Les effets de l'ouverture de sentiers sur l'invasion des zones protégées de l'île de la Réunion (Archipel des Mascareignes, Océan Indien) par des espèces végétales exotiques

Stéphane Baret, Dominique Strasberg

Revue d'Écologie (La Terre et La Vie)

The effects of opening trails on exotic plant invasion in protected areas on La Réunion Island (Mascarene archipelago, Indian Ocean)

Stéphane Baret, Dominique Strasberg

Abstract
Exotic species have become a major problem in protected areas. A better knowledge of ecosystem functioning is necessary to improve management practice that facilitates habitat conservation. We suggest that small-scale disturbances such as trail opening and the effects on exotic plants invasions are underestimated. We have, therefore, quantitatively measured the effect of forest trail building on invasive plant colonization, a phenomenon in constant progression but never studied before. Our study shows that *Rubus alceifolius*, the widespread exotic invasive plant on La Réunion, is able to germinate on both narrow and wide trails, but persists only on the wider ones. Our results show moreover that wide forest trails favored establishment of exotics and significantly decreases species richness. We suggest to forest managers that building narrow forest trails is preferable as it will stop the persistence and propagation of *R. alceifolius* and other non-indigenous plants.

Résumé
Les effets de l'ouverture de sentiers sur l'invasion des zones protégées de l'île de la Réunion (archipel des Mascareignes, océan Indien) par des espèces végétales exotiques. — L'envahissement par les espèces exotiques est devenu un problème majeur au niveau des aires protégées. Une meilleure connaissance de leur impact sur le fonctionnement des écosystèmes est nécessaire pour améliorer les pratiques de gestion et la conservation des habitats. Nous suggérons que les effets des perturbations comme l'ouverture de chemins forestiers et l'invasion par les plantes exotiques sont sous-estimés. Nous avons donc quantitativement mesuré le rôle de l'ouverture de chemins forestiers dans la colonisation par les plantes envahissantes, un phénomène en progression constante mais jamais étudié auparavant. Notre étude montre que *Rubus alceifolius*, la plante exotique envahissante la plus répandue à La Réunion, est capable de germer aussi bien au niveau d'un chemin étroit que d'un chemin large mais ne persiste qu'au niveau du chemin le plus large. Nos résultats montrent aussi qu'un chemin large favorise l'établissement des plantes exotiques et détermine une réduction significative de la richesse spécifique. Cette étude conduit à suggérer aux forestiers de construire plutôt des chemins étroits, ce qui permettrait d'empêcher l'installation et la propagation de *R. alceifolius* et des autres plantes non indigènes.

THE EFFECTS OF OPENING TRAILS ON EXOTIC PLANT INVASION IN PROTECTED AREAS ON LA RÉUNION ISLAND (MASCARENE ARCHIPELAGO, INDIAN OCEAN)

Stéphane BARET[1] & Dominique STRASBERG[1]*

RÉSUMÉ. — *Les effets de l'ouverture de sentiers sur l'invasion des zones protégées de l'île de la Réunion (archipel des Mascareignes, océan Indien) par des espèces végétales exotiques.* — L'envahissement par les espèces exotiques est devenu un problème majeur au niveau des aires protégées. Une meilleure connaissance de leur impact sur le fonctionnement des écosystèmes est nécessaire pour améliorer les pratiques de gestion et la conservation des habitats. Nous suggérons que les effets des perturbations comme l'ouverture de chemins forestiers et l'invasion par les plantes exotiques sont sous-estimés. Nous avons donc quantitativement mesuré le rôle de l'ouverture de chemins forestiers dans la colonisation par les plantes envahissantes, un phénomène en progression constante mais jamais étudié auparavant. Notre étude montre que *Rubus alceifolius*, la plante exotique envahissante la plus répandue à La Réunion, est capable de germer aussi bien au niveau d'un chemin étroit que d'un chemin large mais ne persiste qu'au niveau du chemin le plus large. Nos résultats montrent aussi qu'un chemin large favorise l'établissement des plantes exotiques et détermine une réduction significative de la richesse spécifique. Cette étude conduit à suggérer aux forestiers de construire plutôt des chemins étroits, ce qui permettrait d'empêcher l'installation et la propagation de *R. alceifolius* et des autres plantes non indigènes.

