as <100 m h^{-1} due to topography and the density of the rainforest vegetation, but tended to average around 250 m h^{-1}. Plants were removed by hand or, less commonly, small dense areas were subjected to a foliar spray of glyphosate (Sydes and Galway 2008).

Assessing delimitation

We applied the delimitation formula of Panetta and Lawes (2007) to determine the performance of the eradication programs in relation to the delimitation criterion. For each infestation the value for D for year n was calculated according to the following equation:

$$D_n = \frac{A_d}{P_n + \log(A_s+1)}$$

The variable A_d represents the area of infestation newly detected in year n, P_n represents the proportional change in

Table 1. Areas of melastomes, work effort and plant counts where available.

Species	Year	Area searched (ha)	Cumulative net area detected (ha)	Juvenile plants	Reproductive plants	Work effort (person days)
Clidemia hirta	2001	40	3.1	[A]		104
	2002	60	3.2			70
	2003	60	3.4			49
	2004	85	3.5	1 043	20	58
	2005	134	3.9	1 946	1	95
	2006	364	4.1	2 155	8	122
	2007	256	4.2	4 248	9	73
Miconia racemosa	2002	25	0.2	[B]		39
	2003	35	0.3			39
	2004	25	0.3	2 000	20	12
	2005	22	0.4	2 860	3	20
	2006	88	0.5	3 181	5	29
	2007	70	0.5	17 396	1	36
Miconia nervosa	2004			[B]	10	
	2005	21	0.7	264	1	36
	2006	85	1.5	695	6	60
	2007	99	1.6	1 203	4	61

[A] 300–400 plants, many mature.
[B] 100–200 plants.

total infested area between year $n-1$ and year n, and A_s represents the area that is searched in year n. The value of D was log transformed to increase sensitivity of the measure as D approaches 0. This is because delimitation efforts may fail because new detections continue to be made, at a low level but indefinitely, following the identification of the major portion of an incursion (Panetta and Lawes 2007). The eradication program has also undertaken activities to help verify that these species are present only at single locations in Australia.

Assessing extirpation

Where infestations are in the control phase, ongoing survey and control efforts aim to prevent reproduction so as to eventually exhaust the soil seed bank; progress towards the latter would be indicated by declining plant numbers over time (Rees and Long 1993). Accordingly, we examined trends in both the numbers of reproductive individuals ('reproductive escapes') and juvenile plants over the course of the programs. It is not possible to generate an extirpation score as per Panetta and Lawes (2007) for single infestations. Moreover, regardless of how infestations are defined (and this is often problematic), infestations of all three species were in the active control phase (Panetta and Lawes 2007), so the extirpation score would not be particularly informative.

