BIOLOGICAL INVASIONS

Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species

Edited by
David Pimentel

CRC PRESS

BIOLOGICAL INVASIONS

Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species

BIOLOGICAL INVASIONS

Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species

Edited by
David Pimentel, Ph.D.
Cornell University
Ithaca, New York

CRC PRESS
Boca Raton London New York Washington, D.C.

Senior Editor: John Sulzycki
Project Editor: Gail Renard
Production Manager: Pat Roberson
Marketing Manager: Nadja English
Cover Designer: Dawn Snider

Library of Congress Cataloging-in-Publication Data

Biological invasions : economic and environmental costs of alien plant, animal, and microbe species / edited by David Pimentel.
p. cm.
Includes bibliographical references and index.
ISBN 0-8493-0836-4 (alk. paper)
1. Biological invasions—Economic aspects. 2. Biological invasions—Environmental. aspects. I. Pimentel, David, 1925–

QH353 .B57 20002
577′18—dc21 2002017489
CIP

Direct all inquiries to CRC Press LLC, 2000 N.W. Corporate Blvd., Boca Raton, Florida 33431.

Visit the CRC Press Web site at www.crcpress.com

International Standard Book Number 0-8493-0836-4
Library of Congress Card Number 2002017489
Printed in the United States of America 1 2 3 4 5 6 7 8 9 0
Printed on acid-free paper

About the Editor

David Pimentel, Ph.D., is Professor of Ecology, Evolutionary Biology, and Agricultural Sciences in the Department of Entomology, the Department of Ecology and Evolutionary Biology, and the Field of Natural Resources, College of Agriculture and Life Sciences, Cornell University. Dr. Pimentel received his B.S. in 1948 from the University of Massachusetts and his Ph.D. in 1951 from Cornell. From 1951 to 1954 he was chief of the Tropical Research Laboratory, U.S. Public Health Service, San Juan, Puerto Rico. From 1954 to 1955 he was postdoctoral research fellow at the University of Chicago; 1960 to 1961, OEEC Fellow at Oxford University (England); and 1961, research scholar at Massachusetts Institute of Technology. He was appointed assistant professor of insect ecology at Cornell University in 1955 and associate professor in 1961. In 1963 he was appointed professor and head of the Department of Entomology and Limnology. He served as head until 1969 when he returned to full-time research and teaching as professor of ecology and agricultural sciences.

Nationally, he has served on numerous presidential commissions and National Academy of Sciences' committees and boards, and several committees and boards of the U.S. Department of Health, Education and Welfare, U.S. Department of Energy, U.S. Department of Agriculture, U.S. Congress Office of Technology Assessment, and U.S. State Department. He is currently president of the Rachel Carson Council and elected member of the National Audubon Society and the American Institute of Biological Sciences.

His honors and achievements include: being a fellow at Oxford University (England); delivering keynote addresses at numerous international and national scientific society conferences; being appointed a Phi Beta Kappa visiting scholar; being appointed honorary professor, Institute of Applied Ecology, China; receiving the Distinguished Service Award from the Rural Sociology Society; serving on the board of directors, International Institute of Ecological Economics, Royal Swedish Academy of Science; and serving as editor of *Journal of Environment, Development and Sustainability.*

Dr. Pimentel has authored nearly 600 scientific publications, written two books, and edited 20 others.

Contributors

David Pimentel (Editor)
College of Agriculture and Life Sciences
Cornell University
Ithaca, New York

Australia

Mary Bomford
Bureau of Rural Sciences
Canberra, Australia

Deon Canyon
Tropical Infectious and Parasitic Diseases Unit
School of Public Health and Tropical Medicine
James Cook University
Townsville, Australia

Richard H. Groves
CSIRO Plant Industry and CRC Weed Management Systems
Canberra, Australia

Quentin Hart
Bureau of Rural Sciences
Canberra, Australia

Ian Naumann
Agriculture, Fisheries and Forestry–Australia
Canberra, Australia

Rick Speare
Tropical Infectious and Parasitic Diseases Unit
School of Public Health and Tropical Medicine
James Cook University
Townsville, Australia

Ken Winkel
Director, Australian Venom Research Unit
Department of Pharmacology
University of Melbourne
Parkville, Victoria, Australia

Brazil

Murillo Lobo Junior
Embrapa Hortalicas
Brasilia, Brazil

British Isles

Stephen Harris
School of Biological Sciences
University of Bristol
England

Piran C. L. White
Environment Department
University of York
England

Mark Williamson
Department of Biology
University of York
England

India

Jaspal Kaur
Department of Plant Pathology
Punjab Agricultural University
Ludhiana, India

Rama S. Singh
Department of Plant Pathology
Punjab Agricultural University
Ludhiana, India

New Zealand

Nigel D. Barlow
Biocontrol and Biosecurity Group
AgResearch
Canterbury Agriculture and Science Centre
Lincoln, New Zealand

M. N. Clout
Centre for Invasive Species Research
School of Biological Sciences
Tamaki Campus
University of Auckland
Auckland, New Zealand

Angus Cook
Ecology and Health Research Centre
Department of Public Health
Wellington School of Medicine
Wellington, New Zealand

S. L. Goldson
Biocontrol and Biosecurity Group
AgResearch
Canterbury Agriculture and Science Centre
Lincoln, New Zealand

Susan Timmins
Department of Conservation
Wellington, New Zealand

Philip Weinstein
Ecology and Health Research Centre
Department of Public Health
Wellington School of Medicine
Wellington, New Zealand

Peter A. Williams
Landcare Research
Nelson, New Zealand

Alistair Woodward
Ecology and Health Research Centre
Department of Public Health
Wellington School of Medicine
Wellington, New Zealand

South Africa

M. H. Allsopp
ARC–Plant Protection Research Institute
Stellenbosch, South Africa

J. F. Colville
University of Cape Town
Cape Town, South Africa

C. L. Griffiths
University of Cape Town
Cape Town, South Africa

D. Magadlela
Working for Water Program
Cape Town, South Africa

D. C. Le Maitre
CSIR Division of Water, Environment and Foresty Technology
Stellenbosch, South Africa

C. Marais
Working for Water Program
Cape Town, South Africa

M. D. Picker
University of Cape Town
Cape Town, South Africa

D. M. Richardson
Institute for Plant Conservation
Department of Botany
University of Cape Town
Cape Town, South Africa

B. W. van Wilgen
CSIR Division of Water, Environment and Forestry Technology
Stellenbosch, South Africa

United States

T. Aquino
Cornell University
Ithaca, New York

J. Janecka
Cornell University
Ithaca, New York

Yoonji Kim
College of Agriculture and Life Sciences
Cornell University
Ithaca, New York

Lori Lach
Cornell University
Ecology and Evolutionary Biology
Ithaca, New York

Sarah McNair
Utica, New York

Doug Morrison
Cornell University
Environmental Toxicology
Ithaca, New York

C. O'Connell
Holtsville, New York

E. Wong
Cornell University
Ithaca, New York

L. Russel
Federal Way, WA

C. Simmonds
Wyndmoor, PA

T. Tsomondo
Cornell University
Ithaca, New York

J. Wightman
Ithaca, New York

J. Zern
Cornell University
Ithaca, New York

Rodolfo Zuniga
Cornell University
Natural Resources
Ithaca, New York

Contents

section one

Introduction

chapter one

Introduction: non-native species in the world

David Pimentel

Contents

Some 10 million species of plants, animals, and microbes are thought to inhabit the earth, but so far only about 1.5 million of these have been identified. A mere 15 of the approximately 250,000 known plant species provide the world's human population with about 90 percent of its food.[1] These crops are wheat, rice, corn, rye, barley, soybeans, common bean, white potato, sweet potato, cassava, bananas, coconuts, peanuts, sorghum, and millet. Although these crops are now grown in nearly every nation, only one or two of these species originated in any specific country.

Among animals, eight species currently provide the bulk of the meat, milk, and eggs consumed by humans. These leading livestock species are cattle, buffalo, sheep, goats, horses, camels, chickens, and ducks. Farms in the United States feed approximately 100 million cattle, 7 million sheep, and 9 billion chickens each year.[2]

Although much is known about the world's major food sources, relatively little is known about the vast number of plant, animal, and microbe species that have migrated throughout the world and invaded new ecosystems. Every nation now has thousands of non-native, introduced species inhabiting their ecosystems. Many crop and livestock species were intentionally introduced into these ecosystems because native plants and livestock could not provide sufficient food for a country's needs; other species were either intentionally or accidentally introduced nation's ecosystems, along with human invasions.

0-8493-0836-4/02/$0.00+$1.50

The invasion of non-native species into new ecosystems is accelerating as the world's human population multiplies, and as goods are transported ever more rapidly on an increasingly global scale. Several of these non-native species of plants, animals, and microbes were originally introduced for use in agriculture but have since become major pests. In the United States, for example, these include Johnson grass, which was introduced for livestock grazing, and cats, which were introduced for mouse control.

The impact of invasive species is second only to that of human population growth and associated activities as a cause of the loss of biodiversity throughout the world. In the United States, invasions of non-native plants, animals, or microbes are thought to be responsible for 42 percent of the decline of native species now listed as endangered or threatened.[3] The loss of biodiversity caused by invasive species is the result of competition from invasives and the resulting displacement of native species, as well as by predation and hybridization.

Several decades ago the British ecologist Charles Elton[4] investigated invading species worldwide and the widespread environmental damages they caused. He became aware of the need to assemble information about such invasive organisms, including their ecological effects and the difficulty in controlling those that become pests.

The contributors to this book have built on Elton's early studies and share in these pages their investigations into the environmental and economic impacts of invading species. They compare the number of native and non-native species for several regions of the world. Where possible, information is provided on how non-native species invaded an ecosystem, as well as the environmental and economic consequences.

Contributing scientists from Australia, Brazil, the British Isles, India, New Zealand, South Africa, and the United States share their expertise in this volume. Several factors were involved in selecting the nations discussed here, as will be explained below.

1.1 Australia

Australia's relative geographical isolation has not protected the continent from the influx of invasive species. Groves, in his investigation of invasive plants in Australia, reports that the number of introduced plant species is thought to be roughly equal to the number of native species — about 25,000 each.

Groves puts the number of alien plant species that have become established in Australian natural habitats at 2681. A few of the major weed pests include wild oats, skeleton weed, Mexican feather grass, Spanish thistle, serrated tussock grass, and Paterson's curse. The most costly damage inflected by invasive weeds is to crop systems, which suffer an estimated damage of $1.271 billion (Australian) each year. Damage to pasture land accounts for another $494 million per year, while the horticultural industry bears a cost of $213 million each year.

Bomford and Hart expand the knowledge of invasive vertebrate species in Australia and indicate that more than 80 species of non-indigenous vertebrates have become established in Australia. Of these species, more than 30 have become serious pests, among them the European rabbit, feral pigs, feral cats, the dingo dog, feral goats, the European starling, and the cane toad. The direct economic losses caused to agriculture by these introduced vertebrate pests are an estimated $420 million per year. Control costs borne by the government and landholders represent an additional $60 million per year, while another $20 million or so is spent on related research.

Although no estimate is reported here as to the overall number of invertebrates that have been introduced into Australia, several of the major non-native invertebrate pests are discussed by Canyon, Speare, Naumann, and Winkel. These species include the mosquitoes *Aedes aegypti* and *Culex gelidus*, both of which transmit serious diseases; honeybees and wasps, which cause human deaths; red fire ants, which cause human, livestock, and wildlife problems; the cattle tick; screw-worm fly complex; the red-legged earth mite, which damages crops; and the European wood wasp, which attacks forests. The invasive species investigated by the contributors indicate that invertebrates in Australia are responsible for as much as $5 billion to $8 billion in annual damage and control costs. One table listing several of the major pests estimated damages totaling $4.7 billion per year from this group of pests alone.

1.2 Brazil

In his study of plant pathogens introduced into Brazil, Lobo indicates that more than 500 species of exotic pathogenic fungi, 100 virus species, 25 nematode species, and 1 protozoan species are established and attacking crops in Brazil. On average, the alien pathogens are causing an estimated 15% loss in potential production of crops. Lobo estimates that the total losses caused by non-indigenous plant pathogens in Brazil are estimated to be $6.9 billion each year.

1.3 The British Isles

In his study of invasive plants in the British Isles, Williamson found that the number of native plant species is about 1500, while the number of known alien plant species is 1642. The number of alien plants that have become well established in natural ecosystems is estimated to range from 210 to 558 species. Most of the damage and control costs, which range from £200 million to £300 million per year, are associated with the impact of alien species on crops.

A study by White and Harris of vertebrate species that have invaded the British Isles reports that of the 63 mammal species present, 22 species are alien; of the 219 bird species, 12 species are alien; of the 14 amphibian species, 8 are alien; of the 9 reptile species, 3 species are alien; and of the 35 fish species, 13 are alien.

The environmental and economic damage caused by the alien vertebrate species is diverse and costly. The invading European rabbit continues to be a serious pest in the British Isles, even though the rabbit population peaked in the 1950s. When the myxomatosis virus was introduced in 1954, rabbit numbers fell by 99 percent within a few years, but today the annual losses to cereal crops due to rabbit predation in the British Isles are estimated to be £40 million.

In a brief survey of alien arthropod and plant pathogen damage to crops in the British Isles, Pimentel reports that of the 1500 species of insect and mite pests on crops, some 450 are invasive species. The introduced insect and mite species cause an estimated $960 million of damage each year. Interestingly, an estimated 74% of the plant pathogen species in the British Isles were introduced along with the introduced crops. These alien species of microbes are estimated to cause about $2 billion in crop losses each year.

1.4 India

Authors Singh and Kaur emphasize that because about 70% of the population in India is involved in agriculture, plant pathogens are a major concern there. For example, the epiphytotic blight caused by the invasive *Helminthosporium* fungus resulted in major rice losses, setting off a famine in which some 2 million people died. So far, at least 17 major alien plant pathogens have become established in India, and some are causing major crop losses, ranging from 70% to 100%. However, average crop losses from alien plants are estimated to be about 20%, which is still a significant loss in a country in which many people are malnourished and so many citizens depend on crops for their livelihood and sustenance.

1.5 New Zealand

New Zealand, a historically isolated ancient landmass, has suffered severe damage from invasive species. According to Williams and Timmins, the number of native plant species in New Zealand is about 2000 species, while an estimated 1800 species of alien plant species have invaded the island nation. New Zealand's primary industries of agriculture, horticulture, and forestry are based on a total of 140 species, most of which were introduced. Approximately 200 species of invasive plants have become serious weeds that now cost about \$60 million (New Zealand) per year to control. Another \$40 million in crop damages are caused by these invasive weeds.

Clout reports that the natural terrestrial vertebrate fauna of New Zealand was particularly unusual and was dominated by only a few species of birds, reptiles, and bats. Over the years — and especially following the influx of European settlers — more than 90 species of vertebrates, including 32 mammals, 26 birds, and 19 fish — were introduced. Today, approximately 25% of New Zealand's crop losses are attributed to damage by vertebrate species, including the Australian brushtail possum and the European rabbit.

Barlow and Goldson report that an estimated 2200 species of alien invertebrates have invaded New Zealand. This is an extremely large number for such a geographically isolated region. The chief non-native agricultural invertebrate pests include Lucerne insect pests, the common wasp, the Argentine ant, the Mediterranean fruit fly, and the painted apple moth. Each year the invading invertebrate species inflict about \$195 million per year in damage to crops, while control costs are an additional \$242 million.

The environmental and control costs of all pests and weeds in New Zealand come to an estimated \$840 million per year.

In Chapter 13, Cook, Weinstein, and Woodward provide valuable historical perspective of the invasion of New Zealand by alien insects and related arthropods. The impact of humans and all the invading animals, plants, and microbes they intentionally and unintentionally brought with them has resulted in the extinction or endangerment of more than 1000 native plant and animal species in New Zealand. The introduction of the Southern salt-marsh mosquito (*Ochlerotatus [Aedes] camptorhynchus*) appears to be a major health threat to the people of New Zealand. This mosquito is a significant vector of the Ross River virus, of which some 5000 cases per year are now being reported. The annual cost of mosquito control is already about \$14 million. Recent estimates of the environmental damage and control costs for all imported pests in New Zealand are in the billions of dollars per year.

1.6 South Africa

South Africa suffers from a large number of non-indigenous species. In a detailed analysis of plants introduced into South Africa, van Wilgen, Richardson, LeMaitre, Marais, and Magadlela report that more than 8750 species of plants have invaded the vast South African ecosystem. Of this number, about 161 species now rank as serious pest weeds. These invasive weeds are causing loss of natural biodiversity, water shortages, loss of crop and forest production, and increased soil erosion. The authors estimate the annual environmental losses in the mountain fynbos area to be \$11.75 billion; water losses are about \$3.2 billion; reduced stream flow costs are an estimated \$1.4 billion; and water fern impacts in aquatic ecosystems cost about \$58 million. Biological control is proving to be one effective way to control the invading weeds.

Lach, Picker, Colville, Allsopp, and Griffiths report on invertebrate invasions by non-native species into South Africa. Relatively little is known about the total number of alien invertebrate species in the country. A total of 56 species of insects have been introduced as biological control agents for invasive weeds in South Africa, and more than half of these biocontrol species are succeeding at providing some control of the weed pests.

Of the 40 major crop pests in South Africa, it is estimated that 42% are non-native species. Two ant species have invaded South Africa, and one, the Argentine ant, is causing major ecological problems. This ant is displacing native invertebrate species and interfering with the pollination of both native and crop plants, and is a major pest in agriculture both directly and indirectly. In addition, both freshwater and marine ecosystems are suffering environmental impacts from invading invertebrate species. Alien invertebrate pests cause an estimated $1 billion in crop damage and control costs each year in South Africa.

1.7 The United States

Pimentel et al. report that more than 50,000 species of plants, animals, and microbes have been introduced either accidentally or intentionally into the United States in the past hundred years.

Among these are 128 crop species that were intentionally introduced into the United States but have since become annoying weeds or serious pests of agriculture and horticulture. One such pest is Johnson grass, which was introduced as a forage grass but now is a major weed pest throughout the southern United States. The melaleuca tree, intentionally introduced as an ornamental tree, is now spreading rapidly throughout Florida and other Southern states, where it displaces native trees and other vegetation and is removing vital moisture from the Everglades and other ecosystems.

The spread of invasive weeds causes an estimated $34 billion in damage and control costs in the United States each year. When invasive plants displace native vegetation, the native animals and microbes associated with the native vegetation native species are greatly reduced in number. Most of the damage from invading plants in the United States occurs to natural ecosystems, primarily in the South and the West.

Vertebrate species introduced to the United States cause an estimated $39 billion in damage and control costs each year, with rats and cats being responsible for the majority of the problems and losses. Meanwhile, invading invertebrate species, such as pest insects, destroy some $20 billion worth of U.S. crops and forests each year.

Invading plant pathogens attack crops and forests as well, causing an estimated $25 billion worth of damage and control costs annually in the United States. An additional $16 billion is spent in the United States to deal with introduced microbes, such as the HIV (AIDS) and influenza viruses.

1.8 World overview

In a preliminary investigation, Pimentel et al. summarize the economic and environmental damage caused by alien plant, animal, and microbe species in the United States, the British Isles, Australia, South Africa, India, and Brazil. They report that more than 120,000 non-native species of plants, animals, and microbes not only have invaded these nations, but have become well established in the new ecosystems. The invasion of these non-native organisms causes more than $314 billion per year in damage and control costs in these key regions.

Kim reports on the number of humans infected by invading organisms in Australia, Brazil, the British Isles, India, New Zealand, South Africa, and the United States. Surprisingly, little is know about the origins and the spread of several pathogenic diseases that affect human health.

One of the most recent invading infectious organisms, and now one of the best known, is the HIV virus, which causes AIDS. In the seven nations studied, nearly 9 million people are currently infected with HIV/AIDS, with about 7.6 million infected initially in South Africa and India. The World Health Organization (WHO) estimates that $7 billion per year is needed to fight HIV/AIDS.

Worldwide, about 2 billion people are currently infected with tuberculosis (TB), and 2.4 billion are infected with malaria. These two diseases are causing enormous economic hardships and a great many deaths each year. The WHO reports that several billion dollars are needed to control these two major diseases. In India alone, TB costs $3 billion each year in terms of deaths, lost work, and medical treatment.

The information provided in this book reconfirms the diverse and unpredictable roles that non-native species assume as they invade new ecosystems. They often attack vital crops and forests, and they may cause major damage to ecosystems that results in loss of biodiversity, soil erosion, and water loss. In addition, major human and livestock diseases have invaded many countries, resulting in significant health and economic problems.

Alien species invasions will be an ongoing problem in the future as the human population multiplies and becomes increasingly mobile. The increasing movement of goods associated with globalization will also tend to accelerate the spread of alien species as never before.

References

1. Pimentel, D. and Pimentel, M. *Food, Energy, and Society*, rev. ed., University Press of Colorado, Niwot, CO, 1996, p. 363.
2. USDA, *Agricultural Statistics*, U.S. Dept. of Agriculture, Washington, DC, 2000.
3. Nature Conservancy. *America's Least Wanted: Alien Species Invasions of U.S. Ecosystems*, The Nature Conservancy, Arlington, VA, 1996.
4. Elton, C.S. *The Ecology of Invasions by Animals and Plants*, Methuen, London, 1958.

section two

Australia

chapter two

The impacts of alien plants in Australia

Richard H. Groves

Contents

2.1 Introduction

An unknown number of alien plant species have been introduced to Australia, both accidentally and deliberately. A recent publication on Australian plants of horticultural significance[1] lists some 30,000 plant names as being available from 450 nurseries in all states and territories of Australia. This listing includes not just plant species that have been deliberately introduced, but also native plants that are used in horticulture, together with some synonyms and cultivar names. Until more accurate estimates are available, we may assume that the number of alien plant species introduced to Australia at least equals the number of native higher plant species, which are currently estimated to be about 25,000. Not all of these alien species have become naturalized, however. An equally recent listing of the naturalized alien flora of Australia[2] gives a total of 2681 plant species that are known to be naturalized and to have voucher specimens lodged in Australian herbaria. In other words, about 10 to 15% of the total Australian flora is alien and naturalized. Some of these alien and naturalized plant species affect, or are perceived to affect, human activities in some way and may be regarded as weeds. This chapter considers some of these 2681 alien naturalized plant species and their impacts on Australian ecosystems, but

0-8493-0836-4/02/$0.00+$1.50

the coverage is not limited to the smaller proportion of naturalized aliens that are generally regarded as weeds.

Some cosmopolitan species may be regarded either as alien or native.[3,4] For Australia as a whole, it was estimated[2] that this uncertain status applies to only 34 plant species, i.e., 1.1%, which is a small proportion of the total alien flora. These relatively few plant species typically occupy either wetlands or beach strand-lines, where bird- or water-dispersed species predominate. While these cosmopolitan species may be numerically significant among most insular floras, they comprise only a trivial proportion of the flora of the large land mass of continental Australia.

The proportion of the total Australian flora that is alien varies from region to region and from ecosystem to ecosystem. For instance, offshore islands have a high proportion of alien species (60% for Norfolk Island[5] and 48% for Lord Howe Island[6]), whereas the flora of some arid (Uluru N.P.[7]) and alpine (Kosciuszko N.P.[8]) areas are only about 5 to 7% alien.[9] While the percentage of alien species may not have changed greatly over time, the number of naturalized alien species has increased inexorably since the first state floras were compiled. Specht[10] showed a fairly constant rate of increase at about five species per year per state for Queensland, New South Wales, Victoria, and South Australia for the period 1870–1980. More recently, Groves et al.[11] provided evidence that, nationally, this rate may have increased over the recent 25-year period from 1971–95. Certainly, they could find no evidence that the proportion of naturalized alien plants in the Australian flora had decreased, despite almost 100 years of quarantine regulating the entry of alien plant species.

In this chapter, I will discuss the impacts of this increasing number of alien plant species on the Australian community from the perspective of the economics of crop and pasture enterprises, on native plant diversity, and on human and animal health. While in some cases the effects of alien plants on some agricultural systems have been quantified and the cost-benefit ratios of controlling them calculated, few such quantitative estimates are available regarding impacts on natural ecosystems in terms of native plant and animal diversity or human and animal health aspects. Some recommendations are made for further research on the impacts of alien species on Australian ecosystems and on the native plant diversity present in those ecosystems.

2.2 Impacts on agricultural ecosystems

Alien plants influence crop and pasture ecosystems in many ways. The crop systems themselves consist largely of alien economic plants, as few native Australian plants have been domesticated. The plant species that form the basis of pasture ecosystems in southern Australia are also alien, having been introduced mainly from Mediterranean Europe. On the other hand, most of the plants that form the basis of northern (summer-wet) and central (semi-arid, rangeland) grazing systems in Australia are native to those regions. The negative impacts of alien species on agricultural ecosystems in southern Australia will be stressed, because that is where the available data are concentrated. It should also be recognized, however, that some alien species that are useful in pasture ecosystems may impact negatively on crop systems, such as *Trifolium subterraneum*.

2.2.1 Economic aspects

The negative impacts of alien plants on crop and pasture systems throughout Australia in general have been estimated. The incidence of alien plants leads to the need to cultivate land for crops or to re-sow pastures, or to spray with herbicides, or both. The presence of these aliens is associated directly with reductions in crop or pasture yield and with product contamination. Some alien plant species may poison animals or lead to poor animal performance.

Each aspect incurs a financial cost. For Australian crop systems as a whole, Combellack[12] estimated the financial costs of each aspect. For the financial year 1981–82, cultivation to control alien plants cost $592 million (Australian); purchase of herbicides cost $137 million, plus $34 million to apply them. In the same year, losses in crop yields were estimated to be $422 million and product contamination to cost $86 million. These estimates gave a total annual cost of alien plants in crop systems of $1.271 billion for that year. Financial estimates of the negative impacts of alien plants in pasture systems for the same year were $494 million, in horticulture $213 million, and in "non-crop" areas $119 million. For all agricultural systems, Combellack's estimates (based on 1981–82 statistics) total $2.1 billion, which translated into $3.3 billion in 1995-dollar terms (see Jones, et al.[13] for questions on the validity of such an extrapolation). Results of a more recent study showed that product losses and expenditure on control of aliens at current infestation levels in crop systems amounted to $1.133 billion for the 1998–99 financial year.[13] Whatever the accounting system used, alien plants annually cost Australian agricultural producers a substantial amount of money, and the costs are also borne by consumers.

The direct financial cost of some individual weeds in crop and pasture systems has been estimated for a few species. The alien species complex called wild oats (*Avena* spp.) in grain crops was estimated to cost $42 million for the financial year 1987–88.[14] This estimate nevertheless was conservative, because it did not include the cost of grain contamination, increased opportunity provided by the *Avena* to host pathogens, or increased resistance to control methods. The financial impact of skeleton weed (*Chondrilla juncea*) on wheat crops was estimated for the 1972–73 financial year at $20 million,[15,16] of which $18.5 million was attributable to lost productivity and $1.5 million to spraying costs. These two examples suffice to show that the costs associated with the presence of some alien species among crop systems are considerable. With a pattern of increasing resistance to herbicides shown by several of these alien species, especially the annual grasses group, the costs of aliens in crop systems will increase.

Some other negative impacts of alien plants on agricultural systems may add to the annual costs. For instance, recent attempts to prevent the incursion of two alien plant species (*Nassella tenuissima*, or Mexican feather grass, and *Onopordum nervosum*, a Spanish thistle) that have serious potential to modify pasture systems were estimated to generate benefits to producers of $83 million in 2000–01.[17] This estimate was based on a reduction in the probability of these weeds becoming naturalized, and thereby a reduction of potential costs they would impose should they ever become established. So far, both species are available only in the nursery industry (as plants or seeds, respectively, for landscaping) and are not yet known to have escaped cultivation, let alone naturalization.

Alien plants also impact directly on pasture ecosystems. Serrated tussock (*Nassella trichotoma*) is a perennial grass of South American origin that reduces the livestock carrying capacity of southern Australian pastures. Its presence incurs an annual cost of $40 million in New South Wales[18] and, for 1997, about $5.1 million in Victoria.[19] If the weed is not contained in Victoria, that cost estimate could increase to $15 million per year in 10 years' time.[19] The same species is now spreading in Tasmania as well, representing an additional but hitherto unquantified cost to southern Australian pasture production.