SUMMARY. — Exotic species have become a major problem in protected areas. A better knowledge of ecosystem functioning is necessary to improve management practice that facilitates habitat conservation. We suggest that small-scale disturbances such as trail opening and the effects on exotic plants invasions are underestimated. We have, therefore, quantitatively measured the effect of forest trail building on invasive plant colonization, a phenomenon in constant progression but never studied before. Our study shows that *Rubus alceifolius*, the widespread exotic invasive plant on La Réunion, is able to germinate on both narrow and wide trails, but persists only on the wider ones. Our results show moreover that wide forest trails favored establishment of exotics and significantly decreases species richness. We suggest to forest managers that building narrow forest trails is preferable as it will stop the persistence and propagation of *R. alceifolius* and other non-indigenous plants.

There are many alien plant species that threaten natural areas (Lockwood *et al*., 2001) or protected areas (Coles & Landres, 1996) because they are able to naturalize or become invasive (invasive status defined according to Richardson *et al*., 2000) by replacing native flora or even changing ecosystem properties (Vitousek, 1990; Gordon, 1998). For example, D'Antonio & Meyerson (2002) recently reviewed management problems caused by exotic plants in protected areas and their impacts on restoration projects. Such issues are amplified on oceanic islands (Vitousek, 1988; Brockie *et al*., 1988). In the Mascarene archipelago (Indian Ocean), native habitat remnants are threatened with extinction by exotic plant invasions (see Macdonald *et al*., 1991; Strasberg *et al*., 2005). In this archipelago (formed by La Réunion, Mauritius and Rodrigues), tropical rainforests are biologically rich but have

[1] UMR C53 CIRAD/Université Réunion, Peuplements Végétaux et Bio-agresseurs en Milieu Tropical, Université de la Réunion, 15 avenue R. Cassin, BP. 7151, 97415 Sainte-Clotilde Messag. Cedex 9, la Réunion, France
* *Author for correspondence* (e-mail: Dominique.Strasberg@univ-reunion.fr)

Rev. Écol. (Terre Vie), vol. 60, 2005.

been radically reduced and fragmented (Lorence & Sussman, 1988; Olson & Dinerstein, 1998) by human activities such as farming and settlements.

In La Réunion Island (the most preserved island of the Mascarene archipelago), 70% of the 675 native species are restricted to Mascarene and 30% endemic of La Réunion (see Lavergne *et al*., 1999 for a synthesis). However, since human settlement on La Réunion in the 1600s, over 2000 introduced plants have been recorded of which 62 are invasive (MacDonald *et al*., 1991; Lavergne *et al*., 1999). Management of exotic plants is therefore a priority for conservation of the highly endangered Mascarene habitats (Thébaud & Strasberg, 1997; Mungroo *et al*., 1998). In order to preserve biodiversity, it is essential to predict the outcome of forest management practices and the impact of invasive species on Mascarene ecosystems. Although major human disturbances have been reduced in Mascarene rainforests following the implementation of several protected areas (Gargominy, 2003), exotic plant density is still increasing in some protected areas, especially on La Réunion Island (Lavergne *et al*., 1999). We suggest that negative effects of small-scale disturbance such as trail opening have been underestimated and may facilitate exotic plant invasions.

Several studies have already demonstrated the effect of openings (gaps, landslides, lava flows) in the habitat on the presence and abundance of invasive plants (e.g. Strasberg, 1995; King & Grace, 2000; Restrepo & Vitousek, 2001). Other studies also revealed that a high number of invasive species grow along forest roadsides, and that their abundance decreases gradually with the decline in light intensity with penetration into the forest understorey (Amor & Stevens, 1976; Harrison *et al*., 2002). Lonsdale & Lane (1994) and Gelbard & Belnap (2003) suggested that roads act as corridors, and that vehicles may act as vectors of propagules, favoring the spread of invasive exotic species. However, no studies to date have measured the effect of forest trail building on invasive plant establishment, colonization and persistence. In this study, we assessed the effects of trail opening on the colonization patterns of invasive plants in a lowland rainforest protected area on La Réunion (21°00'S, 55°30'E). The aim of the study was to (1) measure the effect of trail width on the dynamics of the widespread exotic plant, *Rubus alceifolius*, and (2) determine the extent of establishment of all alien plants along trails and within the neighbouring forest.