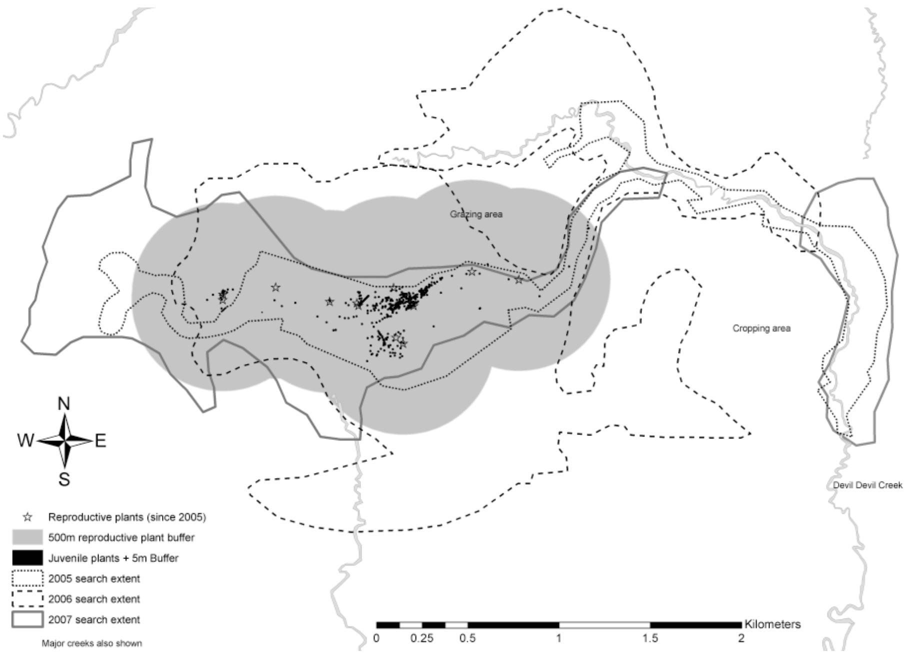

Figure 2. Extent of recent annual searches and location of juvenile and reproductive *Clidemia hirta* plants at an infestation in northern Queensland, as of December 2007.

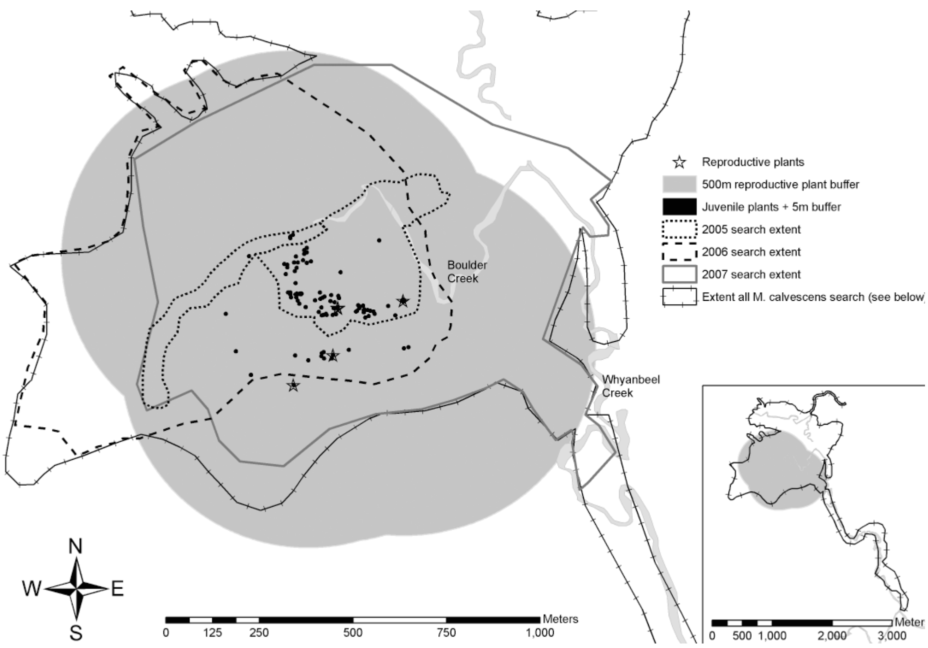

Figure 3. Extent of recent annual searches and location of juvenile and reproductive *Miconia nervosa* plants at an infestation in northern Queensland, as of December 2007.

Results

Delimitation

Extensive field surveys (Figures 2–4) and landholder information have identified the original points of introduction within the infestations. These are the areas of highest plant density and correspond to the initial net infested area. The identification of this original area is reflected in the large proportion of net areas that were detected in the first year or two of surveys (Table 1) and the associated high delimitation scores (Figure 5). A large increase in search areas in 2006 resulted in very small increases in net areas across all three species. A stabilization of the cumulative net area is an encouraging sign in the delimitation of these species, especially *M. racemosa*, where no new areas were discovered in 2007 despite a similar detection effort. At all three infestations adjacent landholders have been contacted repeatedly, shown photos or display specimens, had properties surveyed and been informed about the eradication programs activities.

Increasing the *C. hirta* search area to 363 ha in 2006 helped to verify the extent of the infestation as no further outlying patches were discovered (Figure 2). Prior to 2006 much of the survey activity for *C. hirta* followed forested vegetation downstream (east) of the infested area, but subsequent upstream and outlier surveys in 2006 identified small infested areas 700 m upstream (west) of the original infestation, probably resulting from dispersal by frugivores. This search effort covered much of the suitable forested area in the 500 m dispersal buffer and some additional areas, though further surveys to the west and south of the core area are planned. Some core areas to the north and south of the densest infested areas are less suitable, but easily searched, grazing land. Potted display specimens of *C. hirta* and relevant information were presented at the local Julatten Hall and other extension activities sought to trace forward any potential local spread of contaminated material (S. Clayton personal communication 2007).