Financial estimates of the costs of other individual alien species in southern Australia pastures are also available for two cases in which biological control of the species was proposed but faced opposition from some sectors of the Australian community. In each case, there were demonstrable conflicts of interest arising from the fact that some alien species have both negative and positive effects. For instance, costs of the alien species were estimated as part of the overall decision to allow the release of biological control agents. The first case concerns Paterson's curse (*Echium plantagineum*), which produces alkaloids that affect liver function in grazing animals (especially sheep) but which also produces honey with a pale color preferred by exporters to the Japanese market. Further, while Paterson's curse is a serious pasture weed in most parts of southern Australia, it may be considered as useful fodder for animals in some semi-arid rangelands, especially in northern South Australia, where its common name is Salvation Jane. An independent inquiry into the merits of both the negative and positive aspects of biological control of this weed recommended release of insects to control growth of Paterson's curse on the basis of an economic analysis of the costs ($30 million annually) and benefits ($2 million annually) to Australia.[20]

My second case concerns blackberry (*Rubus fruticosus* agg.). Data were gathered in the 1980s on aspects of biologically controlling blackberry regarding the additional costs to Tasmanian berry growers and honey producers of controlling the rust proposed for release, balanced against the benefits of increasing pasture production by controlling blackberry. The data collected in the early 1980s indicate a total annual cost to Australia of $41.5 million.[21] More recently, both the negative and positive impacts of blackberry infestations have been itemized by James and Lockwood.[22] These authors stress the need to collect much more information on current distribution and impact valuation before an up-to-date economic analysis can be made for the 8.8 million hectares that blackberry now occupies in southern Australia.

These examples collectively show that alien plants can impact directly and significantly on southern Australian crop and pasture systems. In economic terms, the negative impacts of alien species far outweigh any positive ones. The continuing research that leads to improved levels of control of such aliens that negatively impact agriculture is usually highly cost-effective.[15,17] Less attention has been paid to the effects of such alien species on the sustainability of southern Australian agriculture (and specifically its profitability), although with the steadily increasing salinity of groundwater and the increasing resistance of crop and pasture weeds to herbicides, these aspects are urgently in need of increased research attention.

2.3 Impacts on natural ecosystems

2.3.1 Biodiversity aspects

The impacts of alien plants on natural ecosystems are complex and vary with human attitudes and knowledge. Impact assessment in these systems can be highly subjective. For instance, a few people express zero tolerance for alien plants in natural ecosystems. To these people, any alien species in a nature reserve lessens the quality of the natural environment. Other individuals may tolerate some alien plants, such as those with brightly colored flowers in the ground layer, but will be intolerant of spiny shrubs or rampant vines that may prevent access to waterways or viewpoints. At the other extreme are those individuals who do not even recognize some species as being alien to Australia, such as willows (*Salix* spp.) or poplars (*Populus* spp.), in part because they appear so frequently in early paintings of the Australian landscape; some Australians in fact believe such aliens to be native species. The impacts that alien plants in natural ecosystems may have on people vary far more than do alien plants in agricultural systems, where alien species are usually identified more accurately and their costs estimated more realistically.

A consideration of alien plants in natural ecosystems is also made more complex by the fact that the same plant species may affect both agricultural and natural ecosystems. For instance, blackberry is a major weed of pastures, but it is an equally major weed in natural ecosystems, especially along waterways in southern Australia. Furthermore, blackberry is strongly weedy in the establishment phase of forest plantations. When the weediness of St. John's wort (*Hypericum perforatum*) was first recognised, it was as a weed of dairy pastures. Land use changed, as a result of this weed status, from pasture to forest plantations of *Pinus radiata* in some regions. Currently, the same species occurs mainly in natural ecosystems and roadsides along which it spreads, although it continues to be weedy in pine plantations.

A further example is provided by horehound (*Marrubium vulgare*). This native of the Mediterranean region was introduced to Australia as a source of herbal compounds. It spread to become a weed of sheep-grazed pastures in relatively high-rainfall regions, where it is still a problem plant because of its unpalatability. More recently, it has increased in dominance in some semi-arid areas, where its fruits are spread not by sheep, but by native animals, such as kangaroos. In northwestern Victoria's Wyperfeld National Park it is common to see horehound as a major weed in areas where kangaroos congregate and rest overnight.

From these few examples, it is clear that the distinction between alien plants in agricultural and natural ecosystems is far from rigid, and many widespread alien plants may affect both systems. What's more, their relative impacts on each system may change with time as the same species come to have less effect on agricultural systems and more on natural ones.

As with agricultural systems, alien plants impacting natural ecosystems do so either negatively or positively, and some have no apparent effect. Adair and Groves[23] proposed four hypothetical models for assessing the relationships possible between alien infestations and the biodiversity of natural ecosystems (Figure 2.1). Such models were able to relate levels of biodiversity (e.g., native species richness) to some measure of alien plant infestation. Such models require further development and testing, however, before they become generally acceptable, especially to managers of land affected by alien plants.

Consider the following two examples of different impacts of alien plants on some Australian natural ecosystems. The first concerns *Mimosa pigra*, a native of Central America, which has been introduced to the tropical wetlands of the Darwin region of northern Australia. Negative aspects of this leguminous shrub on the ecosystem include the formation of monospecific thickets of shrubs that replace the native sedgeland, on which the endangered magpie goose (*Anseranas semipalmata*) has historically depended for nesting sites and food. Overall, bird abundance was reduced as a result of mimosa infestation, as was lizard abundance.[24] On mimosa-infested sites, there was also less herbaceous vegetation and fewer native tree seedlings than in uninvaded natural vegetation. All these indices of biodiversity were affected negatively by the presence of high densities of *M. pigra*. On the other hand, frog numbers seemed to be unaffected by mimosa density — an example of a neutral impact. In the same ecosystem, presence of *M. pigra* was associated with increased numbers of the rare marsupial mouse *Sminthopsis virginiae*, presumably because of the increased high-quality food supply provided by *M. pigra* seeds and the increased shelter from predators provided by the dense thickets of the alien shrub. The latter is clearly a positive impact if one measures only *Sminthopsis* numbers as an index of biodiversity. Depending on which measure of biodiversity is chosen, impacts may be negative, neutral, or even positive, and three of the four models proposed[23] may apply to this one alien when measured by different indices of biodiversity.

Much the same mix of impacts seems to apply to the second example, which concerns the invasion of an arid river system in central Australia by the tree species tamarisk (*Tamarix aphylla*), a Eurasian species.[25] The banks of the Finke River in central Australia were originally dominated naturally by river red gum (*Eucalyptus camaldulensis*), but after serious flooding of the river system in 1974, tamarisk seeds were washed downstream from homesteads where the trees had been planted for amenity and shade. Within 15

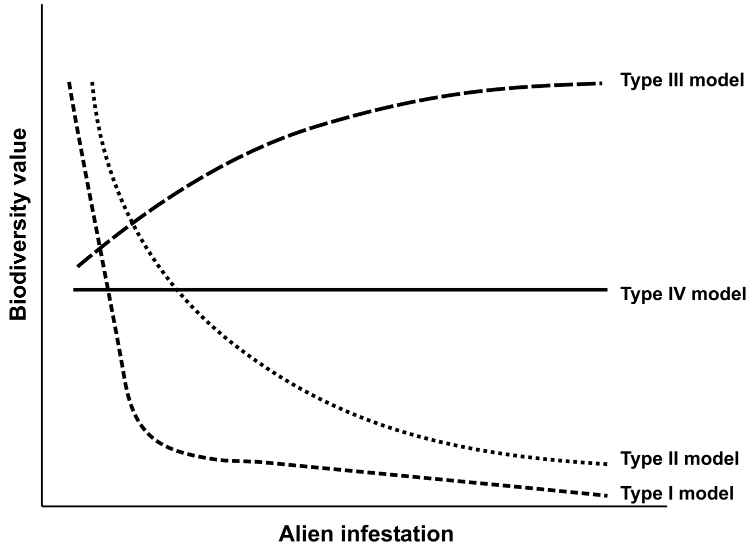

Figure 2.1 Four hypothetical models demonstrating some of the relationships possible between biodiversity value (e.g., number of native plant species) and alien infestation (e.g., weed density). (Redrawn from Adair, R.J. and Groves, R.H., Impact of environmental weeds on biodiversity: a review and development of a methodology, National Weeds Program, Environment Australia, Occasional Publication, Canberra, 1998.)

years of the flood, the eucalypt woodland had changed to one dominated by tamarisk, and various indices of biodiversity had changed markedly. Regeneration of the previously dominant native tree was reduced, the floristic composition of the ground vegetation was changed, and the numbers of reptiles and most birds were reduced — all negative impacts. But, while most bird species declined, some aerial insectivorous species increased — a positive impact — and there seemed to be no effect on the number of granivorous bird species, a neutral effect. The increase in some aerial insectivorous birds may, in turn, lead to positive or negative effects on other aspects of the ecosystem.[25]

For both examples, the impacts are overwhelmingly negative in human terms. Mimosa infestations and the potential spread of this alien into nearby World Heritage–listed Kakadu National Park threaten traditional food-gathering patterns of the resident Aborigines. In addition, the tourist industry is potentially threatened by the associated loss in regional ecosystem diversity. The value of production from pastoral areas in the region adjoining Kakadu is compromised, and the rounding up of cattle on these properties is made more difficult. The negative effects of mimosa on human values justify the large amount of research funds already spent on mimosa control, whereas the impacts of mimosa on biodiversity are mixed. So, too, with the example of tamarisk in relation to its ability to increase the salinity of the invaded region, although less has been spent on its control in the Finke River system. A replay of the scenario for the Finke River may be happening currently in the lower Gascoyne River, near Carnarvon, in Western Australia. This situation may eventually bring increased attention to the control of tamarisk in other regions of semi-arid and arid Australia. This second example is matched by an analogous situation that has developed in the southwestern United States and Mexico, where the closely related *T. ramosissima* is having similarly strong negative effects on the salinity of river systems and on native fish populations in these semi-arid regions.[26]

The next two examples of the impacts of an alien species on natural ecosystems concern individual species, rather than the ecosystems of which the species form a part. The alien climbing species bridal creeper (*Asparagus asparagoides*), native to South Africa, has been shown to directly affect the populations of two rare or threatened native plant species.

The first native species is the sandhill greenhood orchid *Pterostylis arenicola*, indigenous to several sites in South Australia. The orchid is terrestrial; it emerges from root tubers in late autumn each year and forms a small, flat rosette of leaves over winter, then flowers in spring before senescing in late spring.[27] The phenology of the native orchid is matched almost exactly by the phenology of the alien bridal creeper, which sprouts annually from a mat of perennial tubers in autumn, overtops native vegetation, flowers in spring, and forms berries in late spring that are dispersed by both native and alien birds. Its shoots senesce in late spring in summer-dry areas of southern Australia. Thus, the cover of the alien is at its peak when the native orchid rosettes are present, which means that the latter are shaded and rendered less competitive. Sorensen and Jusaitis[27] showed that with bridal creeper absent, the number of orchid rosettes present was about 40 per m^2, whereas with the alien present the number of rosettes was less than 10 per m^2. In this instance, there seem to be no positive or neutral impacts on the ecosystem. This example represents one of the few in Australia in which the effect of an alien on the population of an endangered native plant species has been quantified.

The low shrub *Pimelea spicata* is a minor but once-common component of the shrub layer in Cumberland Plain Woodland to the west of Sydney. At one site where its numbers are greatest (several thousands), it co-occurs with bridal creeper. Again, the phenology of the native matches that of the alien. *Pimelea spicata* has a thick perennial taproot, from which new shoots emerge after drought or fire or other natural disturbances. Shoots elongate and then flower anytime between spring and the following autumn, depending on summer rainfall patterns (the Sydney Basin has year-round rain compared to regions farther south or inland). Bridal creeper's effect on this uncommon native plant is both to smother its shoots through winter and spring, and to compete with it for water and nutrients when its shoot canopy has senesced.[28] The negative impact of bridal creeper on this native plant is expressed both above and below ground, based on results from root and shoot competition studies of the two species.[28] Once again, there seem to be no positive effects on the part of the alien from this species–species interaction in this woodland ecosystem.

These examples of the impacts of bridal creeper illustrate perhaps the most appropriate way to explore the direct interactions between alien and native species, both in controlled conditions in a greenhouse and experimentally in the field. Interactions between alien species and natural ecosystems are more complex and often indirect, and different types of impacts (negative, positive, and neutral) and combinations of those impact types are possible. A further complexity is that alien plants may provide food and refuge for aliens from other taxonomic groupings, whether they be vermin (such as foxes or pigs), pests (for example, insects), or pathogens (crop diseases and the like). The few generalizations that can be drawn from this limited number of examples need further testing as more examples are documented. In terms of the bridal creeper examples, the relationships between alien density and biodiversity value are probably best represented by the Type I or II models.[23] The previous examples of the impacts of alien plants on ecosystems are more complex and involve mixes of Types II, III, and IV response models (Figure 2.1).

2.3.2 Economic aspects

Data on the economic impacts of alien species on natural ecosystems are few and indirect. Panetta and James[29] presented a strategy for collecting and analyzing such data to overcome this deficiency. However, several attempts are now under way to obtain such data by assessing the cost-effectiveness of control programs for alien species. For instance, the financial costs of controlling broom (*Cytisus scoparius*) in Barrington Tops National Park in central coastal New South Wales is being studied,[30] but the results are not yet available. A cost-benefit analysis for the control of bitou bush (*Chrysanthemoides monilifera* spp. *rotundata*), an alien from South Africa now dominating coastal vegetation in eastern Australia, involves both the plant's effect on biodiversity and its threat to public access to beaches. A control program for bitou bush has been implemented that involves strategic use of herbicides, hand-pulling of plants by volunteers, release of a number of highly specific insects from South Africa for long-term biological control, and some revegetation with competitive native plant species. A preliminary economic analysis of the cost-effectiveness of this program arrived at a benefit-to-cost ratio of about 20.[17] Measurements of the impact of bitou bush on biodiversity values is necessary before a survey using the methods of choice modeling can be instituted to assess the economic value the Australian public places on biodiversity. Only then can the impacts of aliens on biodiversity be analyzed economically.

Given that there are few well-documented examples of the influences of aliens on the biodiversity of Australian ecosystems, and even fewer on the financial costs of those impacts, it is clear that more examples are needed. It is surprising that the impacts on species richness (as one measure of biodiversity) of some of Australia's major alien plants are still unknown,[23] even though they are recognized as significant weeds and major programs for their control are under way. This is particularly true for major alien species in northern Australia, such as the rubber vine (*Cryptostegia grandiflora*) and prickly acacia (*Acacia nilotica*). Adair and Groves[24] have suggested that, given active control programs for these aliens, it may be more important to determine threshold levels for declines in biodiversity and identify management barriers to invasion (or reinvasion), rather than simply measuring impacts in some generalized manner.

2.4 Impacts on human health

As with the impacts on agriculture and native biodiversity, aliens may have both positive and negative effects on human health. A relatively small number of the alien plant species introduced to Australia were imported deliberately for their putative herbal properties. I have already mentioned the case of horehound (*Marrubium vulgare*), but others include St. John's wort (*Hypericum perforatum*), variegated thistle (*Silybum marianum*), and, possibly, intentional introductions of dandelion (*Taraxacum officinale*) and pennyroyal (*Mentha pulegium*). Some species that were introduced accidentally are now found to have some benefit to human health. For instance, Paterson's curse (*Echium plantagineum*) is now cultivated in England because its seeds are high in omega-3 acids.[31]

Many plants contain compounds that cause physiological reactions in people that negatively affect their well-being and quality of life, and alien plants in Australia are no exception. Although the examples chosen here apply only to aliens present in Australia, I stress that they apply equally to the same plants present in other countries, whether they be considered alien or native to those environments.

Parthenium weed (*Parthenium hysterophorus*) is native to the southern United States and Mexico, as well as to Central and South America. The North American variant of this species has come to have a major impact on humans in central Queensland.[32] This variant contains parthenin, a sesquiterpene lactone that can cause allergic dermatitis in humans who have continued contact with parts of the plant, especially the flowers or the trichomes on leaves.[32] Cases of dermatitis have been reported from the United States, where the parthenin-containing variant is native, although most such cases, including some deaths,[33–35] have been reported from India, where the plant is an alien. The problem is less acute at this time in central Queensland, but contact dermatitis has been recorded from that region as well.[32] The dermatitis seems confined to adult males in most cases reported, presumably because of parthenin's interaction with male sex hormones. The human health problem caused by parthenium weed could become greater, given its potential to spread further in Australia.[36]

Skin irritation (urticaria) and allergic rhinitis have been reported to occur in Australia after repeated contact with Paterson's curse (*Echium plantagineum*).[37] The causative ingredient is unknown; it may be one of the pyrrolizidine alkaloids that are known to affect animal health[38] and are contained in the plant hairs or other particulate matter. The closely related *Echium vulgare* has been recorded as causing dermatitis, but not urticaria.[39] Other alien plant species known to cause forms of skin irritation include some of the brassicas (*Brassica alba* and *B. napus*), the nettles (*Urtica* spp.), *Erigeron* spp., stinkweed (*Inula graveolens*), and the garden plant *Rhus toxicodendron*,[39] which is not yet known to be naturalized in Australia.

A large number of Australians are affected chronically by hay fever (allergic rhinitis) and chronically or acutely by asthma (allergic bronchitis) as a result of inhaling allergenic pollen produced mainly in spring by a wide range of alien plants. Many of the introduced grasses (especially *Lolium* spp.) are a major source of such pollen, as are radiata pine (*Pinus radiata*), the ragweeds (*Ambrosia* spp.) in southern Queensland, pellitory (*Parietaria judaiaca*), especially in urban Sydney, the privets (*Ligustrum* spp.), the olives (*Olea europaea* and *O. africana*), the poplars (*Populus* spp.), and peppercorn (*Schinus molle*).[39] Medical statistics on the prevalence of this condition are confounded, however, because some native plants also produce allergenic pollen, such as *Atriplex spongiosa* and *Allocasuarina* spp.[39]

A few alien plants contain poisonous compounds, which if ingested may lead to serious illness and death. Examples include thornapples (*Datura* spp.), arum lily (*Zantedeschia aethiopica*), and hemlock (*Conium maculatum*). While contact with leaves of oleander (*Nerium oleander*) may cause eczema, ingestion of its leaves or flowers can cause death, because the toxic glucosides it contains have a digitalis-like action in humans. Gardner and Bennetts[39] report that people have even "been fatally poisoned by eating meat when oleander twigs were used as skewers or spits during its cooking."[39]

It is questionable whether the presumed positive effects of alien plants on human health by way of the increasing recognition in Australia of the value of herbals for well-being will ever outweigh the decrease in that same well-being caused by chronic allergenicity. For the present purposes, however, both are significant aspects of the overall impact of alien (and native) plants on the Australian public and the national economy.

2.5 Impacts on animal health

Many alien plants contain chemical compounds that affect animal health to varying extents, and hence agricultural productivity. Animal health and even animal survival after ingestion of such chemicals depend on many factors, including past grazing history, stage of plant growth, whether the diet is mixed or monospecific, and type of animal (whether monogastric or not), in addition to the level and nature of the actual toxic constituents in the alien plants. Different cultivars of the same plant species may contain different levels of the active compounds, as in subterranean clover (*Trifolium subterraneum*).[40] The following examples illustrate that impacts on animal health can be acute or chronic, negative or positive, depending on the particular situation.

Much of southern Australian animal production depends on pastures that contain the alien species subterranean clover (*Trifolium subterraneum*), phalaris (*Phalaris aquatica*), and ryegrasses (*Lolium* spp.), all of which are Mediterranean in origin. While such species form the very basis of animal productivity in Australia (and therefore have a strongly positive impact), under certain circumstances the different species can strongly influence animal health. Subterranean clover plants contain estrogens that can cause abortions and infertility in sheep that graze on pastures dominated by this species.[40] If the sheep are forced to eat only phalaris or ryegrass in their diet, and especially if they are suddenly moved to pastures in which these species are actively growing, they can suffer and even die from a condition known as "staggers" that is caused by the alkaloids present in phalaris or ryegrass.[40] In these cases, a mixed diet appears to overcome such drastic negative effects.

Other alien plants, such as variegated thistle (*Silybum marianum*), may contain large amounts of nitrate ions in spring that, if ingested in sufficient amounts, can cause blood poisoning.[40] The alien St. John's wort (*H. perforatum*), widespread in southern Australia, contains hypericin, which if ingested in sufficient quantities can cause photosensitization in animals, especially sheep. Because the flowers contain the highest concentrations of hypericin, grazing in pastures dominated by St. John's wort in late spring can lead to a suite of debilitating symptoms that reduce animal condition.[41] Affected animals recover when they no longer ingest St. John's wort. The equally widespread alien plant Paterson's curse (*E. plantagineum*) contains eight pyrrolizidine alkaloids that can interfere with liver function in grazing animals, especially sheep. These alkaloids can cause cumulative liver damage and even death if eaten in large enough quantities over a long enough period in the spring.[40] The problem is exacerbated if the same animals have access to spring-germinating plants such as heliotrope (*Heliotropium europaeum*), which also contains such alkaloids. In all of these examples, symptoms usually can be avoided and animal health maintained by careful pasture and animal management.

Some aliens can cause poisoning in animals, especially those that produce cyanogenic glycosides and glucosinolates, such as members of the Brassicaceae. In some cases these toxic compounds can be broken down in the rumen. Aliens in the Brassicaceae, Oxalidaceae, and Polygonaceae produce oxalates that may be acutely toxic. The many types of poisoning attributable to the many alien plants containing these compounds and some of the factors known to moderate acute or chronic symptoms are all discussed in several texts.[39,40] Anecdotal accounts of the effects of potentially poisonous plants straddle the boundary between fact and fiction. I repeat the words of Connor[42] in one of those texts: "Fact and myth conflict in the realm of poisonous plants; a false reputation for toxicity may, over the years, build up around a harmless plant, but true reports of poisoning, though made public, are sometimes overlooked."

The impacts of alien plants on animal health thus may be strongly negative (when the veterinary symptoms are acute) or weak (chronic states). On the other hand, the very basis of animal production, at least in southern Australia, depends on the positive effects of alien plants introduced from Mediterranean Europe in terms of the availability of high-quality forage, especially in winter, when native grasses are inadequate to sustain introduced livestock. Many of the alien plants that have negative effects on animal health and production evolved in the same Mediterranean region, and their properties have been selected for, either deliberately or inadvertently. It should come as no surprise that negative impacts in their native region are replicated when the same species are introduced to another region such as Australia.

2.6 Conclusions

The impacts of alien plants in Australia vary according to the ecosystem and the index considered. The monetary costs of aliens to the Australian economy are high, especially in terms of losses to agricultural productivity and to human well-being. With the prospects of more alien species naturalizing, and in view of increased resistance to herbicides in some species that are already naturalized, the future appears worrisome. Negative impacts on the diversity of Australian plants, animals, and ecosystems are many, but are largely unquantified scientifically, let alone in economic terms. A greatly increased effort in this regard will be essential to the provision of better information to decision-makers. Positive effects of alien species are reflected in increases in export markets and domestic growth in economic terms, as well as increases in quality of life for humans and in their enjoyment of native plants and animals, natural landscapes, and ecosystems. It is possible that the "services" provided by such natural landscapes may be valued more effectively in the near future. The balance between these two sets of impacts — negative and positive — will influence the future habitability of Australia for its people.

As mentioned earlier in this chapter, the number of naturalized aliens in the Australian flora represents about 10% of the total. I conclude, on the basis of the limited evidence currently available, that only a few of these 2681 species have an impact on the Australian economy, either directly or indirectly by way of effects on agricultural production, native biodiversity, or human welfare. Further, from the limited number of examples cited in this review, the impacts of still fewer alien plant species have been documented, and these refer only to those having major, chiefly negative, impacts. If more examples were available, any bias toward negative impact could be tested more validly and a more balanced appraisal arrived at for the overall impact of alien plants in Australia.

Research results, if acted upon, have the potential to reduce any negative effects of alien species and to increase any positive aspects of indigenous species in natural ecosystems. Research sometimes occurs only when the impacts of aliens have been recognized and even quantified. Future research and management should be aimed equally at those species only recently introduced or naturalized, before their negative or positive effects are expressed fully. Increased collation of knowledge of such species worldwide would help to identify any species not yet present in Australia on which quarantine and research should focus. After all, about the only generalization presently tenable is that if the alien has a negative impact elsewhere, it will most probably have a similarly negative effect if introduced to Australia. International efforts such as this book will help to refine such hypotheses, and thereby reduce the negative impacts of aliens on Australian ecosystems and the people who inhabit and manage them.

Acknowledgments

The first draft of this review benefited greatly from the comments of Jeremy Burdon, Dick Mack, Dick Medd, Trudi Mullett, Dane Panetta, Paul Weiss, and Tony Willis, to all of whom I am grateful.

References

1. Hibbert, M., *The Aussie Plant Finder 2000/2001*, Florilegium, Glebe, NSW, 2000.
2. Groves, R.H., et al., The naturalised non-native flora of Australia: its categorisation and threat to native plant biodiversity, in press, 2002.
3. Groves, R.H., Plant invasions of Australia: an overview, in *Ecology of Biological Invasions: An Australian Perspective*, Groves, R.H. and Burdon, J.J., Eds., 137–149, Australian Academy of Science, Canberra, 1986.
4. Kloot, P.M., The introduced elements of the flora of southern Australia, *J. Biogeog.*, 11, 63–78, 1984.
5. Gilmour, M.P. and Helman, C.E., The Vegetation of Norfolk Island National Parks, A report to the Australian National Parks and Wildlife Service, Canberra, 1989.
6. Pickard, J., Exotic plants on Lord Howe Island: distribution in space and time, 1853–1981, *J. Biogeog.*, 11, 181–208, 1984.
7. Griffin, G.E. and Nelson, D.J., Vegetation Survey of Selected Land Units in the Uluru (Ayers Rock–Mt. Olga) National Park, A report to the Australian National Parks and Wildlife Service, Canberra, 1988.