MATERIAL AND METHODS

The study was carried out in a remnant of tropical lowland rainforest at Grand-Étang (550 m a.s.l.), a protected area on the east coast of La Réunion Island. Mean annual rainfall is high (6 307 mm/year) and the mean annual temperature is 20.4°C (Météo France meteorological data for the 1969-1999 period). Our study was situated in a flat area of 10 ha with ± 90% forest cover and 10% trails and gaps in the canopy. Recently, the Protected Areas Management created two trails of different widths (5-10 m and 1-3 m, average widths were of 7.15 ± 0.51 and 2.15 ± 0.20, n = 10 respectively) that were opened on 18 October 1998. All plants were manually and mechanically eradicated on the trails on this date. *Rubus alceifolius* seedlings appeared rapidly after trail clearing. As it was outside the fruiting time of *R. alceifolius* (between April and July at this elevation, Baret *et al*., 2004), we confirm that all seeds germinating after the trail development (October to November) came from seeds already present in the soil before this date. The leaf area index (LAI, calculated according to the method described by Cournac *et al*., 2002) was measured to assess the canopy gap size and have an idea of light penetrating the understorey. A control measurement was made at the nearest open area.

We measured the effects of trail width on *Rubus alceifolius* Poiret, a native of Southeast Asia and introduced to La Réunion in the 1840s (Jacob de Cordemoy, 1895). This species is present in monospecific patches in the protected area of Grand-Étang where a substantial seed-bank has accumulated (Baret *et al*., 2004). Five months after the trails were opened (in March 1999), *R. alceifolius* plants were found over the full width of the trails. No *Rubus* seedlings were encountered before the trails were opened. Plant growth (main stem length) and the number of nodes produced by each individual along the main stem were measured every 3 weeks over a 4 month period (March-June 1999). To offset variation in size and compare the length of the main stem over time and the number of internodes formed, we differentiated plants that were less than 3 cm long and more than 3 cm long at the beginning of the monitoring period (March 1999). Each individual was monitored in September 1999 and during the 2000 and 2001 fruiting season (April to June, see Baret *et al*., 2004) to record mortality and fruit development.

Five years after the trails were built (in June 2003), we established sixteen 10 m transects perpendicular to the wide trail at 4 randomly selected sites (4 transects at each site). Along each of these transects, we marked 2-plots, one on the trail edge and the other 10 m into the understorey. Using the point-centered quarter method (developed by Cottan & Curtis, 1956) 8 plant species were recorded in each of the marked plots (see Fig. 1). One hundred and twenty eight established plants (> 1 m high) were documented (64 plants in the trail and 64 plants in the understorey).

Mann-Whitney and Chi-square tests on Statistix 7 software were performed to compare *Rubus* growth and exotic/indigenous frequency.

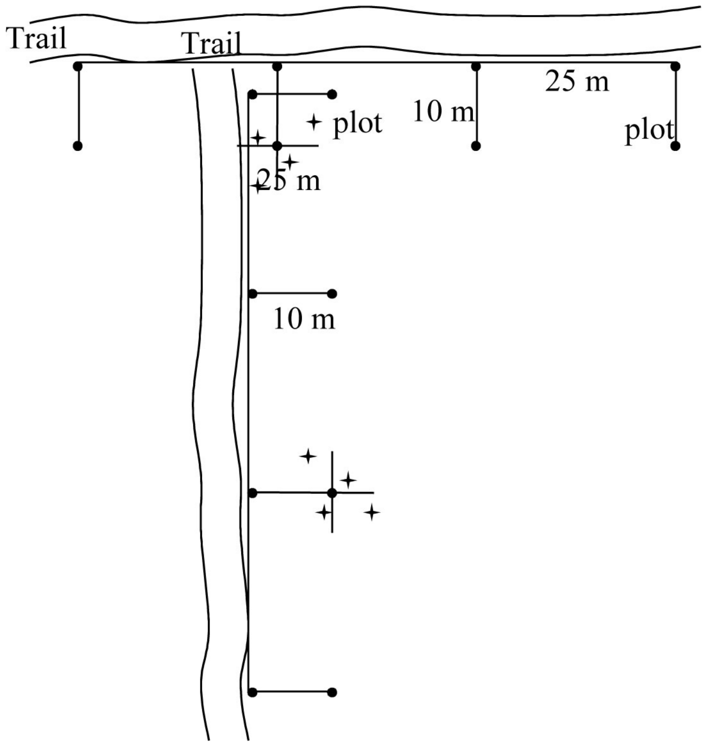

Figure 1. — Schematic representation of plant species recorded along trail edges and understorey. For each plot (black circle), 4 plant species were recorded according to the point quarter method (from Cottan & Curtis, 1956). Stars represent individuals observed at one plot. Sixteen transects were done at 4 different sites along the trail (4 transects by site). Thus, 128 individuals were recorded (64 along trail edge and 64 understorey, see table I).