Core surveys have also identified three new small patches of *M. nervosa*, as well as the 10 reproductive individuals in the 2006 and 2007 records. To date the furthest patch has been found 496 m up hill from the origin of the infestation. The current net infested area for *M. nervosa* is 1.6 ha, with 70–99 ha searched for this species in 2006 and 2007. An additional 112 ha was searched at the same location for *M. calvescens* (Figure 3), another species that is currently targeted under the nationally cost shared 'Four Tropical Weeds' Eradication Program. The *M. nervosa* infestation may have originated from a cultivated specimen, possibly from a local source; this source and possible links with other *M. calvescens* populations have been traced with no further infestations found.

While the source of the *M. racemosa* is also unknown, extension activities and media coverage have targeted local landholders. Surveys for *M. calvescens* over up to 6 km^2 around the town of Kuranda by the same eradication team and collaborators (P. Lawler personal communication 2004) covered many gullies and an additional 44 ha in the Myola area in 2004 (Figure 4). Search efforts during 2007 included a canoe survey for both miconia species and *Hymenachne amplexicaulis* (Rudge) Nees. along the Barron River. Core surveys also identified an additional *M. racemosa* patch in 2006 (five reproductive individuals) in a more open, grassed upstream environment to the west of the initial densely infested area (Figure 4). Additional areas downstream (south east to the Barron River) of the dense infestation are also searched, as scattered plants have been found close to the gully since 2002.

Although Figure 5 and Table 1 show a broad positive trend towards delimitation of all three species, two processes have resulted in the detection of reproductive plants and additional net areas in the last three years. Firstly, expansion of the core surveys has detected outlying patches that appear to have resulted from bird dispersal as discussed above. Secondly, plants that were possibly missed in previous annual surveys have been detected on the margins of the initial infestations. This second process is discussed below in relation to the frequency of surveys and the extirpation criterion.

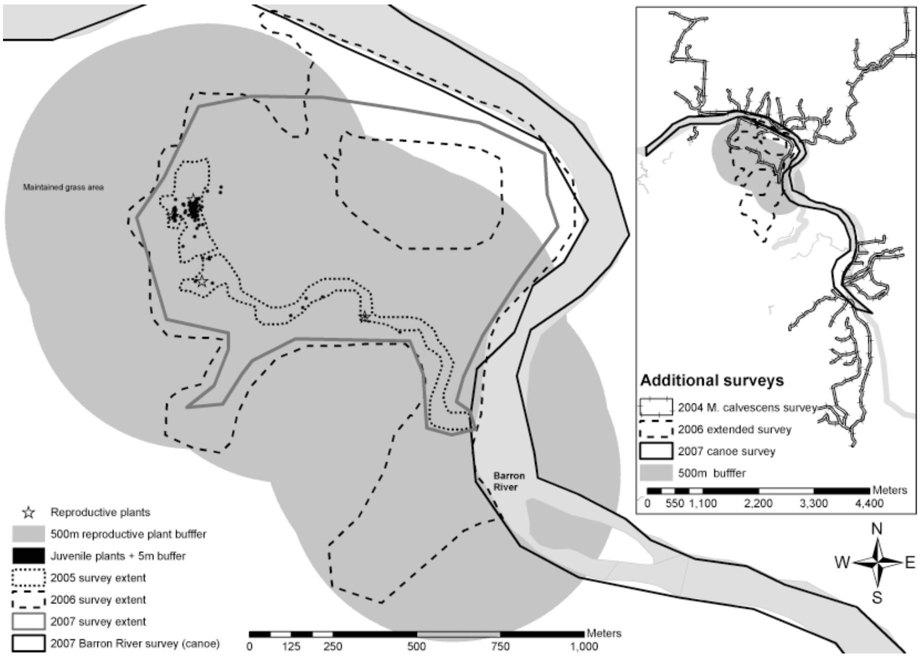

Figure 4. Extent of recent annual searches and location of juvenile and reproductive *Miconia racemosa* plants at an infestation in northern Queensland, as of December 2007. Much of the area to the west of the infested area is maintained lawn.