8. Costin, A.B., et al., *Kosciusko Alpine Flora*, CSIRO/Collins, Melbourne, 1979.
9. Humphries, S.L., Groves, R.H., and Mitchell, D.S., Plant invasions of Australian ecosystems, A status review and management directions, *Kowari*, 2, 1–134, 1991.
10. Specht, R.L., Major vegetation formations in Australia, in *Ecological Biogeography of Australia*, Keast, A., Ed., Dr. W. Junk, The Hague, 1981.
11. Groves, R.H., et al., Recent Incursions of Weeds to Australia, CRC for Weed Management Systems, Technical Series No. 3, Adelaide, 1998.
12. Combellack, J.H., Weed control pursuits in Australia, *Chem. and Ind.*, April, 273–280, 1987.
13. Jones, R., et al., The distribution, density and economic impact of weeds in the Australian annual winter cropping system, CRC for Weed Management Systems, Technical Series No. 4, Adelaide, 2000.
14. Medd, R.W. and Pandey, S., Estimating the cost of wild oats (*Avena* spp.) in the Australian wheat industry, *Pl. Prot. Q.*, 5, 142–144, 1990.
15. Marsden, J.S., et al., *Returns on Australian Agricultural Research*, 84–93, CSIRO, Melbourne, 1980.
16. Cullen, J.M., Bringing the cost benefit analysis of biological control of *Chondrilla juncea* up to date, *Proceedings of the VI International Symposium on Biological Control of Weeds*, Aug. 1984, *Vancouver*, Delfosse, E.S., Ed., 145–152, Agriculture Canada, Ottawa, 1985.
17. CIE (Centre for International Economics), The CRC for Weed Management Systems: An Impact Assessment. A report by the Centre for International Economics, CRC for Weed Management Systems, Technical Series No. 6, Adelaide, 2001.
18. Jones, R.E. and Vere, D.T., The economics of serrated tussock in New South Wales, *Pl. Prot. Q.*, 13, 70–76, 1998.
19. Nicholson, C., Patterson, A., and Miller, L., The Cost of Serrated Tussock Control in Central Western Victoria, A report to the Department of Natural Resources and Environment, Victoria, Melbourne, 1997.
20. IAC (Industries Assistance Commission), Biological Control of *Echium* Species (including Paterson's Curse/Salvation Jane), Industries Assistance Commission Report No. 371, Australian Government Publishing Service, Canberra, 1985.
21. Field, R.P. and Bruzzese, E., Biological Control of Blackberry, Report 1984/2, Keith Turnbull Research Institute, Department of Conservation, Forests and Lands, Frankston, 1984.
22. James, R. and Lockwood, M., Economics of blackberries: current data and rapid valuation techniques, *Pl. Prot. Q.*, 13, 175–179, 1998.
23. Adair, R.J. and Groves, R.H., Impact of environmental weeds on biodiversity: a review and development of a methodology, National Weeds Program, Environment Australia, Occasional Publication, Canberra, 1998.
24. Braithwaite, R.W., Lonsdale, W.M., and Estbergs, J.A., Alien vegetation and native biota in tropical Australia: impact of *Mimosa pigra, Biol. Conserv.,* 48, 189–210, 1989.
25. Griffin, G.E., et al., Status and implications of the invasion of Tamarisk (*T. aphylla*) on the Finke River, Northern Territory, *J. Environmental Manage.*, 29, 297–310, 1989.
26. Loope, L.L., et al., Biological invasions of arid land reserves, *Biol. Conserv.,* 44, 95–118, 1988.
27. Sorensen, B. and Jusaitis, M., The impact of bridal creeper on an endangered orchid, in W*eeds of Conservation Concern*, Cooke, D. and Choate, J., Eds., 27–31, Department of Environment and Natural Resources and Animal and Plant Control Commission, South Australia, Adelaide, 1995.
28. Willis, A.J., personal communication, 2001.
29. Panetta, F.D. and James, R.F., Weed control thresholds: a useful concept in natural ecosystems?, *Pl. Prot. Q.*, 14, 68–76, 1997.
30. Odom, D. and Sinden, J., personal communication, 2001.
31. Lloyd, S., personal communication, 2001.
32. Navie, S.C., et al., *Parthenium hysterophorus* L., in *The Biology of Australian Weeds*, vol. 2, Panetta, F.D., Groves, R.H., and Shepherd, R.C.H., Eds., 157–176, R.G. and F.J. Richardson, Melbourne, 1998.
33. Lonkar, A., Mitchell, J.C., and Calnan, C.D., Contact dermatitis from *Parthenium hysterophorus, Trans. St John's Hosp. Dermatol. Soc.*, 60, 45–53, 1974.

34. Subba Rao, P.V., et al., Clinical and immunological studies on persons exposed to *Parthenium hysterophorus L., Experientia*, 33, 1387–1388, 1977.
35. Towers, G.H.N., Allergic eczematous contact dermatitis from parthenium weed (*Parthenium hysterophorus), Proceedings of the 6th Australian Weeds Conference, Gold Coast, Queensland*, Wilson, B.J. and Swarbrick, J.T., Eds., 143–150, Queensland Weed Society, Broadbeach, 1981.
36. Williams, J.D. and Groves, R.H., The influence of temperature and photoperiod on growth and development of *Parthenium hysterophorus* L., Weed Res., 20, 47–52, 1980.
37. Burdon, J.J. and Burdon, J.G.W., Allergy associated with Paterson's curse, *Med. J. Aust.*, 2, 87–88, 1983.
38. Parsons, W.T. and Cuthbertson, E.G., *Noxious Weeds of Australia*, Inkata Press, Melbourne, 1992.
39. Gardner, C.A. and Bennetts, H.W., *The Toxic Plants of Western Australia*, West Australian Newspapers Ltd., Perth, 1956.
40. Everist, S.L., *Poisonous Plants of Australia*, Angus and Robertson, Sydney, 1974.
41. Bourke, C.A., Effects of *Hypericum perforatum* (St. John's wort) on animal health and production, *Pl. Prot. Q.*, 12, 91–92, 1997.
42. Connor, H.E., *The Poisonous Plants in New Zealand*, 2nd ed., Government Printer, Wellington, 1977.

chapter three

Non-indigenous vertebrates in Australia

Mary Bomford and Quentin Hart

Contents

0-8493-0836-4/02/$0.00+$1.50

Abstract

At least 80 species of non-indigenous vertebrates have established wild populations in Australia, and more than 30 of these species have become pests. Direct short-term economic losses caused by these species amount to at least $420 million per year, mainly in lost agricultural production. Overgrazing and browsing by introduced herbivores also contribute to land degradation, which reduces the capacity for future productivity in many areas, but the value of this degradation has not been estimated. In addition, grazing, predation, and competition by non-indigenous vertebrates are recognised as major threats to many endangered native species and communities, and these costs have not been quantified either.

Efforts to control non-indigenous vertebrate pests in Australia cost governments and landholders more than $60 million each year. Another $20 million or so is spent annually on research to control these pest species.

3.1 Introduction

Continents and large land masses are typically less susceptible to exotic species invasions.[1] Australia, however, is an exception. Over the past 200 years, many exotic animals have been deliberately imported, both legally and illegally, into Australia for transport, food, wool, leather, sport, pets, pest control, or by migrants who wanted to see familiar animals from their home countries. Other species, such as black rats and house mice, have been imported accidentally. Following import, some species, such as rabbits and foxes, were legally released into the wild; others, such as feral goats and pigs, escaped domestication; still others, such as Indian mynahs, were released illegally. Exotic vertebrate species that have successfully established wild populations on mainland Australia include 25 mammals, 20 birds (plus a further 7 species that are now established on offshore islands), 4 reptiles, 1 amphibian, and at least 23 freshwater fish species (Table 3.1).

Exotic species that have become established in Australia typically possess some or all of the following attributes: a good climate match between their overseas geographic range and Australia; a history of establishing exotic populations outside Australia; a high reproductive rate; a generalist diet; and an ability to live in human-disturbed habitats.[2,3] Disturbance of environments, particularly the clearing and modification of vegetation and the resulting fragmentation of habitats, have further facilitated the establishment and spread of many species.[4]

Many of the exotic species that have established widespread wild populations are now considered major pests of agriculture and the environment.[5–11] The law requires private landholders to control agricultural pests, and government conservation agencies have a responsibility to reduce the impacts of exotic species on endangered native species and communities.

Exotic animals have major direct impacts on Australia's livestock industries through predation and competition for pasture. In stable environments with reliable rainfall, the presence of feral herbivores often reduces livestock carrying capacity and the productivity of stock by reducing pasture biomass. In environments with highly variable rainfall, which are the norm in Australia's rangelands, pasture biomass varies greatly, and competition between stock and feral animals only occurs when pasture biomass is low.[12] It is at such times that feral animals can cause severe land degradation, because although livestock managers can destock paddocks, ferals continue grazing until large areas are almost completely denuded of vegetation. The result is permanent degradation of soil and pastures.

Exotic animals can also act as reservoirs and vectors for diseases affecting native wildlife, domestic stock, and humans. There are also potential losses that would occur if new diseases entered Australia and became established in feral animal populations. For

Table 3.1 Introduced exotic vertebrate species that have established widespread populations on mainland Australia and their pest status. Other introduced species have only established localised populations on the mainland[a] or have only established on offshore islands.[b]

	Serious pest	Moderate pest	Minor or non-pest
Mammals	European rabbit *Oryctolagus cuniculus* Feral goat *Capra hircus* Feral pig *Sus scrofa* European red fox *Vulpes vulpes* Dingo/feral dog *Canis familiaris* Feral cat *Felis catus* House mouse *Mus domesticus*	Feral horse *Equus caballus* Feral donkey *Equus asinus* Feral buffalo *Bubalus bubalis* Feral camel *Camelus dromedarius* Feral cattle *Bos taurus* Black rat *Rattus rattus*	European brown hare *Lepus capensis* Brown rat *Rattus norvegicus*
Birds	European starling *Sturnus vulgaris* Indian myna *Acridotheres tristis*	Mallard *Anas platyrhynchos* Rock dove (feral pigeon) *Columba livia* Spotted turtledove *Streptopelia chinensis* Blackbird *Turdus merula* House sparrow *Passer domesticus* European goldfinch *Carduelis carduelis* Senegal turtledove *Streptopelia senegalensis*	Cattle egret *Ardeola ibis* Skylark *Alauda arvensis* Tree sparrow *Passer montanus* Nutmeg manikin *Lonchura punctulata* Greenfinch *Carduelis chloris*
Amphibian	Cane toad *Bufo marinus*	—	—
Freshwater fish	European carp *Cyprinus carpio* Mosquitofish *Gambusia holbrook*i Mozambique tilapia *Oreochromis mossambicus*	Weather loach *Misgurnus anguillicaudatus* Tench *Tinca tinca* Redfin perch *Perca fluviatilis* Rainbow trout *Oncorhynchus mykiss* Brown trout *Salmo trutta*	Goldfish *Carasius auratus* Guppy *Poecilia reticulata*

[a] Localised mainland populations. Birds: ostrich, *Struthio camelus*; red-whiskered bulbul, *Pycnonotus jocosus*; song thrush, *Turdus philomelos*; mute swan, *Cygnus olor*; peafowl, *Pavo cristatus*; Barbary dove, *Streptopelia risoria*; redpoll, *Carduelis flammea*. Mammals: Asian house rat, *Rattus tunezumi*; Indian palm squirrel, *Funambulus pennanti*; chital deer, *Cervus axis*; rusa deer, *Cervus timorensis*; banteng, *Bos javanicus*; hog deer, *Cervus porcinus*; fallow deer, *Dama dama*; red deer, *Cervus elaphus*; feral sheep, *Ovis aries*; sambar deer, *Cervus unicolor*. Reptiles: house gecko, *Hemidactylus frenatus*; mourning gecko, *Lepidodactylus lugubris*; red-eared slider, *Trachemys scripta elegans*; flowerpot snake, *Ramphotyphlops braminus*. Freshwater fish: three-spot gourami, *Trichogaster trichopterus*; red devil/Midas cichlid, *Amphilophus citrinellus*; three-spot cichlid, *Cichlasoma trimaculatum*; Burton's haplochromine, *Haplochromis burtoni*; Niger cichlid, *Tilapia mariae*; roach, *Rutilus rutilus*; one-spot live bearer, *Phalloceros caudimaculatus*; sailfin molly, *Poecilia latipinna*; platy, *Xiphophorus maculatus*; brook trout, *Salvelinus fontinalis*; green swordtail, *Xiphophorus helleri*; chinook salmon, *Oncorhynchus tshawytscha*; oscar, *Astronotus ocellatus*.

[b] Offshore island populations. Birds: wild turkey, *Meleagris gallopavo*; helmeted guinea fowl, *Numida meleagris*; red jungle fowl, *Gallus gallus*; California quail, *Lophortyx californicus*; ring-necked pheasant, *Phasianus colchicus*; chaffinch, *Fringilla coelebs*; Java sparrow, *Lonchura oryzivora*. Mammal: Pacific rat, *Rattus exulans*. Reptiles: wolf snake, *Lycodon aulicus*; skink, *Lygosoma bowringii*.

Sources: birds[16]; mammals[17]; reptiles[18]; fish[19] and P.J. Kailola (pers. comm.); plus supplementary information referenced in text.

example, rabies could establish in wild dogs and foxes, and foot-and-mouth disease in feral pigs and feral goats. Prevention of and preparedness for the possible entry of such exotic diseases is costly.

Environmental costs, such as threats to the survival of native species through competition and predation, are hard to establish and quantify. This is because the threat posed by introduced species is often one of a suite of factors threatening native species survival, with habitat disturbance and destruction and changed fire and water regimes also playing a significant role for many native species that are threatened by introduced vertebrates. Changes in the composition and cover of the vegetation caused by grazing vertebrate pests are likely to influence populations of ants, termites, and topsoil micro-arthropods. Vegetation changes may have long-term effects on efforts to maintain soil structure. Some exotic species hybridize with native species and so pose a threat to their survival.

As well as being pests, many introduced vertebrates are valued as a resource.[13] Hunters and anglers value deer and trout as important game species, and in some areas fees are charged to take them. Feral horses, camels, goats, and pigs are captured or shot for their meat and hides and are an important commercial resource. Many landholders make significant profits from their harvests that can offset other control and damage costs. There is a valuable export industry in feral pig and feral goat meat for human consumption. The total value of exported goats and goat products was about $30 million in 1992–93[14]; this figure is probably now much higher because of increased prices for goat meat. The feral pig harvesting industry is valued at $10 million to $20 million per year.[13] Deer, camels, horses, and goats are sometimes harvested for domestication. In the past, Australia was one of the world's most important exporters of fox and cat pelts, which generated significant export income, but with the decline in world fur trade, this is no longer the case. Rabbit fur is used to make felt hats. Rabbits and cats are also a significant subsistence food source for some Aboriginal groups, providing high-quality fresh food and economic savings to the communities.[15] Trout are a significant resource for recreational angling, and carp are harvested commercially for human consumption and for the production of fish bait, pet and stock food, and fertilizer.

This chapter addresses the impacts and economic costs of wild-living, introduced vertebrates in Australia. Harm caused by domestic animals is not considered here, and the cost of environmental harm to indigenous species and communities is not quantified. All dollar values presented in this chapter are expressed in 1999–2000 Australian dollars.

3.2 Damage and control costs of major pest species

The figures presented for agricultural damage costs are based on extrapolations of government agency estimates, landholder surveys, and other information referenced in the text.

Only qualitative accounts are given for environmental damage caused by introduced vertebrates in Australia, and no attempt has been made to price such impacts — for example, the threat many introduced vertebrates pose to endangered native Australian species, or land degradation caused by overgrazing and browsing. We recognize that the cost of this damage, in many people's perception, is probably at least equivalent to the short-term agricultural damage for which costs are estimated in this chapter.

The figures presented for agricultural and environmental damage control costs are based on estimates supplied by government agencies. When such estimates are unavailable, we assume the spending is equivalent to that in areas with similar pest numbers for which data are available. Estimates for landholder spending are based on the assumption that the average Australian landholder spends $250 per farm per year, a conservative estimate that takes into account the following factors:

- Not all enterprises have pest problems, and pest damage and control activity vary from year to year, particularly for damage caused by mice and birds.
- A variety of economic and social factors may lead many farmers to neglect pest control even when damage is evident.
- Some pest control actions, such as exclusion fences and nets, have a high initial outlay followed by relatively low annual maintenance costs.

As with damage control costs, the figures presented for research costs for vertebrate pest control are based on available records or estimates supplied by government and research agencies. When agencies were unable to provide estimates for individual species or states, we conservatively estimated expenditures based on equivalent spending for that pest in similar areas.

Control and research cost estimates are likely to be very conservative, as they do not fully account for salaried positions and associated infrastructure.

3.2.1 Rabbit (Oryctolagus cuniculus)

European rabbits were brought to Australia by the first European settlers for food, fur, and skins, and they have since become Australia's most widespread and significant pest animal. The rate of spread of the rabbit in Australia was the fastest of any colonizing mammal anywhere in the world, as rapidly as 100 km per year in the rangelands. The scale of the impact of the rabbit in Australia is considered to be unique in the history of exotic animal introductions.[20]

Rabbit grazing results in fewer livestock, reduced wool production, lower lambing percentages, lower weight gain, more frequent breaks in the wool, and earlier stock deaths during droughts. The extent to which rabbits reduce the carrying capacity of land for livestock is not well quantified. About 12 to 16 rabbits eat as much as one sheep does, but competition between sheep and rabbits only occurs when pasture biomass is relatively low, for example, less than 250 kg per hectare in the sheep rangelands of New South Wales, a condition that usually occurs only after periods of low rainfall.[21]

Rabbit numbers declined greatly in Australia in 1997–98, particularly in lower-rainfall areas, due to the release of a biological control agent, rabbit hemorrhagic disease (RHD). So far there has been little recovery of rabbit populations. Before RHD was released, average densities of rabbits annually consumed 10 tons of dry pasture per km^2,[22] and rabbits took more pasture than sheep in many areas.[20] Sheep farmers were often unable to rest pastures, because if stock were taken off, rabbit and kangaroo numbers would build up. Thanks to recent declines in rabbit numbers caused by RHD,[23] this high consumption rate is likely to have dropped, particularly in low-rainfall areas.

Rabbit grazing leads to pasture degradation and a lack of regeneration or even the destruction of important fodder trees, shrubs, and perennial grasses, particularly during and following droughts. Perennial grasses and shrubs are replaced by less stable annual species. Rabbits also expose extensive areas of bare soil that leads to soil erosion, loss of soil fertility, and siltation of dams.[20]

Rabbits damage crops, too, including cereal and horticultural crops. Rabbits also cause extensive losses to forestry and tree plantations, preventing regeneration and damaging tree plantings. This increases the cost of tree planting programs because of the need to erect tree guards. Damage from browsing rabbits can approximate one year's loss of growth, equivalent to \$800/ha at clear-felling, and rabbit control costs in private forests can run as high as \$80/ha during the period when trees are vulnerable to rabbit damage.[20]

Rabbits threaten the survival of at least 17 native plants.[8] The replacement rate of many of the trees and shrubs in the southern rangelands was not sufficient to prevent their disappearance in the long term prior to the release of RHD.[20] Since RHD was released in 1996, many shrub and tree species have regenerated, but it is too early to determine if RHD will keep rabbit numbers low enough for long enough to allow these new plants to survive to maturity.[24] Low rabbit numbers need to be sustained to prevent the extinction of several threatened native tree species. Even an apparently successful germination can be wiped out by rabbits as many as 15 years after the event.[25] Mulga (*Acacia aneura*), which lives to 250 years, is very palatable to rabbits and stock.[26] It is the most important drought fodder tree in Australia. Rabbits, not domestic stock, are preventing regeneration of mulga.

Rabbits also threaten the survival of many native animal species, such as the greater bilby (*Macrotis lagostis*), a small burrowing mammal, through competition for food and habitat destruction.[8,20] The destruction of sandhill canegrass by rabbits reduces populations of birds, too, such as the Eyrean grass wren (*Amytornis goyderi*). Overgrazing by rabbits modifies habitats, making them unsuitable for the endangered plains-wanderer (*Pedionomus torquatus*), a small nocturnal wader. The distribution and abundance of many species of birds and other animals will be seriously affected if rabbits cause a long-term decline in tree and shrub populations in the rangelands.

Rabbits occur on 48 Australian islands, and their environmental impacts there can be catastrophic. Rabbits introduced onto Phillip Island caused the extinction of an endemic parrot (*Nestor productus*) and two endemic plants and severely reduced other vegetation. Since the eradication of rabbits on Phillip Island in 1986, the vegetation has shown considerable recovery. Many islands are important for seabirds, the nesting sites of which are often affected by rabbits.[20] For example, the Gould's petrel (*Pterodroma leucoptera*) only nests on Cabbage Tree Island, and its long-term future is in doubt because of vegetation changes caused by rabbits. Rabbits also maintain large predator populations. For example, winter-nesting seabirds no longer nest on Macquarie Island because of cat predation. Shooting of cats was ineffective, but rabbit control is reducing cat numbers.[27]

3.2.1.1 Rabbit agricultural costs

Estimates of agricultural losses caused by rabbits vary. Annual crop losses to rabbits in South Australia were estimated at $7.5 million,[28] annual losses to Australian sheep production due to rabbits were estimated at $130 million,[29] and annual losses to Australian agricultural production were estimated at $600 million,[30] including $300 million for wool losses, $70 million for sheep meat, $150 million for cattle, and $80 million for crops. These estimates assume that markets would be available for any additional agricultural production occurring in the absence of rabbits, but this may not in fact be true, particularly for wool. Since these estimates were made, the value of wool has varied, and rabbit numbers have declined because of RHD. The RHD-induced decline in rabbit numbers has been estimated to result in a benefit of at least $165 million a year to wool and sheep producers in Australia.[31] It is probable that annual losses to sheep and wool production due to rabbits currently are around $100 million per year. Other agricultural industries have probably benefited less from RHD, so total agricultural losses due to rabbits may still be at least $200 million a year.

3.2.1.2 Rabbit agricultural and environmental control and research costs

Australian government agencies spend an estimated $10 million or more per year on rabbit control. It is likely that landholders spend at least an equivalent amount, so a conservative estimate for total annual rabbit control costs is more than $20 million per year. Rabbit-control research costs have been around $5 million per year for the past five years, with much of the efforts being directed toward biological control.

3.2.2 Fox (Vulpes vulpes)

Fox predation on lambs can be significant. A study in Victoria indicated that foxes took 7% of the lambs.[32] Foxes reduced lambing success by an average of more than 25% on two sheep properties in South Australia.[33] Foxes may account for up to 30% of all lamb mortalities in some areas in western New South Wales where foxes are common.[34] High lamb losses can occur where lambing is out of step with or isolated from neighboring flocks. Foxes also prey on calves, goat kids, and free-range poultry, although these losses are as yet unquantified.[32]

The fox is a serious threat to native wildlife, including many rare and endangered species.[9,32] In Western Australia, the removal of foxes in some areas has caused substantial and consistent population increases in some marsupial species.[32] After eight years of fox control in two rock wallaby (*Petrogale lateralis*) colonies, populations increased four- to fivefold.[35] Following fox control on Dolphin Island, the sightings of Rothschild's rock wallabies (*Petrogale rothschildi*) increased nearly thirtyfold. Following fox control for five years in Dryandra State Forest, numbat (*Mymecobius fasciatus*) numbers increased significantly. In New South Wales, fox control has been shown to increase mallee fowl (*Leipoa ocellata*) survival.

Foxes were identified as a factor limiting success in seven out of ten mainland reintroductions of endangered native mammals.[36] Reintroductions to islands and mainland sites that had predators such as foxes and cats had a success rate of only 8%, compared to a success rate of 82% on island sites with no predators.

There is no practical method for assessing the economic impact of foxes on wildlife, although the impact may be considerable, particularly for ecotourism for viewing native species such as kangaroos, koalas, and penguins and rarer native species in wildlife parks, as well as in the wild. Expensive fox control is often needed to allow this to occur.[32] For example, on Phillip Island there is a $50 million tourist industry built on viewing little penguin (*Eudyptula minor*) populations. For the period 1987 to 1992, 202 foxes were destroyed, while in the same period 499 penguins were identified as having been killed by foxes.[32]

3.2.2.1 Fox agricultural costs

If we assume that foxes take 5% of all viable lambs Australia-wide, the annual cost of fox predation on lambs is around $40 million.

3.2.2.2 Fox agricultural and environmental control and research costs

Governments spend an estimated $2 million on fox control annually, and landholders probably spend another $5 million. Annual fox control research costs are around $4 million per year, and are mainly directed at baiting foxes with poisons or immunocontraceptives.

3.2.3 Feral goat (Capra hircus)

Feral goats are a cost to primary producers because they contribute to long-term changes to perennial vegetation caused by overgrazing, especially during droughts. Feral goats contribute to damage to vegetation, soils, and native fauna in the large areas of pastoral land that are overgrazed, although their share is generally less than that of other herbivores.[14] Feral goats also affect perennial vegetation by eating established plants and by preventing regeneration of seedlings. Browsing by goats can kill established plants by defoliation. Goats are particularly prevalent in habitats with perennial shrubs and trees, many of which are palatable, and most of these are ultimately eaten by goats.

At a density of two per km^2 (the average density of goats in Australia in the early 1990s), feral goats annually consume 0.73 tons of dry matter per km.[2,14] This consumption by goats includes unpalatable vegetation and woody tissue not normally eaten by livestock and native fauna, and if not eaten by goats, much of this matter would be consumed by invertebrates, small vertebrates, and decomposers. Rangelands with 240 mm of annual rainfall can on average support at least 20 goat-sized herbivores per km^2. Therefore, at the average density of two per km^2, feral goats would consume about 10% of the food eaten by the suite of large herbivores present.[14]

At four sites in southwest Queensland monitored between 1994 and 1997, feral goats represented 3 to 30% of the total grazing pressure, and all four sites had a total grazing pressure above the estimated safe total carrying capacity.[37] Livestock and kangaroos were the other main contributors to grazing pressure. In 1997–98 the average cost of harvesting a feral goat was around $2, and the farm gate price was $16 to $38 per goat. At this high price, feral goats are not considered a pest by most landholders, but are harvested for profit.

Costs to other production values include the costs to farmers of keeping feral goats from mating with their quality domestic goats, and costs to production foresters caused by goat damage to their seedlings. Feral goats can damage fences and contaminate bodies of water as well. The presence of feral goats in Australia also increases the contingent cost of ensuring against the outbreak of exotic diseases of livestock.[14]

Feral goats affect native fauna primarily by competition for resources such as food, water, and shelter, and by contributing to changes in ecosystems, although these effects have not been quantified.[10,14] Feral goats occur on a number of Australian islands, including Lord Howe Island, which is a World Heritage Site.

3.2.3.1 Feral goat agricultural costs

Annual losses to agricultural production, mainly the ranching industry, due to feral goats are around $20 million per year.[14]

3.2.3.2 Feral goat agricultural and environmental control and research costs

Governments spend an estimated $2 million on feral goat control annually. Farmers also control feral goats, but the average price for harvested goats in recent years has been more than $20 per animal,[37] so the profits farmers make from selling the feral goats makes their control cost-neutral. Annual feral goat control research costs are around $1.5 million per year.

3.2.4 Feral pig (Sus scrofa)

Feral pigs prey on newborn lambs. Feral pig predation on newborn lambs has been measured at 32%[38] and 18.7%.[39] Feral pigs also eat or root up pasture that could otherwise be used by domestic stock. Pasture destruction may be considerable in areas of higher stable rainfall but is likely to be small (less than 3%) in more arid, variable-rainfall areas.[12]

Pigs damage water sources, including bore drains and bore outlets, water supply channels in irrigation areas, floodgates and levy banks around flood-prone property, and water troughs and distribution pipes.[40] They also foul farm dams and waterholes by wallowing and defecating.[12] Feral pigs damage fences, too. Feral pigs also reduce yields of cereal grain, sugarcane, and fruit and vegetable crops.[40]

The most important environmental impacts that feral pigs are likely to have are habitat degradation and predation. Feral pig rooting leads to erosion and loss of regenerating forest plants. Erosion caused by pig rooting also leads to reductions in water quality and silting of downstream swamps.[12] Feral pigs eat native plants, including their foliage and stems, fruits and seeds, and rhizomes, bulbs, tubers, and roots. The effect of pigs on rare or endangered plants and on plant succession in Australia is unknown. Animals reported to be eaten by feral pigs include earthworms, amphipods, centipedes, beetles and other arthropods; snails, frogs, lizards, snakes, the eggs of the freshwater crocodile (*Crocodylus johnstoni*); turtles and their eggs; and small ground-nesting birds and their eggs.[12] Feral pigs reportedly also destroy the nests and eat the eggs and young of larger ground-nesting birds, such as cassowaries (*Casuarius casuarius*), scrubfowl (*Megapodius reinwardt*), and brush turkeys (*Alectura lathama*). Feral pigs may also compete with brolga (*Grus rubicundus*) and magpie geese (*Anseranas semipalmata*) for food. The effects of this predation and competition on animal populations are unknown.

Feral pigs may help spread root-rot fungus (*Phytophthora cinnamomi*), which is responsible for dieback disease in native vegetation. The spread of the fungus has also been associated with soil disturbance and reduction of litter cover caused by pigs.

Feral pigs can be hosts or vectors of several diseases and parasites currently present in Australia that affect livestock and humans. The major diseases of concern are leptospirosis (*Leptospira* spp.), brucellosis (*Brucella suis*), melioidosis (*Pseudomonas pseudomallei*), tuberculosis (*Mycobacterium* spp.), porcine parvovirus, sparganosis (*Spirometra erinacei*), Murray Valley encephalitis, and other arboviruses.[12] Feral pigs are the wild vertebrate species of most concern in Australia because of their potential to harbor or spread exotic diseases and parasites of livestock, should such diseases breach Australia's quarantine barriers.[12] The most significant exotic disease of concern is foot-and-mouth disease (FMD), a highly contagious viral disease of ungulates (including pigs, cattle, sheep, goats, and deer). Other diseases of concern include swine vesicular disease, African swine fever, Aujeszky's disease, trichinosis (or trichinellosis), and classical swine fever. Outbreaks of any of these diseases or parasites could have severe repercussions for livestock industries.[41] For example, an outbreak of FMD could cost Australia more than $3 billion in lost export trade, even if the outbreak of the disease were eradicated immediately.[12] If the outbreak persisted, continuing losses could be $300 million to $400 million a year, depending on whether trade was affected in just one state or territory or countrywide.