RESULTS

Establishment, growth and length of juvenile period of *R. alceifolius*

No *R. alceifolius* germination was observed in the understorey. However, 37 *R. alceifolius* plants were detected within a 50 m long section of both trails 5 months after they were opened (March 1999), even though no plants were present at opening. Twenty-seven of these plants (including 12 < 3 cm long) were found along the wide trail (LAI from 1.08 ± 0.06) and 10 (all < 3 cm long) along the narrow trail (LAI from 2.1 ± 0.06). The growth (length of the main stem) of these plants was monitored from March to June 1999. Five months after opening the forest canopy (March 1999), mean stem length was significantly higher along the wide trail (7.0 ± 1.73 cm long, with 9.4 ± 0.57 internodes) than along the narrow trail (1.6 ± 0.25 cm long, with 6.0 ± 0.83 internodes). Over the 4-month monitoring period, we noted significant differences in stem growth per 21 days between all plants growing at both sites and also for most plants that were under 3 cm long (Tab. I). Significant differences were also noted with respect to the number of nodes created between each monitoring date (Tab. II). In September 1999, all plants growing along the narrow trail had wilted. Twenty-one plants were still alive along the wide trail, and the five most developed plants bore fruit during the 2000 fruiting season (April to June) and 15 others during the 2001 season.

TABLE I

Growth of the main stem (mean in cm ± SE) of Rubus alceifolius *plants per 21 days (between 5 and 8 months after trail building) and according to the trail width*

Time (days)	Wide trail (5-10 m)		Narrow trail (1-3 m)
	All plants	Plants < 3 cm	All plants < 3 cm
t + 21	2.36 ± 0.71 [a]	0.37 ± 0.12 [A]	0.20 ± 0.06 [b, A]
t + 42	2.31 ± 0.65 [a]	0.46 ± 0.25 [A]	0.10 ± 0.05 [b, B]
t + 63	1.59 ± 0.48 [a]	0.26 ± 0.13 [A]	0.06 ± 0.03 [b, B]
t + 84	1.24 ± 0.45 [a]	0.29 ± 0.17 [A]	0.02 ± 0.02 [b, B]
t + 103	0.84 ± 0.31 [a]	0.18 ± 0.06 [A]	0.03 ± 0.02 [b, B]
Total	8.14 ± 11.16 [a]	1.56 ± 0.70 [A]	0.40 ± 0.12 [b, B]

A Mann-Whitney test was used to compare plants growing along the 5-10 m wide trail with those growing along the 1-3 m wide trail. This comparison involved all plants (lower case letters a and b) growing along the wide trail, and then those less than 3 cm long (capital letters A and B). A change of letters between means indicates that for each time period considered (i.e. 21 days) the corresponding distribution is significantly different at the 99% threshold.

TABLE II

Number of new nodes (mean in cm ± SE) created by Rubus alceifolius *plants per 21 days (between 5 and 8 months after trails creation) and according to the trail width*

Time (days)	Wide trail (5-10 m)		Narrow trail (1-3 m)
	All plants	Plants < 3 cm	All plants < 3 cm
t + 21	1.18 ± 0.14 [a]	1.00 ± 0.15 [A]	0.30 ± 0.15 [b, B]
t + 42	1.19 ± 0.09 [a]	1.20 ± 0.20 [A]	0.50 ± 0.22 [b, B]
t + 63	0.92 ± 0.08 [a]	0.80 ± 0.13 [A]	0.40 ± 0.16 [b, A]
t + 84	1.04 ± 0.10 [a]	1.00 ± 0.00 [A]	0.40 ± 0.22 [b, B]
t + 103	0.70 ± 0.10 [a]	0.50 ± 0.17 [A]	0.30 ± 0.15 [b, A]
Total	4.57 ± 0.27 [a]	4.50 ± 0.34 [A]	1.90 ± 0.55 [b, B]

The statistical analyses were the same as those described in table I.