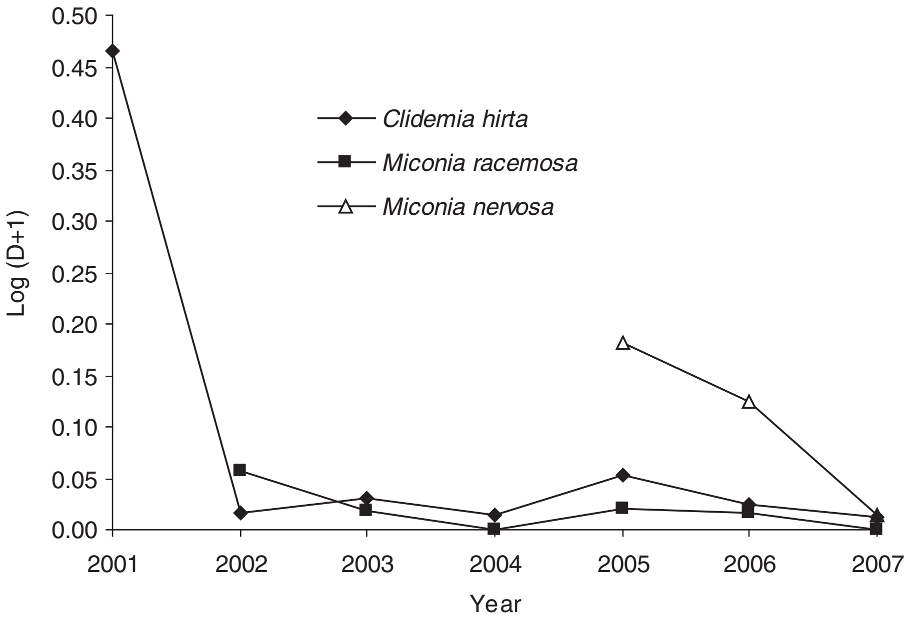

Figure 5. Temporal trends in the delimitation measure (*D*) for *Clidemia hirta*, *Miconia nervosa* and *M. racemosa*.

Extirpation

The total number of plants controlled is listed in Table 1. As noted earlier, the taskforce activities led to a greater accounting of seedlings in 2006 and 2007. Juvenile plant counts (Figure 6a–c) indicate that despite some variation in rainfall, even the drier years (2002, 2003 and 2005) have been conducive to germination and seed production events. The discovery of 'a few hundred' *C. hirta* plants in 2001 and the two years of relatively dry conditions that followed may have contributed to an underestimation of the size of this infestation. Continued emergence in the initial infested area, with no recent local reproductive input, indicates that seed persists in dense *C. hirta* infestations for at least six years.

The detections of reproductive plants of *C. hirta* in 2007 and of *M. racemosa* in 2004, 2005 and 2007 were within areas previously searched. The *C. hirta* plants occurred on a creek bank area of variable width (where parallel search tracks had become widely spaced) in an area outside the regularly visited core area.

The occurrence of *M. racemosa* reproductive plants in 2004, 2005 and 2007 resulted from lapses in the frequency and extent of surveys beyond the initial infested area with changes to program staffing, as well as a lower search effort in 2004. Much of the seedling emergence noted in Table 1 and the higher level of seedling emergence recorded in 2007 occurred in the initial infested area. Although there has been one plant (2007, missed by the 2006 taskforce) with fruit nearing maturity in this area since 2004, continued seedling emergence illustrates that an active and persistent seed bank has developed from the reproductive plants present in 2004 and prior to discovery.

The reproductive *M. nervosa* plants detected in 2006 and 2007 have been found in newly searched areas.

Variation in the counts of non-reproductive plants was due partly to variations in both detection and recording, with more inclusive counting of all plants in 2006 and 2007 taskforce events. For all three species the initial infested areas accounted for over 90% of plants and remain active zones of seedling emergence 2–6 years after discovery.