3.2.4.1 Feral pig agricultural costs

Annual losses to agricultural production, mainly the pastoral industry, due to feral pigs are around $100 million.[12] This includes a contingent cost of ensuring against the outbreak of exotic diseases of livestock of about $5 million per year.

3.2.4.2 Feral pig agricultural and environmental control and research costs

Governments spend an estimated $2.5 million on feral pig control annually, and landholders probably spend an equivalent amount, bringing the total control costs to around $5 million a year. Research on feral pig control averages about $1.5 million per year.

3.2.5 House mouse (Mus musculus)

Mice form plagues in grain-growing areas, and they do the most damage when winter crops are sown, when they flower and set seed, and when summer crops mature.[42] Nearly all crop types can be damaged during mouse plagues, particularly grain and oilseed crops and many horticultural crops.

Apart from the damage to crops, mice damage farm equipment, machinery, and vehicles; building insulation; household items; and personal possessions. The average loss to grain growers of the three most recent major mouse plagues is estimated conservatively to be about $48 million.[42] Major plagues now occur every year or two.[42] In addition, there is damage from local plagues, such as one in 1994 in the Murrumbidgee Irrigation Area of New South Wales, which caused an estimated $7 million in damage to rice, maize, and soybean crops.[43] Even at non-plague densities, mice can cause millions of dollars worth of damage to crops.

Mouse plagues also cause losses to pig and poultry farmers in the form of increased feed costs (which rose as much as 50% during the 1993 Victoria plague), stress, and injuries from attacks by mice. The total losses experienced by intensive livestock producers during the 1993 mouse plague were on the order of $600,000 in the worst-affected area in north-western Victoria.[42] Depletion of grazing pastures is commonly reported during mouse plagues.[42]

Mouse plagues also cause damage in rural townships, including damage to equipment (particularly electrical equipment), spoiling and consumption of products, lost business opportunities from not stocking and selling products at risk (such as packet food and grain), and the costs of protecting goods and of cleaning to maintain health and hygiene standards. The most significant cost is the labor required to mouseproof, bait, trap, clean, and search for and dispose of carcasses. Rural suppliers, food retailers, hospitality outlets, schools, hospitals, and telephone communications and grain-handling facilities record high losses. Estimated total costs to retailers, community services, and residents in a 1993 plague in South Australia exceeded $1 million.[42]

During a mouse plague in 1984, the annual rodenticide market was valued at $27 million, compared with $5 million in a non-plague year.[44]

3.2.5.1 Mouse agricultural costs

The average annual loss to Australian grain growers is at least $27 million.[42] Additional losses to other agricultural products and off-farm losses due to mouse plagues probably average at least $500,000 per year.

3.2.5.2 Mouse agricultural and environmental control and research costs

Governments spend an estimated $2 million on mouse control annually. Landholders spend more, with their total annual control costs coming to about $8 million a year. Government spending on mouse control research is an additional $2.5 million, of which about half is directed toward developing an immunocontraceptive biocontrol agent.

3.2.6 Wild dog (Canis familiaris)

The threat of predation of livestock by wild dogs (including feral dogs, dingoes, and their hybrids) determines the distribution of sheep and cattle in Australia, and sheep are not run in many areas that would otherwise be suitable for them in the absence of wild dogs. Wild dogs often kill far more sheep than they eat, so even a few wild dogs can cause heavy stock losses. In a survey in eastern Australia, 12% of the respondents said they reduced sheep numbers or did not run sheep in order to minimize wild dog predation.[45] Sheep are the most commonly attacked animal, followed by cattle and goats.[46] Attacks on young calves are the major cause of cattle losses to wild dogs. It has been suggested that in Queensland, calf losses due to predation by wild dogs may be up to 30%.[47]

Losses other than direct maimings and killings of livestock caused by wild dogs are difficult to quantify. Wild dogs sometimes chase sheep without following through with an attack, which can lead to harm such as mismothering of lambs. Rams sometimes survive severe scrotal injuries, with some being fully castrated by wild dogs.

Predation by wild dogs may have an impact on the survival of remnant populations of endangered fauna. For example, predation by the dingo was implicated in the extinction of the Tasmanian native-hen (*Gallinula mortierii*) from mainland Australia.[48] Hybridization between introduced feral dogs, which were introduced by Europeans about 200 years ago, and the dingo (*Canis familiaris dingo*), which was introduced to Australia by indigenous peoples about 3500 years ago, threatens the survival of the dingo on the mainland.

In recent years, dingoes have become a major tourist attraction at sites in "outback" Australia and on Fraser Island in particular. Consequently, many visitors and residents feed dingoes to encourage contact for close viewing and photographs. This has led to many dingoes and other wild dogs losing their fear of people, and occasionally attacking them.[45]

The prevalence of hydatidosis (causal agent *Echinococcus granulosus*), a fatal disease in humans, is often linked to sylvatic cycles in wild dogs and wildlife. Hydatidosis also leads to the condemnation of offal from up to 90% of slaughtered cattle from endemic areas in Victoria.[45] In southeastern Queensland, bovine hydatidosis prevalences of 2.2 to 55.7% have been reported. Prevalences of 0.5 to 7% were found in northeastern Victoria, despite an extensive hydatid control program aimed at domestic and farm dogs.[45] Where wild dogs co-occur with foxes — for example, in coastal southeastern Australia — the control of human hydatidosis becomes difficult.

Wild dogs and foxes pose a risk of maintaining and spreading rabies if it were introduced to Australia. If rabies were to become endemic in Australia, interaction between free-roaming domestic dogs and wild dogs would be the most likely avenue for rabies transmission to humans.

3.2.6.1 Wild dog agricultural costs

Annual losses to agricultural production, mainly the ranching industry, due to wild dogs are at least $20 million.[45]

3.2.6.2 Wild dog agricultural and environmental control and research costs

Governments spend an estimated $4 million or more on wild dog control annually, and landholders probably spend at least $2.5 million in direct control; in addition, maintenance of the wild dog control fence costs as much as $10 million per year. This fence, which was established in the early 20th century, currently extends for 5614 km across three states. The current costs of replacing or extending the fence can be as high as $8500/km, and ongoing inspection and maintenance costs $300 to $2000/km per year. Wild dog control research expenditure is about $1.5 million per year.

3.2.7 Feral cat (Felis catus)

Field experiments have shown that cat predation causes major declines in small vertebrate populations.[49] The effects of feral cat predation on native fauna were evaluated.[50] On the Australian mainland, 38 species of mammals, 47 species of birds, 48 species of reptiles, and 3 species of amphibians have been recorded in the diet of feral cats. Nineteen species of endangered or vulnerable mammals, 6 species of endangered birds, and 2 species of endangered or vulnerable reptiles are at high risk from feral cat predation on mainland Australia.[50] On offshore islands, 4 species of endangered or vulnerable birds are at high risk from feral cat predation. There is also a potential for feral cats to compete with native predators, but no scientific evidence is available.

Two pathogens that use the cat as a definitive host can cause disease in many native species.[50] *Spirometra erinacei* is a large tapeworm that infests the gut of carnivores; *Toxoplasma gondii* produces toxoplasmosis, which can cause lethargy, poor coordination, blindness, and death. Antibodies to toxoplasmosis and signs of infection have been recorded in at least 30 species of native mammals[51] and in several species of birds. Toxoplasmosis can also be transmitted from feral cats to domestic stock and humans, and it can cause lamb carcasses to be condemned.[52] Feral cats can also assist in the spread of sarcosporidiosis, which causes economically significant condemnations of sheep in Australia, particularly on Kangaroo Island.

3.2.7.1 Cat environmental damage costs

No estimates have been made of the cost of feral cat predation on native fauna. Cats are not considered an agricultural pest.

3.2.7.2 Cat environmental control and research costs

Governments spend at least $1 million on feral cat control annually. Annual cat control research costs also amount to approximately $1 million.

3.2.8 Feral donkey (Equus asinus)

Donkeys compete with stock for water and pasture in northern Australia, and they also denude ground cover and contribute to erosion.[6] The effect of donkeys on native fauna is unknown, but habitat destruction may be a problem. Their potential role in spreading livestock diseases is limited by their remoteness.

Air and ground shooting campaigns have been conducted, and large numbers of donkeys have been shot, including an estimated 76,000 between 1980 and 1982.[6] Reinfestation is a major problem. Current government spending on feral donkey control is relatively minor, and no estimates are available of landholder spending.

3.2.9 Feral horse (Equus caballus)

Horses on rangelands destroy fences, foul watering points, and consume fodder, hence reducing productivity for livestock.[53] Their grazing and fouling of water may also have a detrimental impact on native species. Feral horses also have a potential role in the spread of exotic diseases, although this is limited by their remoteness from significant domestic horse populations. Control costs are not quantified, but methods include rounding up animals and dispatching them to slaughterhouses, and shooting, mainly from helicopters.[6,53]

3.2.10 Feral buffalo (Bubalus bubalis)

In 1985–86 feral buffalo numbers in northern Australia were estimated at 350,000. Since then, their numbers have been greatly reduced by a large-scale control program to eliminate brucellosis and bovine tuberculosis from Australia, the Brucellosis and Tuberculosis Eradication Campaign.[6] The spread of these livestock diseases posed a threat to Australia's meat industry.[54] Buffalo are shot from helicopters or rounded up into corrals, using four-wheel-drive vehicles and helicopters, for transport to slaughterhouses.

Prior to their widespread control, feral buffalo extensively damaged freshwater swamps by forming trails between tidal rivers and floodplains that allowed sea water to enter and kill large areas of paperbark (*Melaleuca* spp.) forest.[6] They also selectively ate native grass (*Hymenachne acutigluma*) and changed the structure of monsoon forests. They trampled nesting grounds of the rare pig-nosed turtle (*Carettochelys insculpta*). Buffalo damage was especially significant in areas that have major conservation values, such as Kakadu National Park.[6]

3.2.11 Feral camel (Camelus dromedarius)

Camels can damage fences and watering points, and there is likely to be some competition with livestock where camels reach higher densities. The potential role of camels in spreading livestock diseases is probably insignificant at low population densities. It is possible that browsing and grazing by feral camels reduces shelter for small desert mammals. Camels are sometimes controlled on cattle stations, usually by trapping at water points, rounding up, and shooting.[6]

3.2.12 Black rat (Rattus rattus)

Black rats cause losses as high as 30% in macadamia orchards in some years, equivalent to around 100 tons or $350,000 worth of nuts on some individual farms.[42] Annual average losses are probably around 5%, so at this level the total national damage is on the order of $3 million per year.[42] The macadamia industry in Australia is rapidly expanding, so this figure could increase substantially unless effective control methods are developed and implemented. Black rats also damage citrus, avocado, and banana crops, but the extent and severity have not been evaluated.

The potential impact on owls (*Circus, Ninox*, and *Tyto* spp.) from the use of anticoagulant rodenticides in orchards has raised concern.[42]

Predation by black rats on offshore islands is thought to adversely affect native species, including eight native birds, two reptiles, and one insect,[55] and is also thought to have contributed to the extinction of two additional bird species. Competition by black rats on islands may also adversely affect two mammal species.[55]

3.2.13 Cane toad (Bufo marinus)

The diet of cane toads is primarily composed of arthropods, and effects on invertebrate communities have not been quantified, but it is possible that the toads compete for food with some native species. Cane toads also take bees around commercial hives, but the economic costs are unquantified.

Cane toads may compete with native species for habitat.[56] They may also eat native frogs and their eggs, although this appears to be uncommon.[57] Because cane toads are toxic, they can poison native predators that attempt to eat them. Native frogs that eat cane toad eggs or tadpoles could be poisoned, but there is little evidence for this.[58] There is anecdotal evidence that local populations of four quoll species (*Dasyurus* spp.) and 16 goanna species (*Varanus* spp.) that eat cane toads are threatened,[59] but there is little clear evidence that cane toads are the principal cause of declines in these species.[7]

3.2.13.1 Cane toad control and research costs

Governments spend around $500,000 on cane toad research annually. There are no significant cane toad control programs.

3.2.14 European starling (Sturnus vulgaris)

European starlings cause high levels of damage to fruit crops, particularly grapes and stone fruit, and they attack winter-sown cereals at germination.[60] Crop damage due to starlings is difficult to quantify, as it is usually combined with damage caused by other birds, but average bird damage losses to grape crops have been estimated at around 10%, and starlings would be a significant contributor to this total damage.[61] Damage is caused by grape removal or damage or by secondary spoilage through molds, yeasts, bacteria, and insect damage. Bird damage can also cause undesirable early harvests and the resultant downgrading of otherwise premium fruit.

Starlings also take feed from cattle feedlots, piggeries, and poultry farms.[60] In addition to the food they take, they spoil much more with their droppings. There is also a risk that they could assist in the spread of diseases such as salmonella and tuberculosis at these sites.[62] Starlings nest in roof and ceiling cavities, causing fire hazards and parasite infestations, and they deface buildings with their droppings.

European starlings compete with native birds for food and nesting hollows, in many cases displacing native species.[60,63]

3.2.14.1 Bird agricultural costs

Starlings are the worst introduced bird pest of agriculture in Australia. No estimates are available of total agricultural losses caused by introduced birds, but they are likely to be at least $10 million per year.

3.2.14.2 Bird control and research costs

Governments and landholders spend an estimated $3 million or more per year on bird pest control, and probably about half of this is directed at introduced birds, such as starlings. An intensive monitoring and control program in Western Australia has prevented starlings from establishing there. Governments spend about $500,000 annually on bird pest research.

3.2.15 House sparrow (Passer domesticus)

House sparrows damage fruit, vegetable, grain, and oilseed crops.[16,61] They also deface buildings with their droppings and block gutters and downpipes, although this damage is relatively minor. Sparrows congregate at feedlots, piggeries, and poultry farms, and they could assist in the spread of diseases such as salmonella and tuberculosis at these sites.[62]

House sparrows are aggressive around their nests and compete with native birds for nest sites and food.[64]

3.2.16 Indian myna (Acridotheres tristis)

Indian mynas are still colonizing Australia and do not yet occur in large numbers. Mynas compete with native birds, such as the crimson rosella (*Platycercus elegans*), and with mammals, such as the sugar glider (*Petaurus breviceps*), for nest hollows.[65] Mynas are also minor pests of some fruit, such as grapes and figs. They can also nest in building cavities (especially chimneys) and bring irritating bird mites into buildings. In Hawaii, the Indian myna is a major disperser of seeds of the harmful introduced weed *Lantana camara*.[66] This weed is a serious threat to native communities in Australia, and it is likely that the Indian myna could play a similar role here.

3.2.17 European blackbird (Turdus merula)

Blackbirds damage grapes and stone fruit.[61] They can also spread weeds, such as blackberry (*Rubus* spp.) and sweet pittosporum (*Pittosporum undulatum*), and damage garden plants. Blackbirds are aggressive toward native birds, and they may compete with them for food and displace them, but there is no published evidence.

3.2.18 Mallard (Anas platyrhynchos)

Mallards interbreed with the native Pacific black duck (*Anas superciliosa*), and the hybrid offspring are fertile. Hence mallards are a conservation risk for this native duck and may eventually replace it.[63] The Pacific black duck is also an important game bird in Australia, and hunters prefer it to the mallard.

3.2.19 Nutmeg manikin (Lonchura punctulata)

Nutmeg manikins compete with native birds for food, and it is believed the species may be replacing native finches, such as the chestnut-breasted finch (*Lonchura castaneothorax*), in some areas.[63]

3.2.20 European carp (Cyprinus carpio)

Carp occur in huge numbers and reach up to 90% of total fish biomass in the waterways of the Murray–Darling Basin, Australia's most productive agricultural region. They increase costs to domestic and irrigation water suppliers, agriculture, recreational and commercial fisheries, and tourism.

Carp contribute to increased nutrient, algae, and suspended-sediment concentrations. This reduces water quality for stock, and increases pump wear and the cost of water treatment. The costs of this have not been estimated.

Carp have detrimental effects on aquatic plants and invertebrates, and they reduce water quality.[67] The role of carp in the decline of Australian native fish populations has been the subject of much speculation, but scientific evidence is lacking. There may be some competition between carp and native fish for both food and habitat, and carp may make aquatic habitats less suitable for other fish. Carp may have contributed to the decline of several threatened species, including dwarf galaxias (*Galaxiella pusilla*), trout cod (*Maccullochella macquariensis*), Yarra pygmy perch (*Edelia obscura*), and variegated pygmy perch (*Nannoperca variegata*). [68]

Recreational fishing in Australia is worth billions of dollars per year. Few anglers seek carp, and some may cease visiting areas where carp are abundant, which could have substantial negative impacts on industries supported by recreational fishing.[67] In Tasmania, the image of a high-quality trout fishery has been tainted by the introduction of carp. In an analysis of the effects of carp in the Gippsland Lakes in Victoria, a rough estimate of the costs to the community over 5 years was $175 million.[67] This included losses to the native commercial fishery and losses to recreational fishing, tourism, and commerce.

3.2.20.1 Carp control and research costs

Governments spend an estimated $1 million on carp control and about $500,000 on carp control research each year.

3.2.21 Brown trout (Salmo trutta) and rainbow trout (Oncorhynchus mykiss)

Brown trout and rainbow trout are aggressive and territorial, and they adversely affect many species of native fish through competition for food and habitat, predation, and habitat alteration. They are thought to have replaced native species in some habitats.[69,70]

Competition for food between brown trout and native species such as Macquarie perch (*Maquaria australasica*), river black fish (*Gadopsis marmoratus*), trout cod (*Maccullochella macquariensis*), and some Galaxiids (*Galaxias* spp.) have led to a severe decline in the numbers of these species.[69,71,72] Brown trout also prey on invertebrates such as yabbies, beetles, and tadpoles and can reduce their numbers as well.[71,72]

3.2.22 Mosquitofish (Gambusia holbrooki)

There is circumstantial evidence that mosquitofish harm native fish and frogs by competing for food and habitat, by aggressive behaviour, and by predation on eggs and

Table 3.2 Estimated Agricultural Damage Costs and Agricultural and Environmental Control and Research Costs for the Major Vertebrate Pests in Australia

Species	Agricultural losses ($million)	Control costs ($million)	Research costs ($million)
European rabbit	200	20	5
Red fox	40	7	4
Feral goat	20	2	1.5
Feral pig	100	5	1.5
House mouse	27	10	2.5
Feral dog and dingo	20	10	1.5
Feral cat	0	1	1
Non-indigenous birds	10	1.5	0.5
Cane toad	0	0	0.5
European carp	?	1	0.5
Totals	**$417 million**	**$57.5 million**	**$18.5 million**

hatchlings.[19,73,74] Declines in native fish populations have been observed in most places where mosquitofish have been introduced.[75]

3.2.23 Tilapia (Oreochromis mossambicus)

Tilapia prey on native fish species and compete with them for food and habitat.[76] They also remove plants, which may reduce habitat quality for native fish. Tilapia are thought to pose a major threat to native fish species in Australia, but the species is still in the early stages of establishing here, and its impacts have been little studied.[77]

3.3 Summary and discussion

Agricultural costs attributable to the major introduced vertebrate pests in Australia are difficult to estimate accurately owing to a shortage of reliable data, but they total at least $420 million per year for direct short-term losses (Table 3.2). Longer-term losses are also likely to be large.

Landholders and governments in Australia spend more than $60 million a year controlling introduced vertebrate pests. Beyond the resources spent on control, an additional cost is the value of lost opportunities to take profit from alternative investment of this expenditure.[12] There are also flow-on effects to related industries and the community, which are unquantified. Australian governments also spend about $20 million a year on research to control these pest species. Another cost to governments is the reduced tax revenue as a result of the reduced income of primary producers.

Two major impacts of introduced vertebrates in Australia, for which damage costs are not estimated in this chapter, are the contribution to long-term land degradation caused by introduced herbivores[20] and the contribution to declines and extinctions of small native mammals caused by introduced carnivores, particularly foxes and cats.[78] Grazing and browsing species, particularly rabbits, are responsible for preventing the regeneration of trees and shrubs that hold sandy soils together in Australia's dry interior.[26] The introduction of myxomatosis as a rabbit biological control disease in the 1950s and the later introduction of RHD in the 1990s allowed some trees and shrubs to regenerate. But field strains of myxomatosis have become less effective, and rabbits have developed genetic resistance to this virus; such effects will probably also occur with RHD. No biological control agents are available for foxes or cats. Conventional control methods, such as warren ripping for rabbits and poison baiting, trapping, and shooting for carnivores, are expensive. There are also animal welfare concerns associated with such control measures, and the measures often have harmful effects on non-target species.

There are likely to be significant changes in community and political attitudes toward the presence, impact, and management of non-indigenous animals in Australia over the next 20 years. For example, recent surveys in Australia have shown that some introduced species are already accepted by many people as a normal part of the landscape, despite the harm they cause. There is also increasing pressure to make pest control safer and more humane.

Community attitudes toward genetic modification and viruses will affect the ability of scientists to introduce potentially safer and more humane biological control techniques for pest animals, such as viral-vectored immunocontraception.[79] If developed, such techniques could enable more cost-effective population control over large areas. This is particularly important for a country such as Australia where large property sizes often make pest control using conventional techniques unfeasible. Expanding the range of pest control techniques available will also overcome the possibility of pest species becoming resistant to current control techniques. However, it is too early to determine whether biotechnology research will deliver such effective new control techniques.

Awareness of the harm done by introduced vertebrates also has consequences for managing the risk of new exotic species being introduced and becoming established.[2] Particular caution is required for species with attributes similar to those that have already become established as pests and for species that find a good climate match in Australia.[3]

Acknowledgments

We wish to thank members of the Vertebrate Pests Committee for information on pest animal research and control costs, and Glen Saunders and John Parkes for providing constructive comments on the draft manuscript.

References

1. Ebenhard, T., Introduced birds and mammals and their ecological effects, *Swedish Wildl. Res. Viltrevy*, 13, 1–107, 1988.
2. Bomford, M., *Importing and Keeping Exotic Vertebrates in Australia: Criteria for the Assessment of Risk*, Bureau of Rural Resources, Canberra, 1991.
3. Duncan, R.P., et al., High predictability in introduction outcomes and the geographical range size of introduced Australian birds: a role for climate, *J. Anim. Ecol.*, 70, 621–632, 2001.
4. Newsome, A.E. and Noble, I.R., Ecological and physiological characters of invading species, in *Ecology of Biological Invasions: An Australian Perspective, Groves*, R.H. and Burdon, J.J., Eds., Australian Academy of Science, Canberra, 1986, 1–20.
5. Rolls, E.C., *They all Ran Wild*, Angus and Robertson, Sydney, 1969.
6. Wilson, G., et al., *Pest Animals in Australia: A Survey of Introduced Wild Mammals*, Bureau of Rural Resources and Kangaroo Press, Canberra, 1992.
7. Olsen, P., *Australia's Pest Animals: New Solutions to Old Problems*, Bureau of Resource Sciences and Kangaroo Press, Canberra, 1998.
8. Environment Australia, *Threat Abatement Plan for Competition and Land Degradation by Feral Rabbits*, Biodiversity Group, Environment Australia, Canberra, 1999.
9. Environment Australia, *Threat Abatement Plan for Predation by the European Red Fox*, Biodiversity Group, Environment Australia, Canberra, 1999.
10. Environment Australia, *Threat Abatement Plan for Competition and Land Degradation by Feral Goats*, Biodiversity Group, Environment Australia, Canberra, 1999.
11. Clarke, G., et al., *Environmental Pests in Australia*, Unpublished report, CSIRO, Canberra, 2001. (Available from Geoff.Clarke@ento.csiro.au).

12. Choquenot, D., McIlroy, J., and Korn, T., *Managing Vertebrate Pests: Feral Pigs*, Australian Government Publishing Service, Canberra, 1996.
13. Ramsay, B.J., *Commercial Use of Wild Animals in Australia*, Bureau of Resource Sciences, Australian Government Publishing Service, Canberra, 1994.
14. Parkes, J., Henzell, R., and Pickles, G., *Managing Vertebrate Pests: Feral Goats*, Bureau of Resource Sciences and the Australian Nature Conservation Agency, Canberra, 1996.
15. Wilson, G., McNee, A., and Platts, A., *Wild Animal Resources — Their Use by Aboriginal Communities*, Bureau of Rural Resources, Australian Government Publishing Service, Canberra, 1992.
16. Long, J.L., *Introduced Birds of the World*, Reed Books, Sydney, 1981.
17. Strahan, R., *The Mammals of Australia*, Reed Books, Sydney, 1995.
18. Cogger, H.G., *Reptiles and Amphibians of Australia*, Collins, Sydney, 1994.
19. Arthington, A.H. and McKenzie, F., *Review of Impacts of Displaced–Introduced Fauna Associated with Inland Waters*, Centre for Catchment Management and In-stream Research, Griffith University, 1997.
20. Williams, K., et al., *Managing Vertebrate Pests: Rabbits*, Australian Government Publishing Service, Canberra, 1995.
21. Short, J., The functional response of kangaroos, sheep and rabbits in an arid grazing system, *J. Appl. Ecol*., 22, 435–47, 1985.
22. Newsome, A.E., Ecological interactions, in *Australian Rabbit Control Conference Proceedings*, Cooke, B.D., Ed., Anti-rabbit Research Foundation of Australia, Adelaide, 1993, 23–25.
23. Neave, H., *Overview of Effects on Australian Wild Rabbit Populations and Implications for Agriculture and Biodiversity*, Rabbit Calicivirus Disease Program Report 1, Bureau of Rural Sciences, Canberra, 1999.
24. Sandell, P. and Start, T., *Implications for Biodiversity in Australia*, Rabbit Calicivirus Disease Program Report 4, Bureau of Rural Sciences, Canberra, 1999.
25. Henzell, R., Rabbits, feral goats, mulga and rangeland stability, *Aust. Vert. Pest Cont. Conf*. 9, 18–21, 1991.
26. Lange, R.T. and Graham, C.R., Rabbits and the failure of regeneration in Australian arid zone Acacia, *Aust. J. Ecol*. 8, 377–81, 1983.
27. Brothers, N.P., Skira, I.J., and Copson, G.R., Biology of the feral cat *Felis catus* (L.) on Macquarie Island, *Aust. Wildl. Res*. 12, 425–36, 1985.
28. Henzell, R., *Proclaimed Animal Research in South Australia – Cost-benefits, Future Directions and Related Issues*, Animal and Plant Control Commission, Adelaide, 1989.
29. Sloane, Cook and King Pty Ltd, *The Economic Impact of Pasture Weeds, Pests and Disease on the Australian Wool Industry*, report prepared by Sloane Cook and King Pty Ltd for the Australian Wool Corp., 1988.
30. Acil Economics and Policy Pty Ltd, *The Economic Importance of Rabbits on Agricultural Production in Australia*, report prepared for the International Wool Secretariat, 1996.
31. Manson, A., Identification of public and private benefits from agricultural research, development and extension: some preliminary results from a study of the Australia New Zealand Rabbit Calicivirus Program, paper presented at the 42nd Annual Conference of the Australian Agricultural and Resource Economics Society, Armidale, New South Wales, 1998.
32. Saunders, G., et al., *Managing Vertebrate Pests: Foxes*, Australian Government Publishing Service, Canberra, 1995.
33. Pinnington, G., Animal Control Technologies Pty Ltd Newsletter, Melbourne, 1999.
34. Lugton, I., Fox predation on lambs, in *Proceedings Australian Sheep Veterinary Society AVA Conference, Gold Coast*, Hucker, D.A., Ed., 1993, 17–26.
35. Kinnear, J. E., Onus, M.L., and Sumner, N.R., Fox control and rock-wallaby population dynamics II, An update, *Wildl. Res*. 25, 81–88, 1998.
36. Short, J., et al., Reintroduction of macropods (Marsupialia: Macropodoidea) in Australia – A review, *Biol. Cons*. 62, 189–204, 1992.
37. Thompson, J., et al., *Feral Goat Management in South-west Queensland, Department of Natural Resources, Queensland*, Coorparoo, 1999.