EXOTIC *VS*. INDIGENOUS PLANT ESTABLISHMENTS: COMPARISON BETWEEN TRAILS AND UNDERSTOREY

Preliminary measurements of LAI give meaning of abbreviations the first time used (n = 10 for each site) differ in wide trails (1.08 ± 0.06), understorey (4.53 ± 0.01) and in the open area (0.33 ± 0.01). Transects along the wide trail revealed an important contrast between established plants in the forest trail (82.8% exotic *vs*. 17.2% native plants) and the surrounding understorey (12.5% exotic *vs*. 87.5% native plants) (see Tab. III). Exotic plants were more frequent along the trail edges than in the understorey ($P < 0.0000$, $\chi^2 = 63.42$). Species richness was also different. In a sample of 64 individuals, twenty plant species were recorded in the understorey (Tab. III), only one being exotic, *Psidium cattleianum* (Tab. I). In a sample of 64 individuals along trail edges, 11 plant species were recorded (5 indigenous and 6 exotic), and exotic plant individuals outnumbered indigenous ones (Tab. III).

TABLE III

Exotic and indigenous plant species recorded along trail edges and understorey (64 individuals recorded in each case), according to the point quarter method (see text)

Species	Number of individuals	
	Trail edges	Understorey
EXOTICS		
Ardisia crenata	4	0
Boehmeria penduliflora	12	0
Lantana camara	2	0
Ludwigia octovalvis	1	0
Psidium cattleianum	16	8
Rubus alceifolius	18	0
INDIGENOUS		
Acantophoenix rubra	0	1
Antirhea borbonica	0	2
Aphloia theiformis	0	1
Badula nitida	0	2
Casearia coriacea	0	7
Coffea mauritiana	0	1
Cyathea borbonica	0	3
Cyathea excelsa	0	1
Cnetis glabra	0	2
Ficus mauritiana	2	1
Gaertnera vaginata	0	20
Geniostoma borbonicum	6	0
Habenaria cyginum	0	2
Labourdonnaisia callophylloides	0	1
Memecylon confusum	0	1
Molinea alternifolia	0	1
Pandanus purpurascens	0	6
Pittosporum senacia	0	1
Scleria sp.	1	0
Sticherus linearis	1	0
Tambourissa elliptica	0	1
Weinmannia tinctoria	1	2
Total of exotic individuals	53	8
Total of indigenous individuals	11	56

DISCUSSION

Significant disturbances, such as forest trail construction, present opportunities to measure exotic plant colonization. The effects of this on invasive plant establishment have never been quantified in previous studies of alien plant dynamics.

The results showed that wide canopy opening provided sufficient light for the germination, establishment and persistence of *R. alceifolius*, the widespread exotic plant on La Réunion (see Macdonald *et al.*, 1991). In contrast, the lower light intensity under narrow openings enabled germination but not the establishment of *R. alceifolius*, which wilt within the first year. Although several studies have shown that different *Rubus* species germinate and grow when sufficient light is available (Scott & Draper, 1967; Amor, 1974; Salonen, 1994), none has demonstrated the impact of canopy opening by the construction of a new forest trail on the colonization potential of an invasive alien species. As our experiment was set up in the non-fructiferous season, we are confident that *R. alceifolius* seedlings recorded on the recent trails came from the seed-bank already present in the soil before trail construction (see Baret *et al.*, 2004 for an example on this area). Thus, seedlings observed were linked only to the opening of the canopy and the new available light, and not to the transport of propagules along the trail. Our results suggest, moreover, that under these favorable conditions *R. alceifolius* had rapid growth and some individuals bore fruit within only one and a half years of germination within the wide trail. These data are in line with the theories put forward by Rejmánek & Reichard (2001) who considered that a short juvenile period is a key trait of invasive plants, especially in disturbed habitats. This period is crucial and governs the invasion of new areas by these plants. The bright red fruits produced by *R. alceifolius* attract fruit-eating birds, which subsequently disseminate these seeds (see Mandon-Dalger, 2002), facilitating its germination (Mandon-Dalger *et al.*, 2004). These seeds accumulate in the forest soil and since *Rubus* seeds are estimated to remain viable for about 10 years (Marks, 1983), they could germinate once the canopy is opened. The initial plant then spreads by terrestrial layering or sprouting to form monospecific patches (Baret *et al.*, 2003).

Our study also quantified that forest trails favor exotic plant establishment (82.8% along trail edges *vs.* 12.5% 10 m within understorey). Furthermore, we have observed that species richness was higher in the understorey than along trail edges. Except for *Psidium cattleianum* that can grow in the understorey, all other species observed 10 m away from the trail were indigenous. Species present along trail edges were mostly exotic. Among the 6 exotic plant species recorded, 5 are considered very invasive (Macdonald *et al.*, 1991), and one, *Ludwigia octovalvis*, is naturalized.