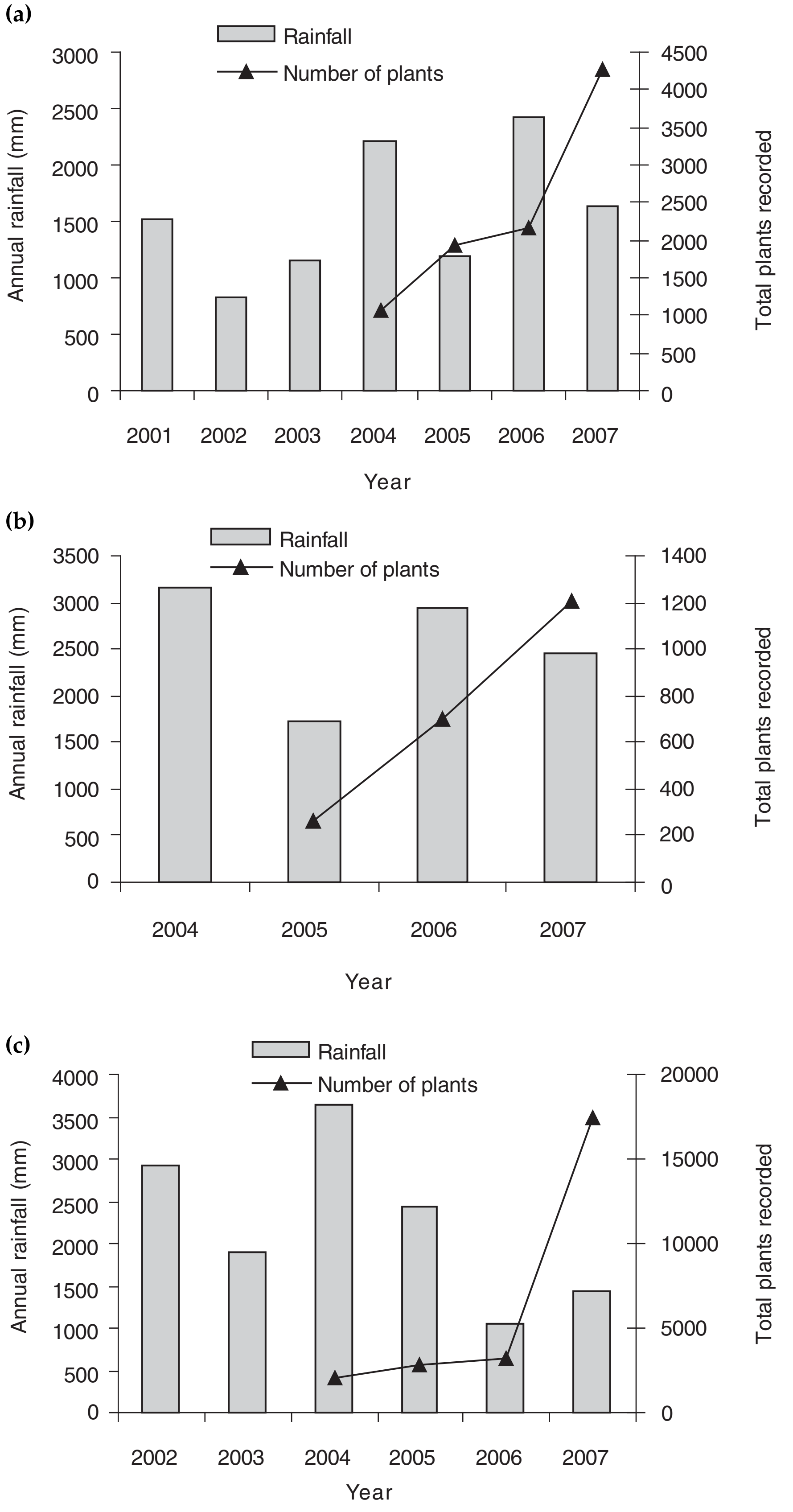

Figure 6. Seedling emergence of (a) *Clidemia hirta*, (b) *Miconia nervosa* and (c) *M. racemosa* in relation to annual rainfall (State of Queensland 2000).