38. Plant, J.W., et al., Neonatal lamb losses due to feral pig predation, *Aust. Vet. J.* 54, 426–429, 1978.
39. Pavlov, P.M., Kilgour, R.J., and Pederson, H., Predation by feral pigs on merino lambs at Nyngan, New South Wales, *Aust. J. Exp. Agric. Anim. Husb*. 21, 570–574, 1981.
40. Tisdell, C.A., *Wild Pigs: Environmental Pest or Economic Resource*? Pergamon Press, Australia Pty Ltd, Sydney, 1982.
41. Pech, R.P. and McIlroy, J.C., A model of the velocity of advance of foot-and-mouth disease in feral pigs, *J. Appl. Ecol.* 27, 635–650, 1990.
42. Caughley, J., et al., *Managing Vertebrate Pests: Rodents*, Bureau of Resource Sciences, Canberra, 1998.
43. Croft, D. and Caughley, J., A survey of the MIA mouse plague — at what cost? *Farmers' Newsletter* 145, 40–41, 1995.
44. Redhead, T. and Singleton, G., An examination of the PICA strategy for the prevention of losses caused by plagues of house mice *Mus domesticus* in rural Australia, in *Vertebrate Pest Management in Australia: A Decision Analysis/Systems Analysis Approach*, Norton, G.A. and Pech, R.P., Eds., Project Report No. 5, CSIRO Division of Wildlife and Ecology, Canberra, 1988, 18–37.
45. Fleming, P., et al., *Managing the Impacts of Dingoes and Other Wild Dogs*, Bureau of Rural Sciences, Canberra, 2001.
46. Fleming, P.J.S. and Korn, T.J., Predation of livestock by wild dogs in eastern New South Wales, *Aust. Rangl. J*. 11, 61–66, 1989.
47. Allen, L. and Gonzalez, T., Baiting reduces dingo numbers, changes age structures yet often increases calf losses, *Aust. Vert. Pest Cont. Conf*. 11, 421–428, 1998.
48. Baird, R.F., The dingo as a possible factor in the disappearance of *Gallinula mortierii* from the Australian mainland, *Emu* 91, 121–122, 1991.
49. Risbey, D.A., et al., The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment, *Wildl. Res*. 27, 223–235, 2000.
50. Dickman, C.R., *Overview of the Impacts of Feral Cats on Australian Native Fauna*, Australian Nature Conservation Agency, Canberra, 1996.
51. Moodie, E., *The Potential for Biological Control of Feral Cats in Australia*, Australian Nature Conservation Agency, Canberra, 1995.
52. Hone, J., et al., Impact of wild mammals and birds on agriculture in New South Wales, *J. Aust. Inst. Agric. Sci*. 47, 191–199, 1981.
53. Dobbie, W.R., Berman, D.M., and Braysher, M.L., *Managing Vertebrate Pests: Feral Horses*, Australian Government Publishing Service, Canberra, 1993.
54. Wilson, G.R. and O'Brien, P.H., Wildlife and exotic animal disease emergencies in Australia: planning an effective response to an outbreak, *Disaster Manage*. 1, 30–35, 1989.
55. Stevenson, P.M., Rat control: Norfolk Island style, *Aust. Rang. Bull*. 38/39, 47–48, 1997.
56. Freeland, W.J. and Martin, K.C., The rate of range expansion by *Bufo marinus* in northern Australia, 1980–84, *Aust. Wildl. Res*. 12, 555–559, 1985.
57. Crossland, M.R. and Alford, R.A., Evaluation of the toxicity of eggs, hatchlings and tadpoles of the introduced toad *Bufo marinus* (Anura: Bufonidae) to native Australian aquatic predators, *Aust. J. Ecol*. 23, 129–137, 1998.
58. Crossland, M.R., A comparison of cane toad and native tadpoles as predators of native anuran eggs, hatchlings and larvae, *Wildl. Res*. 25, 373–381, 1998.
59. Burnett, S., Colonising cane toads cause population declines in native predators: reliable anecdotal information and management implications, *Pac. Cons*. Biol. 3, 65–72, 1997.
60. Agriculture Western Australia, 1998, http://www.agric.wa.gov.au/agency/Pubns/infonote/infonotes/starling.html
61. Bomford, M., *Bird Pest Impact and Research in Australia: A Survey and Bibliography*, Bureau or Rural Resources Working Paper 3/92, 1992.
62. Weber, W.J., *Health Hazards from Pigeons, Starlings and English Sparrows*, Thompson Publications, Fresno, CA, 1979.
63. Frith, H.J., *Wildlife Conservation*, Angus and Robertson, Sydney, 1979.

64. Blakers, M., Davies, S.J.J.F., and Reilly, P., *The Atlas of Australian Birds*, Melbourne University Press, Melbourne, 1984.
65. Pell, A.S. and Tidemann, C.R., The impacts of two exotic hollow-nesting birds on two native parrots in savanna and woodland in eastern Australia, *Biol. Cons.* 79, 145–153, 1996.
66. Pimentel, D., et al., Environmental and economic costs of nonindigenous species in the United States, *BioScience* 50, 53–65, 2000.
67. Koehn, J.D., Brumley, A.R., and Gehrke, P.C., *Managing the Impacts of Carp*, Bureau of Rural Sciences, Canberra, 2000.
68. Wager, R. and Jackson, P., *The Action Plan for Australian Freshwater Fishes*, Australian Nature Conservation Agency, Canberra, 1993.
69. Arthington, A.H., Ecological and genetic impacts of introduced and translocated freshwater fishes in Australia, *Can. J. Fish. Aquat. Sci*. 48, 33–43, 1991.
70. Crowl, T.A., Townsend, C.R., and McIntosh, A.R., The impact of introduced brown and rainbow trout on native fish: the case of Australasia, *Rev. Fish Biol. Fish.* 2, 217–241, 1992.
71. Fletcher, A.R., Effects of introduced fish in Australia, in *Limnology in Australia*, De Dekker, P. and Williams, W.D., Eds., CSIRO, Melbourne, 1986.
72. Cadwallader, P.L., *Overview of the Impacts of Introduced Salmonids on Australian Native Fauna*, Australian Nature Conservation Agency, Canberra, 1996.
73. Lloyd, L.N., Ecological interactions of Gambusia holbrooki with Australian native fishes, in *Proceedings of the Australian Society for Fish Biology's Workshop on Introduced and Translocated Fishes and their Ecological Effects*, Australian Government Publishing Service, Canberra, 1990, 94–97.
74. Komak, S. and Crossland, M.R., An assessment of the introduced mosquitofish (*Gambusia affinis holbrooki*) as a predator of eggs, hatchlings and tadpoles of native and non-native aneurans, *Wildl. Res*. 27, 185–189, 2000.
75. McKay, R.J., Introductions of exotic fishes in Australia, in *Distribution, Biology and Management of Exotic Fishes*, Courtenay, W.R. and Stauffer, J. R., Eds., John Hopkins University Press, Baltimore, 1984, 177–199.
76. Arthington, A.H., Introduced Cichlid fish in Australian inland waters, in *Limnology in Australia,* De Dekker, P. and Williams, W.D., Eds., CSIRO, Melbourne, 1986, 239–348.
77. Arthington, A.H. and Cadwallader, P.L., Cichlids, in *Freshwater Fishes of South-Eastern Australia*, McDowall, R.M., Ed., Reed Books, Chatswood, NSW, 1996, 176–180.
78. Recher, H.F. and Lim, L., A review of current ideas of the extinction, conservation and management of Australia's terrestrial vertebrate fauna, *Proc. Ecol. Soc. Aust*. 16, 287–301, 1990.
79. Robinson A.J., et al., Biocontrol of pest mammals in Australia: progress towards virally vectored immunocontraception for the rabbit and mouse, *Recent Adv. Microbiol.* 7, 63–98, 1999.

chapter four

Environmental and economic costs of invertebrate invasions in Australia

Deon Canyon, Rick Speare, Ian Naumann, and Ken Winkel

Contents

4.1 Introduction

An enormous number of exotic invertebrate pests have made their way to Australia from other parts of the world, often precipitating environmental and economic consequences that can reach devastating levels. Several published studies have addressed exotic invasions in Australia, including a recent contribution by New.[54] However, few of these studies

0-8493-0836-4/02/$0.00+$1.50

have presented a comprehensive picture of the related costs involved, and medical pests are not usually included.

Along with the human colonization of Australia beginning in the late 1780s, a number of exotic invertebrate organisms, brought by ship from various ports around the world, also colonized this island continent. Outbreaks of disease caused by the importation of mosquitoes and other insects, such as lice, proliferated throughout tropical areas, especially in prospecting townships, where squalid conditions often prevailed. Food stores and timbers contaminated with exotic insects, in addition to indigenous insect populations, would prove to have a significant impact on future agricultural and forestry operations. But this is nothing compared to today's busy and far-reaching global traffic, in which exotic insects are easily transported and via which many organisms have become well established in foreign countries.

The traditional barriers of Australia, the sea and the Great Dividing Range, have created distinct climatic zones that have long served to limit the spread of pest populations. However, for as long as humans have created habitats for themselves or their crops, exotic insect pests have found a way to exploit the situation. For decades the pattern has been set in which civilization and advances in basic hygiene have played the leading role in ridding countries of imported vector-borne disease. The question remains as to whether this will continue to provide protection in the face of increasing global tourism and traffic. There are already indications that increased population movement, the freer movement of products and animals as a result of world trade agreements, and the decreased time taken to move between countries are changing the global distribution of insect vectors and their related diseases.

This chapter will examine exotic invertebrates in four main areas: medical, veterinary, agricultural, and marine. Each section will focus on several important species and will attempt to outline current situations. Economic costs relating to the introduction of these species have been related where possible from cost analyses, but in some cases a best estimate is presented. Environmental costs are difficult to determine for most pests, because the true dimensions of their impact are often unknown. How, for instance, would one estimate the damage done to the environment by the practice of spraying insecticide over a mangrove mosquito breeding site, apart from simply monitoring local animal populations? While the local effects may be more easily obtained, the broader effects are confounded by too many factors to enable a reasonable level of certainty in any conclusion. Where possible, environmental costs are stated, but these costs are not always of a monetary nature. It is very difficult to compare medical costs with other costs because of many inestimable elements. Medical costs register far below agricultural costs, but the intangible factors, such as those relating to suffering and psychological effects, may translate to a lifetime of lost production or social damage.

The estimated costs in some areas have been divided by the number of years they pertain to so that an annual amount could be generated. Pest control costs have been used for agricultural items, since that figure is more comparable to the figures generated in the medical section.

From the information collated in this chapter, a conservative estimate of the annual cost of exotic invertebrates in Australia would be in the range of $1 billion (Australian), while an estimate including production loss and other intangibles would be $5 billion to $8 billion annually. No figure is presented for potential losses that several recently introduced invertebrates may cause, although their impact is expected to be considerable in the years to come.

4.2 Invasions of medical importance

4.2.1 Aedes aegypti

Dengue fever, dengue hemorrhagic fever (DHF), and dengue shock syndrome are various forms in which the dengue virus manifests itself in humans. The peridomestic mosquitoes *Aedes aegypti* and *Ae. albopictus* are responsible for biological transmission of the four serotypes. Cross-protective immunity lasts for about 2 months,[40] and immunity to a particular serotype is lifelong.[35] Dengue viruses are particularly effective organisms, since they are able to replicate to a high level in mosquitoes and produce a high viremia in humans, which in turn facilitates the infection of other mosquitoes. Globally, *Ae. aegypti* is responsible for most urban infections, while *Ae. albopictus* is responsible for the rural infection cycle. In Australia, only *Ae. aegypti* is present, and dengue has never been endemic. The importation of *Ae. albopictus* has been detected and prevented on several occasions. Known as the Asian tiger mosquito, this species has achieved global distribution since its introduction into the United States in 1980. Preventing the establishment of this species in Australia is a major focus of vector control strategy in the north. Two other potential vector species are present, *Ae. scutellaris* and *Ae. katherinensis*. However, at this point these species do not appear to play an important role in dengue transmission.

Dengue is advancing on a geographical basis, prompting the World Health Organization to place dengue on the agenda of its infectious diseases arm, the Committee for Tropical Disease Research. The WHO estimates that every year, 100 million cases of dengue fever and 500,000 cases of DHF occur, with an average case fatality rate of 5%. Thus, 25,000 to 30,000 fatalities are caused by dengue hemorrhagic fever each year. In Puerto Rico, the DALYs (disability adjusted life years) lost per million people increased by 25% from 1984 to 1994, placing the economic impact of the disease in the same order of magnitude as malaria, tuberculosis, hepatitis, STDs (excluding AIDS), the childhood cluster (polio, measles, pertussis, etc.), and the tropic cluster (Chagas disease, schistosomiasis, and filariasis).[51]

4.2.1.1 History of epidemics

Dengue has manifested itself in epidemic form in Australia ever since 1879. A general infection rate of 75% has been proposed for all areas experiencing dengue up until the 1953–55 epidemic. Since then, infection rates have ranged from 2 to 38%, depending on geographical area.[45] The relationship between death and percentage of population infected varies substantially. Hayes and Gubler[40] suggested that one to seven DHF cases would typically result from every 100 dengue fever cases, and that prior to the development of modern hospital management, 50% of all DHF patients would die.

The earliest known dengue epidemics occurred from 1897 to 1901 and spread throughout most of Thursday Island, Townsville, Cairns, Cooktown, Pt. Douglas, Charters Towers, Normanton, Mackay, Ingham, and Bowen to Brisbane, with cases inland at Hughenden, Barcaldine, and other locations. This widespread epidemic penetrated into New South Wales by 1898. Cases continued to be reported, including at least three deaths in Charters Towers[15] and three deaths in Brisbane from 1899–1901.[27] The population of Queensland was around 500,000 in 1900,[12] so based on an infection rate of 75%, it is possible that 375,000 people were infected with dengue. Cases continued to a lesser degree until 1904–06, when the virus traveled north to infect the entire population of Thursday Island, and south to cause an extensive epidemic in Brisbane, where 94 deaths occurred. One death was also reported in Sydney.

Using the previous logic, an estimated 190 DHF cases probably occurred, with a maximum of 19,000 cases of dengue fever. If only 15% actually reported to a health clinic for examination,[45] a probable 126,730 people were infected in and around the Brisbane region. The population of Brisbane at the time was about 126,000.[15] Thus the rate of 1 death to 1000 possible cases seems likely. Over the period from 1885 to 1923, 52 deaths were recorded in the Townsville region,[49a] which arose from some 52,000 probable infections. From 1916 to 1919 and in 1924, New South Wales and Queensland were broadly struck with a similar infection rate, and the number of infected people has been estimated at 600,000[15,49a,71] in each of the two epidemics.

From 1938–39, dengue made another appearance, which led to the 1941–43 epidemic. At that time, 486 cases were reported in Townsville, equivalent to an estimated 610 infected individuals. During 1941–43 a little-known epidemic swept Queensland down to Brisbane, with up to 85% infection rates in some towns. In Townsville alone, 5000 cases were reported, with 25,000 probable infections. Judging from past performance, and taking other areas into account, it is estimated that this figure could at least be doubled. This epidemic also swept north to Darwin and initiated the highly successful campaign to eliminate *A. aegypti* from the Northern Territory. Dengue struck again in 1953–55, this time infecting 75% of the population and producing an estimated 15,000 cases.[25] In 1981–83, dengue returned to Queensland and was confirmed in 458 people. Using the notification rate of 15% found by Kay et al.,[45] it is possible that 3100 people were infected in this epidemic. From 1991 to 2000, 2294 confirmed cases have been reported, translating to a probable 15,500 infections.

4.2.1.2 Cost estimation

Cumulatively, this leads to an estimated figure of 1,837,940 dengue infections in Australia since the introduction of *Aedes aegypti* and dengue, which is certain to be conservative due to a dearth of information on numerous places that experienced epidemics. Based on this estimate, 1,819,340 people were infected prior to the 1980s, with a 75% infection rate, and 18,600 infections have resulted since the 1980s, with an infection rate of 15%. This gives a total possible population within the affected areas of 2.5 million people.

Gubler and others[63] estimated a cost of $80 (Australian) per capita for the 1977 Puerto Rico epidemic, a figure that included medical costs, control efforts, lost work, and lost tourism revenue. If Gubler's figure is used to calculate the cost of all Australian dengue epidemics, the result is an all-inclusive estimated total cost of $147 million, or an average of $1.3 million per year.

The costs appear to have been higher in Australia, and this may be related to the population structure in North Queensland. McBride et al.[49b] calculated the average time lost through illness in the 1992–93 Charters Towers epidemic to be 10.5 days. Using the total number of infected people, this results in 19,298,370 workdays being lost in total, or 175,440 days per year. With each workday valued at $96, based on an average income of $35,000, the annual cost to Australia since the introduction of dengue comes to almost $17 million in current Australian dollars.

However, since epidemics are much smaller these days, the current situation must be viewed in different terms. The estimated cost of work lost prior to 1990 in today's dollars is close to $2 billion. In the past 10 years, however, 15,500 infections have led to an estimated 162,750 lost days, worth a total of $15.6 million, or $1.56 million per year.

Local city councils have indicated that their labor costs for the control of exotic mosquitoes and related diseases range from $2000 to $6000 per year, with brief major jumps occurring during epidemics. The cost of insecticides for exotic mosquito control is minimal during non-epidemic years, but ranges from a few thousand to nearly a million dollars per council per epidemic. The Charters Towers City Council determined that the cost of vector control, including insecticides and staff, for its 1992–93 epidemic was $750,000; with a population of 8500 at that time, the cost therefore was $88 per capita.

In a similarly sized epidemic, the Townsville City Council estimated direct costs to be at least $500,000 for a population of 110,000, resulting in $5 per capita. Thus, it is problematic to use the per capita method in the modern environment, where epidemics cause similar numbers of infections with similar costs regardless of the population size. The population in North Queensland is comparatively widely distributed and small, and dengue-related expenses can be considerable, even though large populations are not involved. The epidemic costs in Townsville and Cairns, including annual maintenance costs, averaged at least $200,000 per year during the past decade. The Tropical Public Health Unit in Cairns recently dealt with a number of small epidemics lasting over a period of 3 years, from 1997 to 1999. The unit formed a Cairns-based vector control team called the Dengue Action Response Team (DART) and have estimated an annual cost of $200,000, equating to $2 per capita, since the formation of this team. Over the past decade, approximately 15,500 infections have occurred within an area containing a human population of not more than 300,000, at a control cost of around $400,000 per year.

If all epidemics are taken into account, the cost of the introduction of *Ae. aegypti* to Australia, including lost work and control costs, has been considerable at around $17 million per year. Since 1990, however, the costs have been more reasonable, averaging around $2 million per year. These figures do not include intangible costs to individuals and society, those involving quality of life and general well-being. Intangibles are similar in nature to environmental costs, where quality is also difficult to measure except in great leaps and bounds.

Data submitted by representatives of pesticide companies and city councils in North Queensland suggest that the control costs relating to non-exotic mosquitoes far exceed the costs of controlling exotic mosquitoes.

It is unlikely that dengue will become endemic in Australia, owing to a lack of potential reservoir hosts and the sparse population outside of urban centers. There is no evidence to suggest that dengue was ever endemic, with early records indicating incoming ships as the primary source.[71] Nevertheless, since 1990, a significant number of dengue cases have occurred every year, indicating that frequent viral importation and high vector numbers may result in an endemic-like situation in which it will be difficult to distinguish between epidemics. Since the Cairns airport was made an international airport, the number of travelers from dengue-endemic countries has increased significantly. The recently rewritten management plan for dengue aims to lower dengue incidence by reducing vector breeding through education programs, encouraging greater awareness of the disease among the medical community, and improving surveillance, including the use of serological testing. This new strategy and the formation of DART have been essential instruments in dealing with the increasing volume of viremic importations.

4.2.2 Culex gelidus

In 1995, an outbreak of Japanese encephalitis (JE) occurred in the Torres Strait Islands in Northern Australia. Japanese encephalitis is a serious disease, causing an average hospital stay of 14 days and a mortality rate of 10 to 50%. Forty percent of survivors experience mental or physical crippling and require 1 to 5 years of rehabilitation, while 10% require chronic care.[63]

During a 3-week period, three residents of the outer island Badu (population 700) manifested typical symptoms of acute illness, with headaches, fever, convulsions, depressed level of consciousness, and coma; two patients died.[56] A seroprevalence survey confirmed JE infection in 35 Badu people (16%), 20 other outer island people (1.5–11%), and 63 pigs (70%).[36] There is a vaccine available that is 95% protective. In this epidemic, the majority of inhabitants of the northern Torres Strait (3500 people) were vaccinated by

Queensland Health in the same year. Sentinel pigs were established in 1996, and almost all had seroconverted by March of that year, as well as most horses that were tested. Among humans, in early 1998 an adult male, working on a boat at the mouth of the Mitchell River on the west coast of the Cape York Peninsula, and a 12-year-old unvaccinated child from Badu were diagnosed as having JE. This is the first time that the disease has been recorded on mainland Australia, and there is now some concern arising from several seroconversions of wild pigs on the mainland.[57]

The Queensland Health Tropical Public Health Unit in Cairns, the most active responsible state health authority, declined to provide an estimate on the cost of JE to Australia, citing the difficulty of obtaining data from the various agencies involved, as well as confidentiality issues. It is likely, however, that several million dollars are involved in such efforts as serological surveillance, a comprehensive vaccination program, and the building of a new piggery to act as a permanent sentinel station.

Viral isolations had suggested that the mosquito responsible for these outbreaks was *Culex annulirostris*, the vector of Murray Valley encephalitis and a common native swamp breeder.[36,37] However, a previously misidentified alien mosquito species, *Culex gelidus*, widely distributed in the Torres Strait, mainland Queensland, and the Northern Territory, has now been found, and JE has been isolated in it.[17] This exotic mosquito is capable of transmitting not only JE but also Batai, Getah, and Tembusu viruses, and it thus seems more likely that JE will become more prevalent on the mainland. Northern Territory medical entomologist Peter Whelan said that this mosquito could become a threat, because it breeds around piggeries, dairies, sewage treatment works, and slaughterhouses. "From our latest work," Whelan said, "we can now say that it's too late to eradicate this mosquito."[16] The potential now exists for much larger, more widespread epidemics and the associated higher costs.

4.2.3 Honeybees and wasps

Relatively little information is available on the economic costs of exotic venomous invertebrates in Australia. A review of the literature reveals that the greatest calculable economic impact is attributable to bees and wasps. Indeed, no information is available, for example, on the impact of exotic arachnids. Note that the attribution of specific health costs to bee stings, as distinct from wasp stings, is complicated by the failure of the current health classification system to resolve the two diagnoses. This is further discussed below.

Since its arrival in 1822, the European honeybee (*Apis mellifera*) has become widespread throughout all the states and territories of Australia. By 1998 more than 670,000 hives were officially registered.[33] Apart from the considerable income generated by honeybees, their stings are a leading cause of death from venomous bites and stings in Australia. For example, 25 bee-sting-related fatalities had been registered by the Australian Bureau of Statistics during the preceding 22 years in 1981.[39] A more recent analysis identified at least 43 fatalities attributed to both bees and wasps in Australia during the 19-year period to December 1997, second only to snakebite fatalities.[48] Cases directly attributable to honeybee stings[50] showed a mortality rate of 0.12 per million population per year.

Similarly, bee stings are a leading cause of deaths reported by emergency rooms and hospitals. Under the current diagnostic system, bee and wasp stings are coded as a single category. National hospitalization data for the year July 1996 through June 1997 listed bee and wasp stings as the cause of 977 new in-patient cases.[48] This hospitalization rate was second only to spider bites over the same period. Similarly, an analysis of emergency room data in Victoria revealed that bee and wasp stings accounted for 41% of all cases involving venomous bites and stings.[74]

From data available for emergency rooms in the United States, and assuming corresponding costs per case,[48] it can be estimated that in Australia, bee and wasp stings are responsible for at least $10 million each year in direct hospital expenditures. Although there are no data currently available on the extent of less severe morbidity in humans or on the impact of bee stings on domestic animals and livestock, it appears that the effect of honeybees on native plants and animals is minor.[33] Clearly, then, the economic impact of honeybees in Australia is overwhelmingly positive.

In contrast to the net positive value of the honeybee, exotic wasps — notably the European wasp, *Vespula germanica* — inflict damage without offering any benefits. A native of Europe, western Asia, and northern Africa, the European wasp was first introduced to Australia in 1954 but only became established in 1959 in Hobart, Tasmania.[18,64] This vespid arrived on the mainland in 1977, and lacking any natural predators, has rapidly expanded its range ever since. By 1991, an estimated tens of thousands of nests were being destroyed in metropolitan Melbourne annually,[18] with wasp densities of up to 40 per km^2 being reported. These wasps are now found in Tasmania, Victoria, New South Wales, the Australian Capital Territory, and South Australia.[64]

Indeed, the surge in numbers of *V. germanica* in southeastern Australia during the summer of 1997–98 prompted the Victorian government to call for a national control strategy.[49] However a recent analysis of wasp sting mortality in Australia, driven by concern about the lethal potential of *V. germanica*, failed to detect any human fatalities attributable to this wasp during the past 20 years.[51] Research into the morbidity attributable to these wasps has been limited by the disease classification system that combines bee and wasp stings in a single category. An attempt to calculate the economic and health impact of this wasp conservatively estimated the cost to Victoria alone at greater than $2 million annually.[43] This included the effects on horticultural industries, health care, national parks, and tourism, as well as the direct costs of nest destruction.

4.2.4 Red imported fire ants

Two species of potentially medically and ecologically significant exotic ants have been found in Australia. The tropical fire ant, *Solenopsis geminata*, is estimated to have been introduced sometime before 1987 into the Northern Territory, and its current distribution is limited to northern coastal areas.[2] While the species does not appear to have caused significant ecological damage in Australia, it has become a serious problem elsewhere in Southeast Asia and the Pacific, especially in Okinawa and Guam.[42] More recently, the South American fire ant, commonly known as the red imported fire ant (RIFA), *S. invicta*, has been identified in southern Queensland.[58]

By late May 2001, 4000 hectares in Brisbane's southwest region and a smaller area on nearby Fisherman Island had become infested. The mode and timing of these invasions remains unclear. Within these areas, the scattered infestations total only 300–400 ha, so eradication remains theoretically possible. There is some suggestion that native ants may provide a degree of "biotic resistance," but it is difficult to speculate what will happen in Australia. It is probable that a mosaic distribution will result, with the RIFA dominant in more open areas and fewer or no RIFAs in more heavily shaded habitats. The RIFA is not expected to do well in the alpine regions or in the dry interior.

Unfortunately, the United States' experience with the two imported species *S. invicta* and *S. richteri* does not give cause for optimism in Australia. These species have developed resistance to natural and chemical control methods, and they have continued to cause significant ecological and agricultural damage, as well as a variety of health problems, in southeastern states. The health risks range from sting-site pustules and secondary infections to severe late-phase responses and even life-threatening anaphylaxis.[30,67] Sometimes skin grafts or the amputation of an affected limb are necessary.[70] Stings may occur indoors as well as outdoors. In areas where RIFAs are endemic, the most commonly reported cause of hymenoptera venom allergy is now RIFA allergy.[31] It has been estimated that RIFAs sting more than 50% of the people living in endemic areas each year.[24]

Fire ants have a significant impact on agriculture as well: they may damage or remove seeds; damage roots, tubers, stems, and fruit; protect injurious plant-sucking hemiptera; interfere with biological control; present a hazard to hand laborers; damage irrigation systems; build mounds that interfere with mechanical harvesters; and harass livestock, especially young animals. There are also a host of additional effects, such as damage to electrical equipment and structural damage due to undermining. Estimates of the monetary impact of fire ants on agriculture vary enormously. A recent, and perhaps conservative, analysis estimates the combined value of production losses and control costs in North America to be $246 million (U.S.).[62] Less substantiated estimates place the total annual cost at more than $1 billion.

In South Carolina, where all 46 of the state's counties are now infested, fire ants are thought to be responsible for $2.4 million in direct health costs each year. These costs include an estimated 660,000 sting cases and some 33,000 medical consultations.[11]

Regional RIFA control programs were discontinued in the United States because of cost and environmental concerns, and these ants now infest more than 310 million acres in the United States and Puerto Rico. Evolutionary adaptations have facilitated their expansion northward and westward, heightening public health concerns.[46]

If not eradicated or contained, RIFA likely will establish in coastal and adjacent regions of most Australian states.[20] The Queensland government has invested several hundred thousand dollars in initial measures, particularly in the creation of a Fire Ant Control Center employing a 30-person operational control group staff in one of the RIFA outbreak epicenters. A Phase One study with a budget of $750,000 is nearing completion.