The obtained results indicate that forest managers should avoid creating wide forest trails. Indeed, small understorey trails that do not create opening in the forest canopy should be considered. This study provides important information about protected area- management practices where additional forest trail construction is vital for public access and tourism development. For example, the forestry service of La Réunion manage 759 km of forest trails within 100 000 ha (ONF, 2002). This value is an underestimate knowing that not all forestry trails are recorded (ONF, pers. com.). Furthermore, poaching for endemic palms and birds in the rainforest create many new trails in protected habitats. We do not recommend widening trails in native habitats as they promote exotic plants invasion. Further research needs to be done to compare the effects of trails in other threatened island ecosystems.

ACKNOWLEDGMENTS

This study was conducted under the auspices of, and financed by, the Région Réunion project, REG/97/0307. The authors thank Mathieu Rouget and Ingrid Nänni for their advice, Jean-Marie Pausé for field assistance and two anonymous referees for insightful and constructive criticisms.

REFERENCES

Amor, R.L. (1974). — Ecology and control of blackberry (*Rubus fruticosus* L. agg.). II Reproduction. *Weed Research*, 14: 231-238.

Amor, R.L. & Stevens, P.L. (1976). — Spread of weeds from a roadside into sclerophyll forests at Dartmouth, Australia. *Weed Research*, 16: 111-118.

BARET, S., NICOLINI, E., LE BOURGEOIS, T. & STRASBERG, D. (2003). — Developmental patterns of the invasive bramble (*Rubus alceifolius* Poiret, Rosaceae) in Réunion Island: an architectural and morphometric analysis. *Ann. Bot. (London)*, 91: 39-48.

BARET, S., MAURICE, S., LE BOURGEOIS, T. & STRASBERG, D. (2004). — Altitudinal variation in fertility and vegetative growth in the invasive plant *Rubus alceifolius* Poiret (Rosaceae) on Réunion Island. *Plant Ecology*, 172: 265-273.

BROCKIE, R.E., LOOPE, L.L., USHER, M.B. & HAMANN, O. (1988). — Biological invasions of island nature reserves. *Biol. Conserv.*, 44: 9-36.

COLES, D.N. & LANDRES, P.B. (1996). — Threats to wilderness ecosystems: impacts and research needs. *Ecol. Appl.*, 6: 168-184.

COTTAN, G. & CURTIS, J.T. (1956). — The use of distance measures in phytosociological sampling. *Ecology*, 37: 451-460.

COURNAC, L., DUBOIS, M.A., CHAVE, J. & RIÉRA B. (2002). — Fast determination of light availability and leaf area index in tropical forests. *J. Trop. Ecol.*, 18: 295-302.

D'ANTONIO, C. & MEYERSON, L.A. (2002). — Exotic plant species as problems and solutions in ecological restoration: a synthesis. *Restor. Ecol.*, 10: 703-713.

GARGOMINY, O. (2003). — *Biodiversité et conservation dans les collectivités françaises d'outre-mer.* Comité français pour l'UICN, Paris, France.

GELBARD, J.L. & BELNAP, J. (2003). — Roads as conduits for exotic plant invasions in a semiarid landscape. *Conserv. Biol.*, 17: 420-432.

GORDON, D.R. (1998). — Effects of invasive non-indigenous plant species on ecosystem processes: lessons from Florida. *Ecol. Appl.*, 8: 975-989.

HARRISON, S., HOHN, C. & RATAY S. (2002). — Distribution of exotic plants along roads in a peninsular nature reserve. *Biological Invasions*, 4: 425-430.

JACOB DE CORDEMOY, E. (1895). — *Flore de l'île de La Réunion*. Klincksieck, Paris.

KING, S.E. & GRACE, J.B. (2000). — The effects of gap size and disturbance type on invasion of wet pine savanna by cogongrass, *Imperata cylindrica* (Poaceae). *Am. J. Bot.*, 87: 1279-1286.

LAVERGNE, C., RAMEAU, J.C. & FIGIER, J. (1999). — The invasive woody weed *Ligustrum robustum* subsp. *walkeri* threatens native forests on La Réunion. *Biological Invasions*, 1: 377-392.