Discussion

Prior to 2004 searches tended to focus on downstream habitats. The recent increase in up-steam search effort has detected small patches of all three melastome shrubs; these appear to have resulted from dispersal by avian frugivores. Most specimens have been detected in the vicinity of creeks. Presumably these have arisen more as a result of dispersal along frugivore flight paths across more favourable shaded, humid microsites than dispersal by water. To date the strategy of searching a 500 m radius from mature plants, in conjunction with some additional, suitable habitats, has proven effective, with other searches at Julatten for *C. hirta* and those for *M. calvescens* at the other two sites beyond this range not yet detecting additional plants. However, this radius may be increased if information, similar to that for *M. calvescens* (possible dispersal up to 1100 m; Murphy *et al.* 2008) is applied to these shrubs, if plants are found in outlier surveys beyond 500 m or as program resources allow.

A number of detections of reproductive plants, particularly for *C. hirta* and *M. racemosa*, were within areas that had been previously searched. One option to address this issue includes the allocation of more resources to the 3 or 6 monthly surveys, rather than the current dependence on a large annual survey (taskforce) to cover all the less densely infested areas. Local research has shown that *C. hirta* can reach maturity in a year (Graham *et al.* 2008), so plants missed in one year can mature before the next annual survey; increased search frequencies would increase the opportunity to detect plants before they reached maturity. Although searches are conducted using landmarks and compass bearings, searching of winding creek banks during taskforce events may benefit from greater subdivision and demarcation to improve search intensity and coverage.

Progress towards eradication

Delimitation is the fundamental criterion for eradication success (Panetta and Lawes 2005). There is no evidence that these species were sold through nurseries and there are legal barriers to their cultivation. However, it is important that efforts to determine the existence of other infestations are maintained for some years to manage the risks of any additional, unmanaged infestations or of further introductions occurring. Data from the melastome shrub eradication programs demonstrate that the delimitation measure of Panetta and Lawes (2007), originally designed for the purpose of assessing delimitation of a weed incursion comprised of multiple infestations, is equally applicable to the assessment of an incursion confined to a single point of introduction. In the present cases the main areas of infestation were determined in the first year or two after discovery and subsequent new areas are small relative to the original infested area.

If the infestations are not contained then new infestations could arise through links to the original infestation, or populations could appear in areas previously searched. Either scenario would result in an increase in the area requiring treatment, as well as the possible occurrence of reproductive plants. In addition to the delimitation activities, practical containment measures adopted by the programs include implementing property management plans to limit off-site movement of potentially contaminated materials, on-site quarantine arrangements, maintaining weed seed hygiene protocols amongst field personnel and equipment (Sydes and Galway 2008) and the outlier search strategy.

All infestations are currently in an active control phase, with control operations periodically removing seedlings and juvenile plants to run down the soil seed bank. This contrasts with the situation for two other species (*Mikania micrantha* Kunth (Asteraceae) and limnocharis (*Limnocharis flava* (L.) Buchenau) (Limnocharitaceae)) currently targeted for eradication in the Australian wet tropics. For both of these weeds the majority of infestations are in the monitoring phase (Brooks *et al.* 2008). Seedling emergence (presence or absence) will be an important indicator of soil seed bank persistence in the future, with assessment intensity and search strategies explaining much of the current variation in plant counts. Local and overseas sources indicate a persistent *C. hirta* soil seed bank. While there are no available data on seed persistence for the *Miconia* spp., it would be reasonable to expect emergence for several years to come, which indicates that these infestations may not transition to the monitoring phase (Panetta 2007) for some time. Seeds of the congeneric *M. calvescens* are known to persist for more than 14 years (Meyer 2008). Although the net infestation areas are small, all infestations will require an investment in excess of 30 person days per year for the length of the control and monitoring phases.

With little known about the biology and dispersal of the miconia shrubs under local conditions, it can take considerable time and resources to determine the extent of each infestation and the best ways to manage it. The miconia shrubs are not known outside South America but their mass germination on steep disturbed banks, establishment in dense shade and higher light environments, and dispersal up to 500 m into the rainforest, highlight their invasive capacity under local conditions. Similarly *C. hirta* has proven a highly persistent, elusive and resource-demanding eradication target in the seven years since discovery. These biological characteristics and the difficulties relating to both species detectability (see Cacho *et al.* 2006) and the prevention of reproduction in tropical forested environments suggest that should more (or larger) infestations of these species be detected, eradication on a national scale would be unlikely.

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