To determine whether control and eradication procedures derived from the North American experience are likely to be successful, fieldwork so far has addressed a few key questions, such as the optimum treatment for nests and the nature of native ant–fire ant competition. Desk-based research has included a CLIMEX analysis to predict likely distribution in Australia and a cost-benefit analysis of eradication efforts led by ABARE (Australian Bureau of Agricultural and Resource Economics). The cost-benefit analysis will evaluate three scenarios: do nothing, eradication, and ongoing control. There will also be some analysis of the DNA of the various Queensland RIFA populations to determine whether Queensland is dealing with a single, multiple, or ongoing introductions. This work may indicate the point of origin, which is suspected to be the United States, although South America is also a possibility. Preliminary observations suggest that both polygyne and the monogyne forms are present, but that too must be confirmed by more detailed fieldwork and molecular studies. The two forms differ behaviorally, and polygyne forms typically form massive populations.

At present it is hoped that eradication will be possible; however, careful evaluation of the effectiveness of treatment, biological models describing the growth and spread of the ant in Australia, and the cost-benefit analysis will ultimately determine the course of action. An eradication campaign based on existing techniques could easily result in Australia spending $35 million to $40 million on this creature over the next 3 to 5 years.

4.3 Invasions of veterinary importance

In the last half of the 20th century, Australia has had the most stringent importation requirements for vertebrate animals of any country in the world. The requirements have been particularly stringent for livestock, domestic pets, and wildlife, and less so for fish and amphibians. One aspect of these requirements has been that any imported animal must be free of ectoparasites. The policy has been highly successful; no ectoparasites of significance to livestock or domestic pets have become established in Australia in the past 50 years.

4.3.1. Cattle tick

Early introductions of arthropod pests into Australia proved to be enormously costly. The cattle tick, *Boophilus microplus*, has been the most expensive. The cattle tick was introduced to Australia in 1872 by the importation of 12 Brahman cattle from Batavia.[9] It first appeared in Queensland in 1891, Western Australia in 1895, and New South Wales in 1906. Cattle ticks were introduced to Victoria in 1914 via horses from Queensland that were en route to the war in Egypt. However, *B. microplus* did not become established in Victoria, and this state, as well as Tasmania and South Australia, have remained free of the cattle tick.[9] The distribution of the tick is determined by low temperature and humidity, and for that reason it is confined in Australia to northern Western Australia, the northern half of the Northern Territory, coastal Queensland, and northern New South Wales.

Two blood protozoan parasites, *Babesia bovis* and *B. argentina*, and a blood-borne bacterium, *Anaplasma marginale*, use the tick as a vector and cause economic impact. The economic costs of cattle tick include:

- Direct effects of the tick on cattle: loss of condition, anemia and deaths, susceptibility to drought, damage to hides, slow growth rate
- Effects of dipping on cattle: loss of body weight, loss of milk production, deaths during drought, loss of young calves, toxicity
- Control costs: increased stock handling, costs of acaricides
- Market effects: restrictions on cattle movement
- Costs of tick-borne diseases: deaths, slow growth, vaccine costs, treatment costs, handling costs

Davis[22] presented cost estimates, in 1997 Australian dollars, of $87 million for 1959, $87 million for 1973, and $134 million for 1995. The earlier estimates did not take into account government costs associated with control strategies and the costs of dipping yards. On average, acaricides accounted for 11% of the costs, additional labor for 35%, and production losses and animal deaths for 32%.

A quarantine barrier was established on the border between New South Wales and Queensland to halt the southward spread of cattle tick and is maintained at an annual cost of around $3.3 million. The savings and benefits from this tick line were estimated by Davis[22] at $41.5 million a year. Sutherst[68] has predicted that global climate changes will affect pests and diseases in Australia and will have significant impacts on control costs and productivity; various models have indicated a possible increase in costs of $18 million to $192 million per year. It was suggested that insect pests, with their high reproductive rates, short generation times, efficient dispersal rates, and ability to adapt rapidly, will respond quickly to climate changes.

4.3.2 Screw-worm fly

Australia is the only continent with tropical regions that does not have screw-worm fly. The larval stages of screw-worm fly cause cutaneous myiasis. The New World screw-worm fly is classified as a B List disease by the Office Internationale des Epizooties, and the Old World screw-worm fly is classified as a C List animal disease by the Food and

Agriculture Organization. B List diseases are communicable diseases that are considered to be of socioeconomic or public health importance within countries and are significant in the international trade in animals and animal products. C List diseases are a group of animal diseases that are of socioeconomic importance at the local level.[28]

The major species of concern to Australia is the Old World screw-worm fly, *Chrysomya bezziana*, found in Papua New Guinea (PNG), Southeast Asia, India, parts of the Middle East, and Africa. Other species, such as *Dermatobia hominis* from South America, *Cochlyomyia hominovorax* from Central America, and *Cordylobia anthrophagia* from Africa, are of less concern, thanks to Australia's quarantine restrictions.

C. bezziana is an obligate parasite of all warm-blooded animals. Female flies are attracted to open wounds in the skin and lay eggs on the wound edges. The eggs hatch in 12 to 24 hours, and the larvae move into the wound and feed for 5 to 7 days, after which they drop off onto the ground to pupate. During the larval feeding phase, the wound enlarges in diameter and depth. The economic costs of screw-worm fly include occasional animal deaths, declines in production, damage to hides and underlying muscle, the cost of insecticides, and the cost of additional labor for treatment and management protocols. In Australia, where much of the cattle industry in the tropics involves minimal inspection of cattle, the introduction of screw-worm fly would require a marked change in management practices, with frequent livestock inspections for management of unstruck wounds and treatment of wounds already infected by the larvae.

If *C. bezziana* were introduced and became endemic, it would likely occupy a large area of northern Australia. A high probability of establishment exists year-round in tropical regions, except in areas around the Gulf of Carpentaria and in the Northern Territory, where dry midyear weather would limit survival.[6] Low temperatures make establishment in temperate areas unlikely.

The potential area of permanent colonization in Australia extends south to the mid-coast of New South Wales. Comparison of areas suitable for permanent establishment with the potential summer distribution indicates that large additional areas, carrying most of the continent's livestock, could be colonized in the summer months.[66] Since screw-worm fly can infect all warm-blooded animals, its economic impact depends on the juxtaposition of the fly, the climate, and suitable hosts. In northern Australia, the cattle industry would suffer the brunt of the impact, with some impact on sheep in more inland areas. Other species, including goats, horses, and domestic pets, would also suffer cutaneous myiasis from the fly. In 1979 it was estimated that the economic loss to the sheep and cattle industry if screw-worm fly were allowed to spread unchecked would be $101 million annually.[7]

C. bezziana could be introduced to Australia by the illegal importation of infected animals, by importation on infected people, by adult flies flying into Australia, or by flies being carried into Australia aboard boats or aircraft. Since the southern coast of the Western Province of Papua New Guinea is only 3 km from Saibai, the northernmost Australian island in Torres Strait, the possibility of *C. bezziana* being introduced from PNG is considered a significant threat. *C. bezziana* flies labeled with a radioactive tracer have been shown in PNG to deposit eggs a median distance of 10.8 km from their point of release, with a maximum distance of 100 km.[66] It seems feasible, therefore, that adult flies from the PNG mainland could arrive unaided on the northernmost islands of Torres Strait. In addition, the traditional visitors' treaty between Australia and PNG allows for free movement of people between coastal regions of the Western Province and the northern islands of Torres Strait for the purposes of trade and social interaction. Recent restrictions on movement of animals make introduction of infected animals from PNG less likely, but policing is difficult.

Screw-worm flies have arrived in Australia aboard boats and aircraft[59] and within cutaneous myiases on people. In 1988, *C. bezziana* flies were found aboard a vessel in Darwin harbor.[59] There have been no known introductions on animals. The cases on humans have involved the South American fly *Dermatobia hominis*, the tumbu fly *Cordylobia arthropophaga*, from Africa, and the New World screw-worm fly imported on a traveler from Argentina and Brazil.[55,60,61] These cases on humans present little risk, considering that these species involve only one larva per lesion. However, wounds infected with *C. bezziana* can contain thousands of larvae,[65] so the risk of a single person or animal bringing in sufficient numbers to establish the fly in Australia is much higher.

The economic costs of an eradication program, even if the need is detected early, may be quite high. When the New World screw-worm fly, *Cochliomyia hominovorax*, a species very similar in biology to *C. bezziana*, was introduced to Libya in 1988, the eradication campaign cost approximately $75 million (U.S.).[28] The annual regional benefit of eradicating this invasion was estimated to be $480 million (U.S.) at a benefit-to-cost ratio of 50:1.[28] In the United States, the same species in 1960 cost $100 million annually, and elimination from the southern United States and Mexico took more than 20 years and cost nearly $700 million.[28] The cost-benefit ratio for this eradication program was 1:10.[3] The economic cost of an invasion by *C. bezziana* depends on the point of entry; for Brisbane, it has been estimated at $281 million (Australian) per year.[1a]

In Australia, the approach to the screw-worm fly threat combines risk reduction, early detection, and preparedness.[7]

Risk reduction involves:

- Quarantine requirements for animals imported formally into Australia
- Prohibiting the informal movement of animals from PNG to the Australian Torres Strait islands and restricting movement of animals between islands in the Torres Strait
- Insecticidal sprays prior to arrival for aircraft and ships entering Australia
- A cattle-free zone in Cape York Peninsula
- Attempts to reduce feral animal populations on Torres Strait islands

Early detection involves:

- Education to alert Torres Strait islanders and communities on Cape York to screw-worm fly
- Submission of diagnostic specimens from struck animals
- Trapping of flies in traps baited with Swormlure
- A sentinel wounded animal scheme
- Trapping, which was instituted but is now only used to map the distribution of any introductions

Since female screw-worm flies mate only once, the main control method once screw-worm fly is detected is the release of sterile males. A factory to produce sterile male screw-worm flies was established in Port Moresby, but it is currently inactive. The sterile-insect-release method was used effectively as a major tool to eliminate flies in the Libyan outbreak and to eradicate screw-worm fly from the southern United States.

In Australia, the cost of prevention over a 20-year period was estimated in 1979 to be $20.23 million.[7] Modeling of a sterile-insect-release program showed it to be biologically and economically feasible.[3]

Monitoring in Torres Strait has shown that the risk of introduction via the Torres Strait is low.[3] The major risk is the illegal introduction of an infested animal. Ongoing monitoring is recommended. The screw-worm fly is an example of a pest that presents a significant economic cost to Australia even though it has never invaded the continent.

4.4 Invasions of importance to agriculture and forestry

Surprisingly, the statistics compiled annually to describe the economic value of agricultural and forest production in Australia do not reveal the general economic impact of arthropod pests and certainly give little indication of the particular impact of exotic pest species.

Some individual appraisals have identified massive impacts. For example, the introduced red-legged earth mite, *Halotydeus destructor* Tucker (Acarina: Penthaleidae), is believed to cause more than $200 million worth of damage each year to Australian pastures, and stored-grain pests collectively may account for $100 million in losses annually.[20] Even sporadic outbreaks of introduced pests can be extremely damaging. The European wood wasp, *Sirex noctilio* Fabricius (Hymenoptera: Siricidae), killed more than 5 million *Pinus radiata* trees on South Australian plantations between 1987 and 1989; the lost trees were valued at $10 million to $12 million.[26] The damage occurred despite the presence in Australia of effective biological control agents against the wood wasp.

What are we to make of these and other scattered, oft-repeated estimates of impacts when the methods of reckoning generally are not explained? Furthermore, in the absence of systematic, direct measures of losses at the farm gate, mill, or market, or of accurate costings of control measures, how are we to obtain quantitative impressions of the monetary impact of the full range of exotic pests?

This section describes two different approaches to the question. The first uses an estimation technique based on total production values and assumptions regarding crop losses. The second approach embodies more formal economic analysis and is based on more precise data for losses and the costs of control measures. The first approach gives no better than a first approximation of impacts, but it does allow us to sketch the broad picture. The second approach, though more rigorous, calls for data that are available for relatively few industries, commodities, or pests.

4.4.1 Estimation from production values

Table 4.1[14] summarizes a recent application of the estimation technique and lists 48 insect and mite species introduced accidentally into Australia between 1971 and 1995. Each species has been assigned a pest status (major, sporadic, or minor) based on the performance of the species in other countries and on Australian experience post-introduction. Admittedly, the approach is subjective.

Exactly 50% of the species listed in Table 4.1 are classified as major pests. A major pest causes economic loss over a large part of the distribution of a crop and requires control measures most of the time.[41] Clarke assumed that in the absence of control measures, a major pest would cause a loss of 10% or more in value of the commodity.[14] For many major pests, losses are potentially massive. The introduced codling moth, *Cydia pomonella* Linnaeus (Lepidoptera: Tortricidae), whose larvae tunnel into fruit, can render as much as 100% of the apples in an unprotected Australian orchard unmarketable.[32]

Approximately 23% of the species listed in Table 4.1 are classified as sporadic pests. These are pests that usually are unimportant, perhaps controlled by natural enemies or weather conditions, but which occasionally cause economic damage. A sporadic pest can cause damage equivalent to that of a major pest, but on average only once every 5 years. For such a pest, annualized yield losses would amount to 2%. The dock sawfly, *Ametastegia glabrata* Fallen (Hymenoptera: Tenthredinidae), is an example of a newly introduced sporadic pest. Its larvae feed on a range of herbaceous weeds, and the species has become widespread in southeastern Australia. It can become a significant pest in orchards, where populations build up on dock growing as a weed beneath apple trees. On maturation, larvae of the sawfly abandon the dock and tunnel into fruit in search of pupation sites.[46]

Ten percent of the species introduced between 1971 and 1995 are classified as minor pests. A minor pest feeds or oviposits on a valuable host plant but does not inflict economically significant damage. For example, larvae of the introduced oak leaf miner, *Phyllonorycter messaniella* Zeller (Lepidoptera: Gracillariidae), tunnel into the leaves of ornamental oak trees (*Quercus* spp.) and can be very abundant. However, the mines have no discernible effect on the performance of oaks as shade trees, and control measures are not called for.

Table 4.1 lists the host plant or commodity affected by each introduced species, as well as annual production values for hosts or commodities, obtained from publications of the Australian Horticultural Council, the Australian Bureau of Statistics, and the Australian Bureau of Agricultural and Resource Economics. Losses attributed to each introduced pest were estimated by calculating either 10% or 2% of these production values, depending on the status of the pest. Where expert opinion was available, these estimates were revised to reflect more precise knowledge of the impact of particular pests.

In reality, the individual losses and the aggregate annual loss of nearly $4.7 billion are, at best, estimates of potential losses. Control measures generally are applied, with varying degrees of success and at varying costs. In some grain crops, control measures against a key pest can cost 1% of the total production value, with 4% of the yield still being lost to the pest.[44] Effective control of horticultural pests can cost as much as 30% of crop value.[1] If we were to use the low value (i.e., effective control at a cost of 1% of the value of the crop) and assume no residual damage, total control costs would be as listed in the final column of Table 4.1, adding up to nearly $750 million per year.

Of course, the economics of pest management are complex. Decisions on the commitment of resources to pest control are made using various economic threshold, optimization, or decision theory models, or may be made without regard to economics at all.[5]

Table 4.1 does not include the many serious pests that were introduced to Australia before 1971. For example, many of the stored-product pests came with the First Fleet in 1788, and many others were introduced progressively (and probably repeatedly) during the 19th century. Since the compilation of the table's data in 1995, additional exotic species have been introduced accidentally to Australia. Thus Table 4.1 depicts the rate of increase of the economic losses caused by exotic species over the last quarter of the 20th century, rather than the total annual loss due to all exotic species. Nevertheless, it is indicative of the economic impact of exotic pests on Australian agriculture and forestry.

4.4.2 Papaya fruit fly

The papaya fruit fly, *Bactrocera papayae* Drew and Hancock (Diptera; Tephritidae), is one of the very few exotic arthropods for which economic impact and the costs of control in Australia have been documented in any detail. The insect, which attacks a wide variety of tropical and temperate fruit and vegetables, was detected on mainland Australia for the first time in October 1995. The papaya fruit fly (PPF) is a well-known and widely feared polyphagous horticultural pest. When the species was detected in North Queensland, a number of Australia's trading partners promptly imposed trade bans on susceptible fruit and vegetables originating in North Queensland or in Australia generally.

Following the initial discovery of the species in the Cairns area, a quarantine zone was established to prevent the spread of the pest to other parts of Australia. A program involving extensive surveillance and toxic baiting within the quarantine zone began. A cost-benefit analysis of proposals to eradicate the fly from the quarantine zone was performed,[1] and in 1996 an eradication program based on toxic baiting was begun. So far, the eradication program appears to have been successful.

Table 4.1 Estimated Economic Costs of Production Losses and Control Costs ($000) Associated with the Introduction of Insects, 1971–1995

Scientific name	Pest status	Industry	Production loss	Control cost
Mango psyllid	Unknown	Mango	?	?
Spotted clover aphid	Major	Pastoral	500,000	50,000
Abacarus hystrix	Major	Pastoral	500,000	50,000
Acyrthosiphon kondoi	Sporadic	Pastoral	100,000	10,000
Acyrthosiphon pisum	Sporadic	Pastoral/peas	100,000	10,000
Aleurocanthus spiniferus	Major	Citrus	6000	600
Aleurodicus dispersus	Sporadic	Horticultural	10,000	1000
Ametastegia glabrata	Sporadic	Raspberry/grape	7600	760
Apis cerana	Major	Beekeeping	3500	350
Aulacaspis tegalensis	Major	Sugarcane	85,000	8500
Bactrocera frauenfeldi	Minor	Horticultural	N/A	N/A
Bactrocera papayae	Major	Horticultural	200,000	75,000
Bemisia tabaci biotype B	Major	Horticultural	500,000	150,000
Bombus terrestris	Innocuous	N/A	N/A	N/A
Brevennia rehi	Major	Rice/sorghum	25,000	2500
Brontispa longissima	Major	Coconut/palms	1000	100
Chilo terrenellus	Major	Sugarcane	85,000	8500
Coptotermes formosanus	Major	Timber	600,000	60,000
Deanolis albizonalis	Major	Mango	6800	680
Dysaphis aucupariae	Innocuous	N/A	N/A	N/A
Eriophyes hibisci	Sporadic	Hibiscus	50	5
Eumetopina flavipes	Minor	Sugarcane	N/A	N/A
Frankliniella occidentalis	Major	Horticultural	250,000	100,000
Hercinothrips femoralis	Major	Horticultural	75,000	10,000
Heteropsylla cubana	Major	*Leucaena*	5000	500
Hylotrupes bajulus	Major	Softwood timber	200,000	20,000
Hypothenemus californicus	Minor	Wheat	N/A	N/A
Hypurus bertrandi	Innocuous	N/A	N/A	N/A
Idioscopus clypealis	Sporadic	Mango	1,400	140

Table 4.1 (continued) Estimated Economic Costs of Production Losses and Control Costs ($000) Associated with the Introduction of Insects, 1971–1995

Scientific name	Pest status	Industry	Production loss	Control cost
Idioscopus niveosparsus	Sporadic	Mango	1400	140
Ithome lassula	Major	*Leucaena*	5000	500
Lachesilla quercus	Nuisance	Bulk grain	N/A	100
Melittiphis alvearius	Innocuous	N/A	N/A	N/A
Metopolophium dirhodum	Major	Rose/barley	6500	650
Phenacoccus parvus	Major	Vegetables	7000	700
Phyllonorycter messaniella	Minor	Environmental	N/A	N/A
Pyrrhalta luteola	Major	Environmental	100,000	1000
Rhopalosiphon insertum	Sporadic	Fruit/grain	80,000	8000
Ribautiana ulmi	Sporadic	Environmental	N/A	500
Scapteriscus didactylus	Sporadic	Pastoral	100,000	10,000
Scaptomyza flava	Minor	Horticultural	N/A	N/A
Scolytus multistriatus	Sporadic	Environmental	N/A	500
Temnorrhynchus retusus	Innocuous	N/A	N/A	N/A
Therioaphis trifolii f. maculata	Major	Pastoral	500,000	50,000
Thrips palmi	Major	Horticultural	200,000	75,000
Trogoderma variabile	Major	Bulk grain	400,000	40,000
Varroa jacobsoni	Major	Apiary/honey	4,000	400
Vespula germanica	Nuisance	Environmental	N/A	1000
Total			**4,665,250**	**747,125**

Values for the ranching industry include losses in seed production as well as production losses associated with grazing and may be very conservative.

Source: Clarke, G.M., *Exotic Insects in Australia: Introductions, Risks and Implications for Quarantine*, Bureau of Resource Sciences, Canberra, 1996.

This cost-benefit analysis highlighted the extreme complexity of assessing the economic impact.

To start with, the choice of analytical technique is critical. General equilibrium analysis yields not only information regarding the industry or commodity directly affected by the pest, but also information on the consequences for other industries, as well as the macroeconomic effects. General equilibrium analysis requires large amounts of data — for example, to describe inter-industry effects. Partial equilibrium analysis requires less input data. It takes into account changes in the price and availability of commodities, and thus the effect on consumers. It gives a more realistic picture than do partial budgeting techniques, which focus largely on the effects on the particular industry or production system affected by the pest. Partial budgeting requires the least amount of input data.

An industry may have several possible strategies in the face of a new and damaging exotic pest. For example, with the advent of the papaya fruit fly, three options were available to Australian fruit and vegetable growers:

1. They could simply accept the pest and redirect their exports to countries already infested by the PFF. It was estimated that PFF-free markets offered a premium of $9 million per annum that would be lost to Australian producers as a result.
2. They could continue to export to premium, PFF-free markets, but only after fruit has been disinfested, at a cost of $7.59 million per year.
3. They could redirect their exports to the domestic market, resulting in a $28.97 million loss in producer economic surplus. If the PFF dispersed throughout Australia, there would be no disinfestation costs. If the fly did not spread, then disinfestation costs would amount to $12.67 million per year.

Clearly, the second alternative is the most attractive option for producers.

Many of the fruit and vegetables susceptible to the PFF are also susceptible to the native Queensland fruit fly and are protected by regular pesticide sprays. However, the presence of the PFF would require additional spray treatments. For example, bananas would require an additional six sprayings each year at a cost of $46/ha per spraying, and tomatoes would require an additional 10 sprayings, each costing $27/ha. Australia-wide, additional spraying treatments for all susceptible fruit and vegetables would amount to an additional cost of $53.25 million per year. Table 4.2[1] depicts the most likely cost scenario.

The ABARE analysis indicated that the annual cost to Australia as a result of the PFF incursion was likely to be approximately $74 million, which is significantly less than the figures given for the PFF (*B. papayae*) in Table 4.1.

The species was eradicated at an actual cost of approximately $35 million,[23] which is another estimate of the cost of the incursion of this exotic species into Australia.

4.4.3 Banana skipper

A recent cost-benefit analysis of a biological control program against the banana skipper, *Erionota thrax* Linnaeus (Lepidoptera: Hesperiidae), similarly provides insight into both the potential costs to Australia of an invasion by this pest species and the likely cost of control.[73] Biological control of this species has been demonstrated by a recent $700,000 program based in Papua New Guinea.

In the absence of biological control agents, banana skipper could be expected to cause production losses in Australia of up to $65.9 million per year. However, in the presence of biological control agents, these losses could be expected to shrink to approximately $3 million annually. Thus, the invasion of Australia by banana skipper would cost the country a one-off sum of $700,000 (an estimate of the research and development costs of a biological control program) and a recurring annual amount of $3 million. The Waterhouse et al.[73] analysis relied largely on partial budgeting.

4.4.4 Beneficial exotic arthropods

Apart from biological control agents, there are few examples of arthropod introductions to Australia that have created major economic benefits for agriculture or forestry. The honeybee, *Apis mellifera* Linnaeus (Hymenoptera: Apidae), and the leaf cutter bee, *Megachile rotundata* Fabricius (Hymenoptera; Megachilidae), stand out in this respect. The former is the mainstay of the Australian beekeeping industry, which has a gross production value between $60 million and $65 million per year, largely from honey, wax, and queen bees, which are a valuable export commodity.[33,34] Industry costs, principally involving labor and transportation, run at about 80% of revenue. However, the major and most pervasive economic impact of the honeybee is as a crop pollinator, with crops such as

Table 4.2 Estimated Annual Costs of Papaya Fruit Fly in Australia[a]

Source of costs	Within quarantine zone	For remainder of Australia[b]	Australia-wide
	($ million)	($ million)	($ million)
Economic losses on exports	0.08	7.51	7.59
Cost of insecticide treatments	0.37	52.88	53.25
Cost of disinfestation for domestic market	12.67	0	12.67
Total	**13.12**	**60.39**	**73.51**

[a] Undiscounted real values.

[b] Assumes 100% probability of infestation spreading from the quarantine zone to all the other suitable regions in Australia.

Adapted from ABARE, Papaya fruit fly. Cost-benefit analysis of the proposed eradication program, ABARE, Canberra, 1995.

apples, cotton, citrus, onions, and mangoes being particularly dependent. Benefits to Australia of up to $1.2 billion per year have been claimed.[33] This estimate may be conservative. The nitrogen enrichment of New Zealand soils by pasture legumes, all of which are pollinated by honeybees and leaf cutter bees, has been valued at $1.87 billion. Clearly, the economic benefits of introduced bees are substantial, but even these come with some cost, as related in the section on medical impacts.

The Asian bee constitutes a major threat to the entire bee industry because of its ectoparasite *Varroa jacobsoni*, a complex of five species that is currently undergoing renaming. Since 1980, these pests have been globally distributed via the illicit exportation of infested queen bees. The mites have been known for centuries as pests of Asian bees in India, China, Korea, and Southeast Asia, and they now commonly infest European bees. From these locations the mites were introduced into northern Europe and South America, and they are now established in the United States, Irian Jaya, and Papua New Guinea as well.[5] The introduction of this one particular species to the Australian European bee population would lead to significant reductions in crop yields and pollination success.

Currently, surveillance systems are being developed, because the complex of concern has been reported and eradicated from two islands near Irian Jaya, a close Australian neighbor, and the upper portion of the North Island of New Zealand is endemic. Management strategies in New Zealand are focusing on reducing the risk of spread to the southern part of the North Island and the South Island, which so far are free of this pest.[72] A delimiting survey designed to determine the current distribution of *Varroa* was completed in mid-2000. Heavy infestations were found in many areas, from cities to national parks, and it is assumed that feral bees are infected. The total estimated economic impact of this parasite due to such factors as reduced bee numbers — leading to reduced pollination, increased costs of pollination services, and increased costs for beekeepers — has been estimated at $400 million to $900 million (New Zealand dollars). The newly developed management program has been forecast to cost $40 million over the next 2 years. Eradication was considered, with an associated cost of $55 million to $70 million, but it was not pursued since the prospects for success were deemed to be too low.

4.5 Invasions of marine importance

Australia has a long coastline and many ports of call, which leaves it particularly exposed to invasions of invertebrate marine species, primarily via international shipping, but also through such mechanisms as discharge of ballast water, attachment to vessel hulls, importation for the aquarium trade and fish farming, deliberate introductions, movement of fisheries products, and transportation in fishing equipment or anchors. Surprisingly, surveillance and action against marine invaders in Australia have only recently been instituted.

In 1995, the National Introduced Marine Species Port Survey Program was initiated to provide baseline information on the status of introduced species in trading and other coastal ports.[19] To date, 170 exotic species have been found, although it is said that the scale of marine invasions in Australian waters is not really known.[21] Four of the identified exotic invertebrate species have been marked as likely to make significant contributions to economic and environmental costs in the years to come. They are the black-striped mussel *Mytilopsis* ssp., the Northern Pacific seastar *Asterias amurensis*, the sabellid fan worm *Sabella spallanzanii*, and the toxic dinoflagellate *Gymodinium catenatum*. The costs and many other factors relating to the other exotic species are unknown. Three of these species are discussed here.