LONSDALE, W.M. & LANE, L.A. (1994). — Tourist vehicles as vectors of weed seeds in Kakadu National Park, northern Australia. *Biol. Conserv.*, 69: 277-283.

LORENCE, D.H. & SUSSMAN, R.W. (1988). — Diversity, density and invasion in a Mauritius wet forest. *Monogr. Syst. Bot. Missouri Bot. Gard.*, 25: 187-204.

LOCKWOOD, J.L., SIMBERLOFF, D., MCKINNEY, M.L. & VON HOLLE, B. (2001). — How many, and which, plants will invade natural areas? *Biological Invasions*, 3: 1-8.

MACDONALD, I.A.W., THÉBAUD, C., STRAHM, W.A. & STRASBERG, D. (1991). — Effects of alien plant invasions on native vegetation remnants on La Réunion (Mascarene Islands, Indian Ocean). *Environ. Conserv.*, 18: 51-61.

MANDON-DALGER, I. (2002). — *Sélection de l'habitat et dynamique d'invasion d'un oiseau introduit, le cas de bulbul orphée à la Réunion*. Thèse de Doctorat, Université de Rennes 1, France.

MANDON-DALGER, I., CLERGEAU, P., TASSIN, J., RIVIÈRE, J.N. & GATTI, S. (2004). — Relationship between alien plants and an alien bird species on Reunion Island. *J. Trop. Ecol.*, 20: 635-642.

MARKS, P.L. (1983). — On the origin of field plants of the northeastern United States. *Am. Nat.*, 122: 210-228.

MUNGROO, Y., MAUREMOOTOO, J.R. & BACHRAZ, V. (1998). — *Proceedings of the Workshop on Restoration of Highly Degraded and Threatened Native Forests in Mauritius. A UNDP/GEF Project*. National parks and Conservation service, Ministry of Agriculture, Food Technology and Natural Resources, Mauritius.

OLSON, D.M. & DINERSTEIN, E. (1998). — The global 200: A Representation Approach to Conserving the Earth's Most Biologically Valuable Ecoregions. *Conserv. Biol.*, 12: 502-515.

O.N.F. (2002). — *Orientations Régionales Forestières. Vol. 1. Etat des lieux et travaux préparatoires*. AMPAAR, La Réunion.

REJMÁNEK, M. & REICHARD, S. (2001). — Predicting invaders. *TREE*, 16: 545-546.

RESTREPO, C. & VITOUSEK, P. (2001). — Landslides, alien species, and diversity of Hawaiian montane mesic ecosystem. *Biotropica*, 33: 409-420.

RICHARDSON, D.M., PYŠEK, P., REJMÁNEK, M., BARBOUR, M.G., PANETTA, F.D. & WEST, C.J. (2000). — Naturalization and invasion of alien plants-concepts and definitions. *Diversity & Distributions*, 6: 93-107.

SALONEN, V. (1994). — Growth and reproduction of *Rubus saxatilis* L. in relation to availability of light. *Acta Œcol.*, 15: 485-493.

SCOTT, D.H. & DRAPER, D.A. (1967). — Light in relation to seed germination of Blueberries, Strawberries, and Rubus. *HortScience*, 2: 107-108.

STRASBERG, D. (1995). — Processus d'invasion par les plantes introduites à La Réunion et dynamique de la végétation sur les coulées volcaniques. *Ecologie*, 26: 169-180.

STRASBERG, D., ROUGET, M., RICHARDSON, D.M., BARET, S., DUPONT, J. & COWLING, R.M. (2005). — An assessment of habitat diversity and transformation on La Réunion Island (Mascarene Islands, Indian Ocean) as a basis for conservation planning. *Biodiv. & Conserv.*

THÉBAUD, C. & STRASBERG, D. (1997). — *Plant dispersal in fragmented landscapes: a field study of woody colonization in rainforest remnants of the Mascarene Archipelago*. Pp. 321-332, *in*: W.F. Laurance & R.O. Bierregaard (eds). *Tropical Forest Remnants: Ecology, Management, and Conservation of fragmented communities*, University of Chicago Press, Illinois, USA.

VITOUSEK, P.M. (1988). — Diversity and biological invasions on oceanic islands. Pp. 181-189, *in*: E.O. Wilson & F.M. Peter (eds). *Biodiversity*, Washington DC, Nat. Academy Press.

VITOUSEK, P.M. (1990). — Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. *Oikos*, 57: 7-13.