4.5.1 Black-striped mussels

The black-striped mussel invasion of northern Australia was detected in 1999 in three marinas in Darwin, and 400 infested vessels have since been tracked. This was the first known incursion of a serious marine pest into Australian tropical waters, and a great deal of concern was expressed because of the potential for considerable economic and ecological damage. In a month-long eradication operation involving 250 people, more than 100 tons of chlorine and 10 tons of copper sulfate were dumped into infested waters, at a cost of $2 million. Concern stems from the fact that this mussel is a close relative of the zebra mussel, *Dreissina polymorpha*, which invaded the U.S. Great Lakes system in the 1980s and has had an economic impact since of more than $600 million per year. In India, the black-striped mussel has impacted in a similar manner to the zebra mussel by fouling all intertidal and sublittoral structures and vessels in large numbers. In Australia, predictions include infestation of marine oyster farms, marine pumping facilities (ballast and cooling systems), recreational and inshore vessels, and all port facilities. The costs are expected to be similar to the U.S. and India experiences. The potential environmental impact of this organism is predicted to be substantial, with the possibility of vast monocultures in low estuarine habitats.

4.5.2 Northern Pacific seastar

In 1986, the first *Asterias amurensis* (Lutken) seastar specimen was discovered in southern Australian waters near Hobart.[21] The Northern Pacific seastar's natural habitats include cooler coasts, ranging from the Bering Straits down to Canada and Japan. A decade later the species had become well established in the lower Derwent River and parts of several other estuaries and bays; two specimens were also found in Port Philip Bay.[8] In 1974, Australia's Commonwealth Scientific & Research Organisation (CSIRO) reported that the Derwent estuary was "the most polluted river in the world" and advised people not to eat anything from it. In 1999, the CSIRO released a media statement to the effect that there was a link between the pollution and the seastar population, which by then had reached 30 million. Port Philip Bay recorded 50 specimens in early 1998 and 12 million in 1999.[10] Due to the link with pollution, it is possible that the seastars in the Derwent estuary have made a negligible contribution to environmental costs.

It is still too soon to estimate control and economic costs for the Northern Pacific seastar, but they are clearly on the rise, as these incidents involving invasive marine invertebrates have prompted the federal government to endorse a $5 million dollar management program designed to address the threat.[4] This seastar is a well-adapted predator with a predilection for shellfish, but it is capable of consuming any animal tissue encountered and will dig for buried prey. Research into impacts and costs have been initiated, but no conclusive data have yet emerged. Considerable losses, in the range of millions of dollars per year, have been sustained by the shellfish mariculture industries in Japan, raising concern for wild shellfish fisheries in Australia. The Northern Pacific seastar is now recognized globally as a significant pest capable of causing great damage to the marine environment, aquaculture, and commercial and recreational fisheries.[29]

4.5.3 European fan worm

The European or sabellid fan worm, *Sabella spallanzanii*, was first identified in Western Australian waters in 1965. It has since made its way into eastern waters, and in 1992 it was the dominant organism in the polluted Port Philip Bay.[13] While there is no direct evidence in Western Australia to suggest that this species is negatively impacting any fisheries or native species, it is having a significant impact in Port Philip Bay, where scallop farmers are under threat. Seagrass beds have been overgrown, and competition for food has been detrimental to native oysters and other shellfish. *S. spallanzanii* is an efficient filter feeder and has a greater capacity to feed on phytoplankton than on seagrass.[47] This results in a significant and detrimental reduction in the amount of food in the system, which affects the entire ecology. No effort has been made to quantify the impact costs of this organism.

Management measures for ballast-water transfer, the suspected transfer mechanism in many cases, are progressing both on the national and international scene because of the ability of marine organisms to transcend all boundaries, and the necessity of limiting their distribution. In 1997, the International Marine Organization set a target date of 2000 for the implementation of an obligatory international framework for ballast-water management. Mandatory reporting arrangements have been in place in Australia since 1998.

References

1. ABARE, Papaya fruit fly. Cost-benefit analysis of the proposed eradication program, ABARE, Canberra, 1995.

1a. Atzeni, M.P. et al., Comparison of the predicted impact of a screwworm fly outbreak in Australia using a growth index model and a life-cycle model, *Med. Vet. Entomol.*, 8, 281–289, 1994.

2. Anderson, A.N., The Ants of Northern Australia, CSIRO, East Melbourne, Australia, 2000.
3. Animal Health Australia, Disease strategy: Screw-worm fly, AusvetPlan Edition 2.0, 1996. http://www.aahc.com.au/ausvetplan/swffinal.pdf.
4. AQIS, National arrangements for invasive marine species, AQIS Public Relations Bulletin, Commonwealth of Australia, 2000. http://www.aqis.gov.au:80/docs/bulletin/ab800_6.htm.
5. AQIS, Varroa mite. AQIS Public Relations, Commonwealth of Australia, 1999. http://www.aqis.gov.au:80/docs/schools/rl/8043.htm.
6. Atzeni, M.G., Mayer, D.G., and Stuart, M.A., Evaluating the risk of the establishment of screw-worm fly in Australia, *Aust. Vet. J.*, 75, 743–745, 1997.
7. Australian Bureau of Animal Health, Screw-worm fly: Possible prevention and eradication policies for Australia, Australian Government Printing Service, Canberra, 1979.
8. Australian Nature Conservation Agency, The introduced Northern Pacific seastar, *Asterias amurensis* (Lutken), in Tasmania, ANCA, Canberra, 1996.
9. Bureau of Agricultural Economics, The economic importance of cattle tick in Australia, BAE, Canberra, 1959.

10. Bureau of Rural Sciences, Aquatic pests and diseases, BRS, Commonwealth of Australia, 1999. http://www.brs.gov.au:80/fish/status99/aquatic.html
11. Caldwell, S.T., Schuman, S.H., and Simpson, W.M., Fire ants: a continuing community health threat in South Carolina, *J.S.C. Med. Assoc.*, 95, 231–235, 1999.
12. Cameron, D., Well nigh beneath contempt! Urbanisation and the development of manufacturing in Queensland, 1860–1930, "Images of the Urban" Conference, Sunshine Coast University College, University of Queensland, 1997.
13. Clapin, G. and Evans, D.R., The status of the introduced marine fan worm Sabella spallanzanii in WA, CSIRO Technical Report 2, Division Fish, CSIRO, 1995.
14. Clarke, G.M., Exotic Insects in Australia: Introductions, Risks and Implications for Quarantine, Bureau of Resource Sciences, Canberra, 1996.
15. Cleland, J.B. and Bradley, B., Dengue fever in Australia, *J. Hyg*. 16, 317–418, 1918.
16. Cosgriff M., *Culex gelidus* – Australia. ProMED-mail, 2000. http://osi.oracle.com:8070/promed/promed.folder.home?p_cornerid = 61.
17. Cosgriff M., Japanese encephalitis virus, mosquitoes – Australia, ProMED-mail, 2000. http://osi.oracle.com:8070/promed/promed.folder.home?p_cornerid = 61.
18. Crosland, M.W.J., The spread of the social wasp, *Vespula germanica*, in Australia, NZ J. Zool., 18, 375–388, 1991.
19. CSIRO, Black-striped mussel, CSIRO Marine Research, Hobart, 1999. http://www.csiro.au/page.asp?type = faq&id = BlackStripedMussel.
20. CSIRO, Division of Entomology Report of Research, CSIRO, Canberra, 1993.
21. CSIRO, The Northern Pacific Seastar, CSIRO Marine Research, Hobart, 1998.
22. Davis, R., A cost benefit analysis of the removal of the tick-line in Queensland, 42nd Annual Conference of the Australian Agricultural and Resource Economics Society, University of New England, Armidale, 1998. http://www.uq.edu.au/~ecrdavis/Thesis/PDF_Files/cbatline.pdf.
23. De Barro, P., A penny spent is a pound saved: Pre-emptive approaches to managing quarantine threats to primary industries, in *Plant Health in the New Global Trading Environment: Managing Exotic Insects, Weeds and Pathogens*, McRae, C.F. and Dempsey, S.M., Eds., National Office of Animal and Plant Health, Canberra, 1999, 151–162.
24. DeShazo, R.D., Williams, D.F., and Moak, E.S., Fire ant attacks on residents in health care facilities: a report of two cases, *Ann. Int. Med.*, 131, 424–429, 1999.
25. Doherty, R.L., Clinical and epidemiological observations on dengue fever in Queensland, 1954–1955. *Med. J. Aust.*, 1, 753–762, 1957.
26. Elliott, H.J., Ohmart, C.P., and Wylie, F.R., *Insect Pests of Australian Forests*, Inkata, Melbourne, 1998.
27. Evaluation Group, Department of Pedagogics and Scientific Studies in Education, James Cook University, Evaluation of the Dengue Fever Public Education Campaign 1982–1986, Report to the Queensland Department of Health, 1990.
28. FAO, Eradicating the screwworm, Food and Agriculture Organization, Rome, 1992.
29. Fisheries Western Australia, Introduced marine aquatic invaders: Northern Pacific seastar, FWA. 2000, http://www.wa.gov.au/westfish/hab/broc/marineinvader/marine01.html.
30. Prahlow, J.A. and Barnard, J.J., Fatal anaphylaxis due to fire ant stings, *Am. J. Forensic Med. Pathol.*, 19, 137–142, 1998.
31. Freeman, T.M., Hymenoptera hypersensitivity in an imported fire ant endemic area, *Ann. Allergy Asthma Immunol.*, 78, 369–372, 1997.
32. Geier, P.W., The codling moth, *Cydia pomonella* (L.): profile of a key pest, in *The Ecology of Pests*, Kitching, R.L. and Jones, R.E., Eds., CSIRO, Melbourne, 1981, 109–129.
33. Gibbs, D.M.H. and Muirhead, I.F., The Economic Value and Environmental Impact of the Australian Beekeeping Industry, A report prepared for the Australian Beekeeping Industry, Australian Honeybee Industry Council, Maroubra, 1998.
34. Gill, R., Beekeeping and Secure Access to Public Land — how it benefits the industry and society, A report for the Rural Industries Research and Development Corporation and the Honeybee Industries Research and Development Council of Australia, RIRDC Research Paper Series 97/16, 1997.

35. Halstead, S.B., Dengue hemorrhagic fever: a public health problem and a field for research, *Bull. WHO,* 58, 1–21, 1980.
36. Hanna, J.N., et al., An outbreak of Japanese encephalitis in the Torres Strait, Australia, 1995, *Med. J. Aust.* 165, 256–260, 1996.
37. Hanna, J.N., et al., Japanese encephalitis in north Queensland, Australia, 1998, *Med. J. Aust.* 170, 533–536, 1999.
38. Hanna, J.N., Japanese encephalitis — Australia [3], ProMED-mail post, 1998. http://osi.oracle.com:8070/promed/promed.folder.home?p_cornerid = 61.
39. Harvey, P., et al., Bee sting mortality in Australia, *Med. J. Aust.* 140, 209–211, 1984.
40. Hayes, E.B. and Gubler, D.J., Dengue and dengue hemorrhagic fever, *Pediatr. Infect. Dis.* 11, 311–317, 1992.
41. Hill, D.S., *Agricultural Insect Pests of the Tropics and their Control,* Cambridge University Press, Cambridge, 1987.
42. Hoffman, D.R., Reactions to less common species of fire ants, *J. Allergy Clin. Immunol.*, 100, 679–683, 1997.
43. Honan, P., Proceedings of the European wasp strategy meeting, 11 September, Victorian Department of Natural Resources and the Environment, 1997.
44. Hughes, R.D., Predicted consequences of the establishment of the Russian wheat aphid (*Diuraphis noxia*) in Australia, Bureau of Resources Sciences, Canberra, 1992.
45. Kay, B.H. et al., Dengue fever. Reappearance in northern Queensland after 26 years. *Med. J. Aust.*, 140, 264–268, 1984.
46. Kemp, S.F., et al., Expanding habitat of the imported fire ant (*Solenopsis invicta*): a public health concern, *J. Allergy Clin. Immunol.*, 105, 683–691, 2000.
47. Lemmens, J.W.T., et al., Filtering capacity of seagrass meadows and other habitats of Cockburn Sound, Western Australia, Marine Ecology Progress Series 143, 187–200, 1996.
48. Levick, N.R., et al., Bee and wasp sting injuries in Australia and the USA: Is it a bee or is it a wasp and why should we care? in *The Hymenoptera: Evolution, Biodiversity and Biological Control*, Austin, A.D. and Dowton, M., Eds., CSIRO Publishing, Canberra, Australia, in press.
49. Levick, N.R., Winkel, K.D., and Smith, G.S., European wasps: an emerging hazard in Australia, *Med. J. Aust.*, 167, 650–651, 1997.
49a. Lumley G.F. and Taylor F.H., *Dengue*. School of Public Health and Tropical Medicine, Service Publication No. 3. University of Sydney & Commonwealth Department of Health, Australasian Medical Publishing Company, Sydney, 1943.
49b. McBride, W.J.H. et al., Determinants of dengue 2 infection among residents of Charters Towers, Queensland, Australia. *Am. J. Epidemiol.*, 148, 1111–1116, 1998.
50. McGain, F. and Winkel, K.D., Bee and wasp sting related fatalities in Australia, L122, in *XIIIth World Congress of the International Society on Toxicology* Abstract Book, Paris, September 18–22, 2000.
51. McGain, F., Harrison, J., and Winkel, K.D., Wasp sting mortality in Australia, *Med. J. Aust.*, 173, 198–200, 2000.
52. Meltzer, M.I., et al., Using disability-adjusted life years to assess the economic impact of dengue in Puerto Rico: 1984–1994, *Am. J. Trop. Med. Hyg.*, 59, 265–271, 1998.
53. Mumford, J.D. and Norton, G.A., Economics of decision making in pest management, *Ann. Rev. Entomol.*, 29, 157–174, 1984.
54. New, T.R., *Exotic Insects in Australia*, Gleneagles Publishing, Adelaide, 1994.
55. Ng, S.O. and Yates, M., Cutaneous myiasis in a traveller returning from Africa, *Aust.J. Derm.*, 38, 38–39, 1997.
56. Phillips, D., Japanese encephalitis — Australia [1], ProMED-mail post, 1998. http://osi.oracle.com:8070/promed/promed.folder.home?p_cornerid = 61.
57. Preslar, D., Japanese encephalitis — Australia [2], ProMED-mail post, 1998. http://osi.oracle.com:8070/promed/promed.folder.home?p_cornerid = 61.
58. QDPI (Queensland Department of Primary Industries), Animal and plant health: Red Imported Fireants, 2001. http://www.dpi.qld.gov.au/health/3125.html.
59. Rajapaksa, N. and Spradbery, J.P., Occurrence of the Old World screwworm fly *Chrysomya bezziana* on livestock vessels and commercial aircraft, *Aust. Vet. J.*, 66, 94–96, 1989.

60. Rubel, D.M., et al., Dermal myiasis in an Australian traveller. *Aust. J. Derm.*, 34, 45–47, 1993.
61. Searson, J., et al., Screw-worm myiasis in an overseas traveller — case report, *Com. Dis. Intel.,* 16, 239–240, 1992.
62. Semevsky, F.N., Thompson, L.C., and Semenov, S.M., An econometric evaluation of the impact of fire ants on agricultural plant production in the southeastern USA, pp. 144–148, Proceedings 10th All-Russian Myrmecological Symposium, Moscow, 1998.
63. Shepard, D.S. and Halstead, S.B., Dengue (with notes on yellow fever and Japanese encephalitis), *Disease Control Priorities in Developing Countries*, Jamison, D.T., Ed., Oxford University Press, New York, 1993, 303–320.
64. Spradbery, J.P. and Maywald, G.F., The distribution of the European or German wasp in Australia, past, present and future, *Aust. J. Zool.*, 40, 495–510, 1992.
65. Spradbery, J.P., A manual for the diagnosis of screwworm fly, Australian Government Publishing Service, Canberra, 1991.
66. Spradbery, J.P., et al., Dispersal of the Old World screw-worm fly *Chrysomya bezziana*, *Med.Vet. Entomol.*, 9, 161–168, 1995.
67. Stafford, C.T., Hypersensitivity to fire ant venom, *Ann. Allergy Asthma Immunol.*, 77, 87–95, 1996.
68. Sutherst, R.W., Impacts of climate change on pests, diseases and weeds in Australia, Report of an International Workshop, Brisbane 1995, CSIRO Division of Entomology, Canberra, 1996.
69. Sutherst, R.W., Spradbery, J.P. and Maywald, G.F., The potential geographical distribution of the Old World screw-worm fly, *Chrysomya bezziana*, *Med. Vet. Entomol.*, 3, 273–280, 1989.
70. Taber, S.W., Fire Ants, Texas A&M University, College Station, Texas, 2000.
71. Walker, A.S., et al., Dengue fever. *Med. J. Aust.*, Sept. 12, 223–228, 1942.
72. Wallingford, N., New Zealand Beekeeping: Varroa Outbreak, April/May 2000, Bee Keeping in New Zealand, 2000. http://www.beekeeping.co.nz/disease/varroa.htm.
73. Waterhouse, D., Dillon, B., and Vincent, V., Economic benefits to Papua New Guinea and Australia from the biological control of banana skipper (*Erionota thrax*), ACIAR Impact Assessment Series 12, 2000.
74. Winkel, K., Hawdon, G., and Ashby, K., Venomous bites and stings, *Aust. J. Emer. Care*, 5, 13–20, 1998.

section three

Brazil

chapter five

Alien plant pathogens in Brazil

Murillo Lobo Junior

Contents

5.1 Introduction

The introduction of exotic crops in Brazil began in the early 16th century, when Portuguese navigators brought citrus plants, sugarcane, and wheat. The process of enhancement of the genetic diversity proceeded in the ensuing years with the successful cultivation of many other crops, such as coffee, common bean, maize, and several vegetables, fruits, and grains. Today these crops represent the great majority of the Brazilian population's diet.

Seeds, seedlings, and fruits of a large array of crops were freely introduced for almost the next four centuries with a complete absence of phytosanitary protection measures. Little could be done for many years, given the lack of knowledge of the biotic causes of plant diseases, which were proven with the aid of Koch's postulates at the

0-8493-0836-4/02/$0.00+$1.50

end of the 19th century. Finding a favorable environment for their development, and in the presence of susceptible hosts, hundreds of biotrophic and necrotrophic pathogens — among them fungi, bacteria, nematodes, viruses, and viruslike particles — were successfully established in the country.

Plant pathogens introduced in Brazil include at least 550 fungi, 100 viruses, 25 nematodes species, and one protozoan.[1–4] Prokaryotes appear in at least 51 records, including specific pathovars.[5,6] The main consequences of their presence have been expressed as quantitative and qualitative yield losses, increased production costs, environmental pollution, intoxications, and a lower market value for diseased produce. Other important consequences are energy loss, low seed emergence, reduced stand, decreased nutritional value, interdiction of seed production fields, longer periods of crop rotation or fallow, and underutilization of fertilizers and other chemical inputs.[7–10]

Nasser et al. indicated that 10 to 20% of the Brazilian grain yield is lost due to plant diseases.[11] Considering the prevalence of alien pathogens in almost all important crops, as well as increasing disease intensities recently recorded, about 15% of the country's plant production was lost in the year 2000, which resulted in an estimated damage* of 12.45 million tons, accounting for $5 billion worth of losses, out of the 83 million tons harvested.[11] If losses of 10% on vegetables, citrus, and sugarcane are added,** the total losses caused by alien pathogens in Brazil reach $6.9 billion.

Adding qualitative losses, chemical control costs, and environmental damages yields a very high cost to be paid by a country in which 18% of the population is undernourished. The main diseases that enhance this situation, the entrance and spread of alien pathogens in the country, and the economical, social, and environmental consequences are discussed in this chapter.

5.2 Alien pathogens in Brazil

The Brazilian official phytosanitary history started only in the 20th century, when pest control programs addressed to coffee and cotton crops were set up in 1909, followed by the first Brazilian laws dealing with plant protection in 1934. The phytosanitary barriers and other preventive measures were insufficient to prevent the entry of alien pathogens, introduced in the country almost totally through infected seeds or seedlings. Most of these were recorded in the country in the past 40 years (Table 5.1), and many of them have impaired the sustainable production of several crops, especially with the increased agricultural activity over the past 10 years.

The exact geographic origin of all non-indigenous pathogens in Brazil is not traceable. In fact, plant pathogens were already present in the world before the current political frontiers, and their alien status is almost always supported by evidence.[14] Plant pathogens are considered as aliens if specific to an exotic crop or if they have a restricted host range, as do many *Fusarium formae especialis, Xanthomonas* pathovars, and cereals rusts. Pathogens with a wide range of hosts and previously described in other countries may also be accepted as non-indigenous, probably introduced in the country in the past along with their hosts or vectors.

Some relevant diseases, such as soybean sudden death syndrome or the potato blackeye, both caused by *Fusarium solani*, may have their status (alien vs. native) questioned, since their causal agents exist naturally in the country, in spite of their diseases having been previously registered in other countries.[15]

* Terminology used here is in agreement with Zadoks.[12]

** According to crop yield statistics data from the Ministry of Agriculture and Food Supply, the Brazilian Institute of Geography and Statistics, and the Getúlio Vargas Foundation.

Table 5.1 Some Plant Pathogens Recorded in Brazil from 1970 to 2000, Their Common Disease Names, and Their Respective Hosts

Decade	Pathogen	Common name of diseases	Main host
1970	*Ditylenchus dipsaci*	Bulb nematode	Garlic
	Grapevine leafroll virus	Leafroll	Grape
	Hemileia vastatrix	Rust	Coffee
	Physopella zeae	Tropical rust	Maize
	Xanthomonas fragariae	Angular spot	Strawberry
1980	Pea seedborne mosaic virus	Mosaic	Pea
	Phialophora gregata	Brown rot	Soybean
	Phytomonas staheli	Hart rot	Coconut
	Puccinia melanocephala	Rust	Sugarcane
	Xylella fastidiosa	Citrus variegated chlorosis	Orange
1990	*Claviceps africana*	Ergot	Sorghum
	Heterodera glycines	Soybean cyst nematode	Soybean
	Mycosphaerella fijensis	Black sigatoka	Banana
	Spongospora subterranea	Powdery scab	Potato
	Xanthomonas campestris pv. *viticol*a	Canker	Grape
	Zucchini yellow mosaic virus	Mosaic	Cucurbits

Adapted from Kimati, H., et al., Eds., *Manual de Fitopatologia, Doenças das Plantas Cultivadas,* 2, Ceres, Piracicaba, 1997, 774.

5.3 Brazilian agriculture — alien pathogens

Historically, monoculture has been the preponderant characteristic of Brazilian agriculture. Several extensive monocultures marked the country's history in such a way that specific phases of that history are known as "coffee cycle" or "sugarcane cycle." The expansion of these crops was favored by good soil and climate conditions, and they were mainly addressed to overseas markets. Such characteristics favored equally the succession of explosive epidemics caused by airborne pathogens such as the sugarcane smut (*Ustilago scitaminea*) and the slower development of soilborne pathogen epidemics such as the Panama wilt, which attacks bananas (*Fusarium oxysporum* f. sp. *cubense*).[16]

In the 1960s, the country adopted the agricultural production model known as the "green revolution." High-input models, mechanization, and the genetic improvement of crops — addressed to specific phenotypes — sustained Brazilian agriculture progress, allowing the expansion of cropped areas and leading to striking increases in crop productivity, while disregarding such important practices as crop rotation, which became of secondary importance.

A greater amount of information about alien pathogens is available for diseases that affect cash crops or highly important subsistence crops that are produced in the south-central part of Brazil (Figure 5.1), where the more technically advanced agriculture is carried out. In the northern and northeastern* regions, where subsistence agriculture prevails, farmers experience the consequences of non-indigenous pathogens and suffer from a lack of resources for disease control, insufficient technical support, and a lack of high-quality seeds and resistant varieties for these regions.

Many differences have occurred in Brazilian agriculture since the reports of Echandi et al.[20] The 1990s were marked by an expansion of the cultivated area with no tillage systems (13.5 million ha under no tillage in the 1998–99 season), and by increased agricultural activity. The rapid conversion of extensive natural areas into agriculture use has

* The Northeast, which holds 46% of the Brazilian rural population and 25% of the country's agricultural productivity, has an infant mortality rate of 52/1000, indicative of chronic hunger in the region.[17–19]

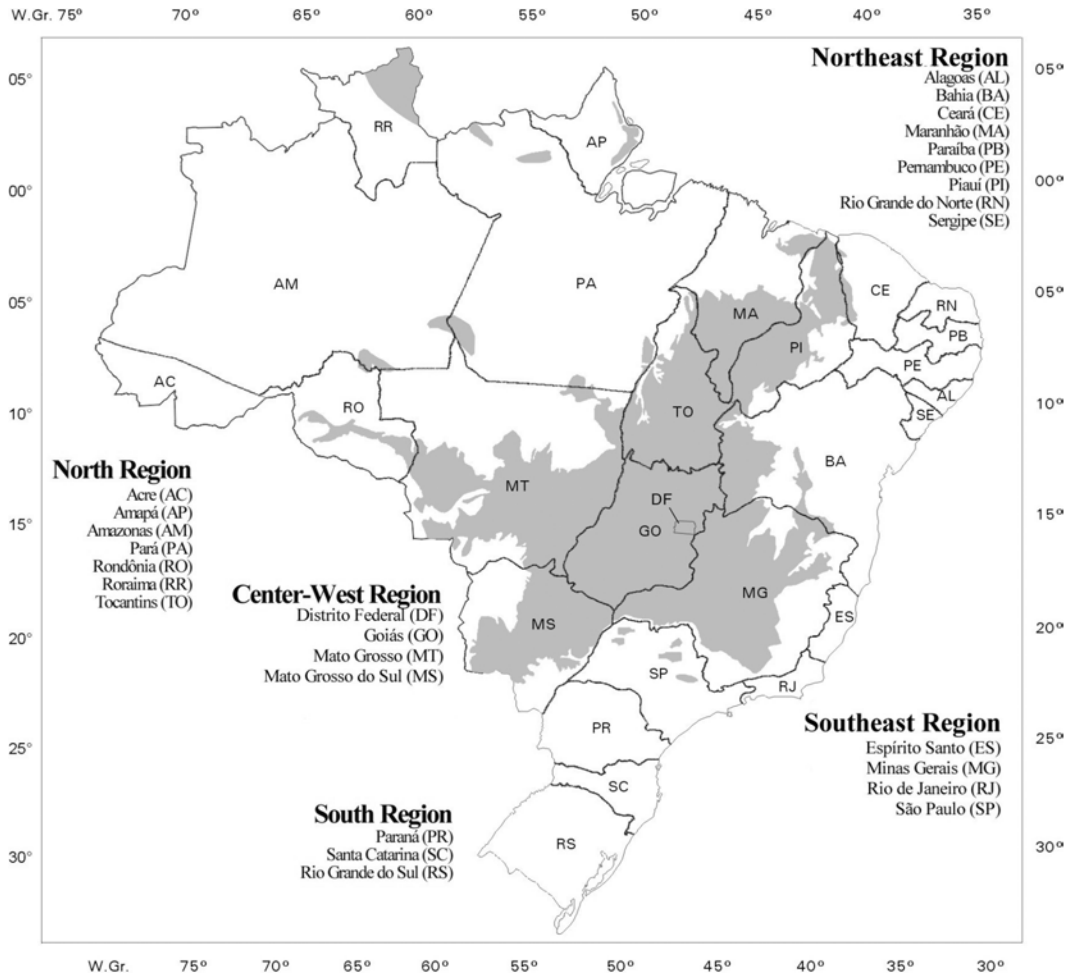

Figure 5.1 Brazilian political map, showing the country's geographic regions and respective states. The gray area corresponds to the Cerrados region (Brazilian savannas).

no parallel in the country's history. Regarding the Cerrados* region, 40% of its 200 million ha were deeply altered, as can be illustrated by the growth in soybean acreage, which rose from 10,000 ha in the 1969–70 cropping season to 13.6 million ha in 1998–99.[18,21] Nowadays, the Cerrados region is responsible for 50% of the grains harvested in the country.

Such progress is reflected in the contribution of Brazilian agribusiness in the Brazilian Gross Development Product (21% of the GDP), representing an income of $137 billion.[22] The soybean (accounting for 25% of Brazilian agricultural exports), sugarcane, maize, and coffee crops each earn more than $2 billion annually.[22]

Plant diseases must be skillfully controlled, or their consequences will directly and indirectly affect Brazil's economy and society. To avoid further damage caused by the establishment of new alien pathogens, governmental efforts were intensified in 1994, when the country joined the World Trade Organization. Since then, restrictions to the free market have been allowed only when supported by scientific or technical advice, and the country invested $100 million in 1998–99 on new and improved phytosanitary barriers.[23]

* Brazilian savannas, which comprise part of the Center-West, North, Northeast, and Southeast regions.

Table 5.2 Average Yields of Brazilian Agricultural Crops Recorded in 2000, according to the Brazilian Institute of Geography and Statistics (IBGE)

Crop	Average yield (tons/ha)
Cotton (grains)	2.4
Rice	3.0
Potato (Sept.–Dec.)	16.0
(Feb.–May)	16.4
(May–Aug.)	25.0
Coffee	1.6
Sugarcane	66.3
Onions	17.3
Common bean (Oct.–Jan.)	0.7
(Feb.–April)	0.6
(April–Sept.)	1.8
Oranges (×1000 fruits per ha)	126.0
Maize (Oct.–Feb.)	2.9
(Feb.–June)	1.9
Soybeans	2.4
Wheat	1.5

The number of recorded pathogens for all agricultural crops increased with the expansion of crop areas, and the diseases from these pathogens are major causes for Brazil's low agricultural productivity (Table 5.2). Several minor diseases in the southern and southeastern regions gained significance in the central-western region and assumed epidemic proportions there, being supported by conditions favorable to the pathogens (temperature, crop residue, and moisture).

5.4 Social and economic impacts

In Brazil, 18 million people (25% of the economically active population) work in the agricultural sector, each earning an average income of $4500 a year, well below most other sectors of the economy.[17,22]

The lack of technical assistance and economic resources to protect against agricultural diseases has resulted in a great deal of crop losses.[24–26] One example is the impact of *Xylella fastidiosa* on coffee crops, where disease severity is proportional to plant stress caused by poor handling of the crop nematodes and exposure to long drought periods. [27,28]

The technology used to cultivate commodity crops on midsize and large farms was largely unavailable to farms of less than 50 ha, which account for 80% of rural farms in Brazil.[18] On these farms, the agricultural activity mainly involves subsistence crops such as common beans (Brazilians' main dietary source of protein) and is carried out by family labor earning an average income of $70 a month.[17]

Common bean crops are a fine example of technical disparities. Traditionally grown by small farmers during the rainy season, common beans are also grown during the dry season (April–September) in large center-pivot-irrigated areas. Diseases affect crops in both systems, but poor disease management on small farms contributes to outstanding yield differences. Small farmers sow their own seeds (which often are not healthy) and seldom spray their crops. Their average yields have been 700 kg/ha, a figure that is slightly above yields in 1966–70, whereas commercial irrigated crops now yield 1800 kg/ha during the dry season.[18,29]

The visual impact of diseases is easily noted in the field, but systematic evaluations of yield losses and damages have been scarce.[36] Often, disease assessments are carried out with subjective evaluations, and farmers are commonly satisfied with yields such as 2000–2500 kg/ha on soybeans.[10]

5.5 Cultural practices and the spread of pathogens

Several common practices in Brazil contribute to an increased risk of soil degradation, as well as to environmental pollution. These practices include an intensive use of the soil, poor sanitation, the planting of monocultures, insufficient crop rotation, inefficient irrigation management, and a trend toward increased plant density.[31–35] In general, these conditions have also been ideal for the emergence of airborne and soilborne pathogen epidemics, as well as a higher disease intensity from formerly minor pathogens.

High temperatures, plentiful sunlight, and short leaf wetness periods are prevalent in the Brazilian climate, and these conditions usually limit spore survival. Nevertheless, many foliar pathogens (e.g., *Alternaria solani, Mycosphaerella musicola, Pyricularia oryzae*) are well adapted to these conditions. The diseases they cause progress through lesion expansion, where the infection of sites adjacent to those already diseased partially suppresses the necessity of spore release.[36] Other non-indigenous plant pathogens, such as viruses, bacteria, nematodes, and soilborne fungi, are also poorly affected by tropical climate, and are well adapted in the country.[36]

A lack of effective control measures resulted in the rapid spread throughout Brazil of introduced airborne pathogens, such as *Hemileia vastatrix*, or coffee rust), which was found in the main coffee production areas only 2 years after it was first recorded in Bahia state in 1970. Nowadays, *H. vastatrix* threatens four billion susceptible coffee plants grown in an area of 2.8 million ha that produces a crop worth $2.5 billion per year.[37,38] Yield losses caused by coffee rust reach 35% under conducive weather conditions if spraying is not done.[28,38] Soilborne pathogens, in turn, usually spread slowly, but sclerotia (*Sclerotinia sclerotiorum* × common bean) and chlamidospore (*Fusarium oxysporum* f. sp. *vasinfectum* × cotton) populations, which build up under intensive cropping, became equally important and destructive. Relevant cases of the impact of alien pathogens on several important crops are related below.

5.5.1 Soybean

More than 40 pathogens together are responsible for annual soybean crop losses of $1 billion in Brazil.[10] These losses include yield reduction, decreases in quality, and poor seed emergence. Soybean crops cover 40% of the country's grain area, and since the late 1980s these crops have experienced the successive emergence of destructive diseases, such as the soybean stem canker (*Diaporthe phaseolorum* f. sp. *meridionalis*), soybean cyst nematode (*Heterodera glycines*), and late foliar disease complex.

From restricted occurrence in the 1988–89 season, *D. phaseolorum* f. sp. *meridionalis* was found in the following season in all main areas of soybean production.[39] In infected fields that had few infected plants in the previous year, pathogen spores were disseminated by rainfalls during the crop's first 40 to 50 days, leading to the extensive destruction of susceptible soybean varieties.[40] Before it was controlled by the planting of resistant varieties in 1997–98, stem canker caused losses of $500 million.[10]

The soybean cyst nematode (SCN, *Heterodera glycines*) was detected in Brazil in 1992,[41] and it soon thereafter became one of the most serious threats to soybean crops. From 5000 ha infested in 1992, SCN spread quite rapidly, leading to an infested area in the 1993–94 season of more than 1 million ha, and severely infecting the main soybean varieties.[42,43] Without resistant varieties or chemical control, farmers have managed SCN through cultivation practices, in order to reduce nematode populations and avoid the pathogen's transport to uninfected areas.[44]

Other currently important diseases on soybean crops are the late foliar disease complex, caused mainly by *S. glycines, C. kikuchii, Colletotrichum truncatum*, and *Phomopsis* sp., which are responsible for yield losses of 20%. Yorinori[10] estimated that damage caused by late-season diseases reached 5 million tons in the 1989–90 season, yielding a loss of $830 million. To avoid losses, fungicide sprayings have been recommended since 1995.

5.5.2 Maize

Maize cultivation occupied 13 million ha of Brazil in 2000, yielding a harvest of 33 million tons, falling short of the 35 million tons needed by Brazil, 58% of which is used for animal feed. This deficit drives increases in cropped area and expansion of the second harvest, or *safrinha*, which is grown between February and June. The *safrinha* area in 2000 was equal to 11% of the main season area. The second harvest also has a high inoculum pressure, which has affected the performance of varieties and hybrids that have intermediate resistance.[45]

Maize, formerly a rustic crop, now has plant diseases as a main constraint to crop production. Several reports indicate an increase in the severity of traditional and secondary fungal diseases throughout the country during the past decade, with a corresponding impact on average yields.[46]

Despite its presence in Brazil since 1902 and its status as a secondary pathogen in other countries, *Phaeosphaeria maydis* is now a key pathogen, due to crop residues in no-tillage systems and a prevalence of susceptible varieties and hybrids. The pathogen, which is thoroughly disseminated throughout the country, kills many young plants and also leads to smaller kernel size. Other formerly minor pathogens are now also relevant, such as *Physopella zeae* (only 11% of hybrids are resistant) and *Puccinia polysora*.[47,48]

The survival of vector populations was also favored, and since 1995 has resulted in the increased importance of the maize stunt (*Spiroplasma kunkelii*), maize bushy stunt (mycoplasmalike organism), and fine stripping disease (*Maize rayado fino virus*).[48,49]

Furthermore, a new emergent disease caused by *Cercospora* sp. was recently recorded, although the pathogen was given scant coverage in recent publications. Uncontrollable *Cercospora* leaf-spot epidemics were recorded in the center-west, causing yield damages of 80% and leading farmers to spray their crops with fungicide.[50]

5.5.3 Common bean

With the aid of irrigation, common-bean monocultures can yield up to 7.5 tons/ha each year if four successive crops are planted. Attracted by high profits, capitalized farmers often grow year-round crops or plant continuous short rotations. Intensive cropping has created ideal conditions for epidemics of the foliar diseases anthracnose, angular leaf spot, and rust, thereby increasing a dependency on fungicide sprayings.[51] Because foliar diseases can be managed well with fungicides, intensive cropping continues, usually until producers meet problems caused by soilborne pathogens, which are more difficult to control.

Sclerotinia sclerotiorum is the most important soilborne pathogen in Brazilian common-bean crops, and is one of the best-known examples of a pathogen that was spread through the country. First recorded in Brazil in 1920, *S. sclerotiorum* was disseminated throughout the southeastern and southern regions in the 1940s and 1950s as a secondary pathogen to common beans and vegetables.[29,52,53] Brazil's first severe *S. sclerotiorum* epidemic was reported in 1976, involving soybean crops in Paraná state (southern region), and from there infected seeds were carried to the Cerrados region in the 1980s.[54] In almost 10 years, *S. sclerotiorum* became present in 50% of the center-pivot-irrigated common-bean crops, causing severe white-mold epidemics and becoming the region's leading soilborne pathogen on common beans. It also attacked processing tomatoes and peas cropped in the dry season (April to September).[30,55,56] Recent reports showed that *S. sclerotiorum* is now present in almost all center-pivot-irrigated areas cropped to common beans, where it can cause up to 100% yield losses.[30,54,56,57]

5.5.4 *Rice, wheat, and small-grain crops*

The phytosanitary status of rice, wheat, and small-grain crops in Brazil is far from satisfactory. Several varieties are susceptible or highly susceptible to the most common diseases and require careful disease management. Rice blast (*Pyricularia oryzae*) is the most important disease of rice in Brazil. Although yield damages usually range between 15 and 30%, losses can reach 100% when control measures are not carried out.[58] The pathogen was also found on wheat and rye in the Cerrados region in 1985, highlighting a need for control measures.[59]

Wheat and small-grain diseases gained in importance as these crops moved from traditional to no-tillage systems, which now account for about 70% of the total acreage. Accumulation of crop residues in the soil cover led to an increase in necrotrophic leaf-spotting pathogens, such as *Bipolaris sorokiniana, Dreschlera tritici-repentis, D. avenae, D. teres, Giberella zeae, Septoria nodorum*, and *Pseudomonas syringae* pv. *coronafasciens*, all introduced and disseminated by infected seeds.[14,32,33,60] These and other pathogens remain viable in the soil on the straw coverage, as a reservoir of initial inoculum for the following crop. Seed treatment with fungicides, combined with crop rotation, has proved to be a successful and widespread control measure for diseases such as speckled leaf blotch (*Septoria tritici*), which is now of secondary importance. [14,32,33,60]

5.5.5 *Vegetables*

In general, vegetable growers produce intensively cultivated, economically valuable solanaceous, cucurbits, apiaceous, carrot, and lettuce crops. Although these crops represent the best examples of successful cropping by small farmers, they usually involve high production costs, a high susceptibility to pathogens, and a dependency on chemicals.

Potatoes are Brazil's leading vegetable crop, yielding 2.5 million tons annually from a crop area of 170,000 ha. All the main varieties planted (e.g., Achat, Bintje, Baraka, and Atlantic) were bred in the Northern Hemisphere and are moderately or highly susceptible to their major alien pathogens, *Phytophthora infestans* and *Alternaria solani*.[61] The same holds true for tomatoes, which yield 2.6 million tons annually from 60,000 ha, much of which is planted with varieties and hybrids that are not adapted to the Brazilian edaphic and climatological conditions.[62] Other alien pathogens on these and other crops, such as *Alternaria dauci, Cercospora carotae, C. beticola*, and *Xanthomonas* spp., also contribute to a dependency on fungicides and antibiotics in intensive cropping systems. The build-up of soil-borne pathogens (e.g., *Fusarium oxysporum* f. sp. *cucumerinum, Sclerotium cepivorum*, and *S. sclerotiorum*) may make the cultivation of their host crops impractical in infested areas.

Vegetables are grown year-round, given sufficient irrigation or rainfall, and proper cultivation practices are critical for disease management. Lack of proper sanitation, improper chemical application, a favorable environment for pathogens, and large continuously cropped areas are considered the major factors that render vegetable crops vulnerable to both native and alien pathogens.[63,64]

Several important viruses and nematodes are prevalent in the tropics, and may be considered native. Among non-indigenous viruses, the importance of geminiviruses on processing tomatoes (and other crops, such as common beans) grew dramatically since 1997, along with the spread and fast reproduction of the white fly (*Bemisia argentifolii* = *B. tabaci biotype B*), leading to losses of up to 100% in the center-west and northeast regions.[66] Frequent sprayings have been necessary for vector control — not often successful — mainly in order to avoid early infections, which precipitate larger losses. In the northeast region, geminiviruses contributed to low productivity and to the generalized occurrence of South American tomato pinworm (*Tuta absoluta*), which resulted in such severe losses that processing plants in the northeast region were shut down due to lack of fruit.

Greenhouse crops, which became popular in the 1990s, are another example of monoculture and high plant density. Attracted by high potential yields, such as 200 tons/ha/year of bell pepper or eggplant, greenhouse crops always face a high risk of epidemics and a dependency on fungicide spraying. Inside greenhouses, powdery mildew (*Oidiopsis sicula*) is a major pathogen that requires weekly sprayings.[65]

5.5.6 Tropical and temperate-climate fruits

The majority of the fruit grown in Brazil is consumed domestically. Injured fruit has a reduced market value, and its export is unfeasible. Spots and decay caused by plant pathogens are major constraints to fruit exports, which came to only $100 million in 2000, excluding oranges.[67]

The coldest regions in southern Brazil are the most appropriate to several Rosaceae crops, which are also grown in some areas in the southeast region. The main alien pathogens in Brazil are well known in the Northern Hemisphere and include *X. fastidiosa* (the most important pathogen affecting plums in southern Brazil), *Venturia inaequalis, X. campestris* pv. *pruni, Monilinia fructicola, Taphrina deformans,* and *Tranzchelia discolor.*[68–70] Fungicide sprayings during the fruit development stage and the removal of infected parts during the dormancy stage are essential for disease management and the harvest of marketable fruits.

Leaf and fruit spots affecting tropical fruits are caused mainly by widespread native pathogens. Some alien pathogens may draw attention due to their harmfulness, such as the nematode *Bursaphelenchus cocophilus* and the protozoan *Phytomonas staheli*, both of which affect coconuts.[4,71] First recorded in Brazil in 1981, *P. staheli* is one of the most important pathogens in coconut crops (and also of oil palm and ornamental palms), causing the death of plants it infects. It is a problem in the northern states, such as Amazonas and Pará, where the only effective control measure is the eradication of infected plants.[4]

The grapevine canker (*X. campestris* pv. *viticola*) was detected in 1998, in the submedium São Francisco river valley, an important area for fruit and vegetable crops in the northeast region.[6] Grapevine canker has caused serious damage, especially to Red Globe and seedless grapes, both of which are important exports. In addition to preventive control, 100 ha of diseased plants were removed in 1999.[72]

5.5.7 Oranges

Citrus monoculture has been practiced in the state of São Paulo since the 1930s. Citrus crops now cover 800,000 ha, from the northeast region of São Paulo state to the western part of Minas Gerais state.[73] This region is responsible for 80% of Brazil's orange crop, whose main diseases are the citrus canker (*Xanthomonas axonopodis* pv. *citri*), citrus variegated chlorosis (CVC, *X. fastidiosa*), and black spot (*Guignardia citricarpa*). All of these are classified as A2 pathogens — i.e., of limited distribution and officially under control.

First detected in 1987 in the southeast region, *X. fastidiosa* was disseminated throughout the center-south and northeastern regions of the country by leafhoppers and infected seedlings.[74,75] It is estimated that 20% of the orange trees in the state of São Paulo are infected by *X. fastidiosa*, exhibiting foliate symptoms and producing fruit with no market value, accounting for $130 million in losses.[76] Efforts to control the disease included *X. fastidiosa* genome sequencing, a $15 million project funded by FAPESP (State of São Paulo Research Foundation).

The citrus canker was first recorded in Brazil in 1957. It was probably introduced by infected stems or fruits brought from Southeast Asia. Despite efforts at pathogen eradication since then, *X. axonopodis* pv. *citri* was disseminated throughout the south and the center-west. The intensity of the disease increased in the 1990s, from nine disease foci in 1992 to more than 4000 in 1999, aided by wounds caused by citrus miner larvae (*Phyllocnistis citrella*).[76] The citrus canker was responsible for damaging 98 million boxes of oranges in 1998–99 — a $500 million loss, equivalent to 10% of the production value of the entire crop.[76,77] A promising program of eradication of diseased plants that is being coordinated by the FUNDECITRUS (Fund for Citrus Plant Protection) spent $20 million in 1999 (86% of those funds coming from the private sector) on the removal of 2 million infected trees.[76] Diseased plants and their neighbors within an area of 30 m^2 are removed if disease incidence is below 0.5%. If this threshold is exceeded, all trees in an orchard section delimited by roads must be removed.[77] All nurseries and domestic orchards in the Fundecitrus area are inspected periodically; 145,000 seedlings and 60,000 trees were removed in 2000.[76] The program proved successful; in December 2000 only 67 disease foci were found and were subsequently eradicated.[78]

5.5.8 Bananas

Brazil is the world's second-ranking banana producer, and the number one consumer of the fruit. Nearly all of its crop is consumed within the country, where it serves as an important source of carbohydrates. Bananas also provide income to small farmers throughout the year. Bananas are Brazil's most important fruit and one of the basic foods in the northern region.

Black sigatoka (*Mycosphaerella fijensis*), the most destructive disease in bananas and plantains of the genus *Musa*, was first found in Brazil in 1998 near the Peruvian border.[79] Along with the orange pathogens discussed above, *M. fijensis* completes the alien pathogen group on the Brazilian A2 list. Two years after its detection, it crossed the few regional phytosanitary barriers and could be found in scattered distribution in the Amazon region states, where it probably was disseminated by infected plants and fruits or by conidia from *M. fijensis* anamorphous (*Paracercospora fijensis*), which remain viable for more than two weeks in plastic, tires, wood, and clothes.[23,80]

Regional estimates of yield losses were not carried out, but the number of susceptible genotypes exceeds those affected by common sigatoka, and includes the high-priced cultivars Maçã and Prata, which may experience 100% losses, and cropped plantains (the AAB group, such as Pacovi and Pacovan, may experience losses of up to 60%).[23] If the disease becomes generalized in a region, the area may be forced to import bananas from other states, or to reduce the available variability by switching from traditional susceptible varieties to resistant ones of lower commercial value.

Fungicide sprayings have been recommended for disease control, but the need for as many as 40 annual sprayings increases production costs by $1000/ha/year, beyond the means of the local population. The major measure for controlling black sigatoka has been to restrict the movement of infected fruit and plants into pathogen-free areas.[23,81,82]

Black sigatoka is the leading threat to banana crops in Brazil. In spite of the tentative confinement of the pathogen to the northern region, *M. fijensis* was recently detected in Mato Grosso state in the center-west region. A 755-ha area that produces an annual yield of 6795 tons of bananas was affected by *M. fijensis* in Mato Grosso, and the fruit was barred for market, causing an estimated loss of $680,000.[23] To avoid the spread of black sigatoka to other regions, a $20 million budget was approved by the Ministry of the Agriculture and Food Supply for disease control and prevention.[23]

5.6 *Phytosanitary policies and the spread of pathogens*

Brazilian federal law specifies rigid criteria to prevent the entry of A1 exotic pathogens into the country, and to prevent spread of those listed as A2.[83] However, Brazil does not have a federal law that establishes patterns of seed health regarding tolerance to pathogens that are already dispersed in the country, and this may permit costly damage to crops. These pathogens include *Ascochyta pisi, C. lindemuthianum, C. kikuchii,* and *S. sclerotiorum*. Only a group of potato pathogens, such as the non-indigenous *A. solani, P. infestans*, and *Spongospora subterranea*, are regulated at present and have defined tolerance patterns.[83] A lack of federal regulations helped several non-indigenous pathogens to spread to many regions by means of infected seeds, such as with *C. sojina*,[84] or infected vegetative parts, such as with *Ditylenchus dipsaci*.[85]

With so many known problems caused by seedborne pathogens in Brazil — such as *Claviceps africana* (which causes $800,000 worth of damage to hybrid sorghum seed production fields), soybean stem canker, white mold on Fabaceae, fusariosis on several crops, stalk rot on maize, viruses and bacteria on vegetables — there are enough arguments to suggest the dimensions of the risks taken by the lack of formal seed health control, and the importance of using healthy seeds to control diseases.[86–88] To limit further damage, it is up to the states to make and enforce policies concerning trends in disease control, in the field and on the borders.

No tolerance patterns regarding imported seeds exist for pathogens not listed in the A1 and A2 lists, either. For instance, lettuce seeds, which can only be marketed in the United States if they present 0% of lettuce mosaic virus in a 30,000-seed lot, can be legally marketed in Brazil with no restriction.[89] Statistics are not updated, but 5695 tons of imported vegetable seeds, representing 30 to 69% of the country's needs, entered the country between 1981 and 1989.[90] The prevalence of imported seeds for bell peppers and tomato hybrids illustrates Brazil's dependence on imported vegetable seeds and highlights the risk of introducing new pathogens that can exacerbate boom-and-bust agricultural cycles.[91]

Considerable intraspecific variability of several non-indigenous pathogens can be found in the country, such as the nine races of *C. lindemuthianum*.[92] Other pathogens can present as many as 20 (*Cercospora sojina*) or 30 (*Uromyces appendiculatus*) races,[84,93] restricting a farmer's crop options.[94,95] The pathogen diversity and high levels of inoculum in many fields have reduced the viability of resistant varieties. Under these conditions, the resistance of common bean varieties to anthracnose, or soybean varieties to stem canker breaks down within 5 years, keeping plant breeding programs and germplasm banks busy producing new varieties and hybrids.[54,96] The worst situation can be found on rice crops: just 2 years are necessary to break down the resistance to leaf blast on newly released cultivars, due to the presence of many *P. oryzae* pathotypes in the country.[58]

The need for defining seed health-tolerance patterns has been exhaustively discussed in Brazilian meetings and seminars during the past 20 years.[88] A longstanding request of plant pathologists, the need for federal regulation of tolerance patterns in seeds, was begun in 1999,[87] when guidelines for seed treatment and for tolerance patterns for 32 pathogens found on 13 crops were proposed (Table 5.3), due to these pathogens' high potential for crop destruction.

Nevertheless, the proposal has been questioned by some sectors of the seed industry.[87] Fearful of not achieving rigid seed health and disease tolerance patterns, these sectors have held back the development of the appropriate legislation. Meanwhile, hybrid or

Table 5.3 Destructive Plant Pathogens Recorded in Brazil and Their Respective Hosts, Submitted for the Definition of Official Tolerance Patterns on Seeds by the Ministry of Agriculture and Food Supply

Pathogen	Host
Colletotrichum gossypii var. *cephalosporioides, Fusarium oxysporum* f. sp. *vasinfectum*	Cotton
Drechslera oryzae, Pyricularia oryzae	Rice
Colletotrichum lindemuthianum, Fusarium oxysporum f. sp. *phaseoli, Sclerotinia sclerotiorum, Xanthomonas campestris* pv. *phaseoli*	Common bean
Diplodia maydis	Maize
Colletotrichum graminicola, Claviceps africana	Sorghum
Colletotrichum truncatum, Sclerotinia sclerotiorum	Soybean
Bipolaris sorokiniana, Drechslera tritici-repentis, Pyricularia grisea, Stagonospora nodorum, Tilletia caries, Tilletia foetida, Xanthomonas campestris pv. *undulosa*	Wheat
Alternaria helianthi, Alternaria zinniae, Sclerotinia sclerotiorum	Sunflower
Clavibacter michiganensis subsp. *michiganensis, Xanthomonas campestris* pv. *vesicatoria*	Tomato
Lettuce mosaic virus	Lettuce

open-pollination seeds continue to be produced in areas with high levels of inoculum, and as a result, losses of 100% have been registered due to infected seeds, such as with cotton crops, recently introduced in the center-west region.[11]

5.7 *Chemical control and its impact*

Alien pathogens are responsible for almost all fungicide and antibiotics sales in Brazilian agriculture, since the chemical control of native pathogens, such as *Crinipellis perniciosa, Microcyclus uley,* and *Ralstonia solanacearum,* is not economically viable. With the omnipresence of alien pathogens and changes in several agricultural production systems, the amount of pesticides used to control plant diseases has increased markedly over the past several decades. From 1964 to 1991, when Brazil's cropped area increased by 76%, the national pesticide market grew by 276.2%.[98] With its generalized dependency on agrochemicals, Brazil joined the world's top five pesticide consumers, purchasing 2557 tons in 1998.[99] Disparities among Brazilian regions are reflected in the consumption of fungicides: the southeastern and southern regions consume more than 72% of the total, and the state of São Paulo is responsible for a third of the nation's consumption. In contrast, the poorest states, in the northeastern and northern regions, are responsible for less than 2%.[99]

From 1989 through 1998 the amount of fungicide sold in Brazil grew 295%, from 147.45 tons to 436.23 tons (Figure 5.2), accounting for 17% of pesticides sold in Brazil in 1998.[99] The amount of fungicide sold decreased in 1999, partly because of a decline in the value of the Brazilian real in relation to the dollar,* and partly because of low prices in the international market for such commodities as soybeans, sugarcane, coffee, and citrus. The low prices of commodities led farmers to cut production costs, and as fungicides became more expensive with the real's weakness, fewer sprayings were applied.[100]

Knowing the risks and consequences of pathogen transmission through infected seeds, many farmers chemically treated their seeds in the 1990s. Using a minimal amount of fungicides at an increase on soybean production costs of 0.5%, 0.06% for maize, and 0.17%

* One Brazilian real (R$) was rated at US$1.00 from 1994 to December 1998. From January 1999 to January 2001, US$1.00 has corresponded to an average rate of R$1.80–2.00.

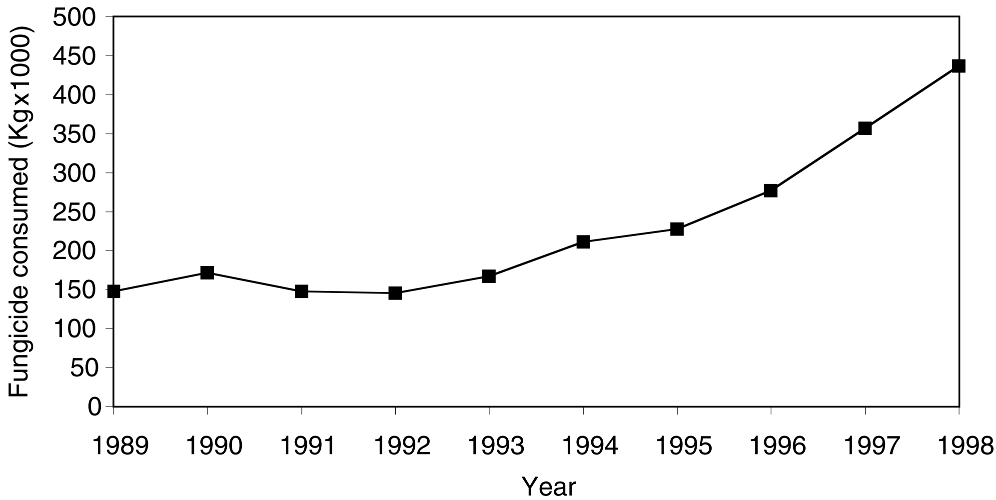

Figure 5.2 Fungicide consumption in Brazil from 1989 to 1998, according to ANDEF (National Association for Plant Protection).[99]

for cotton, farmers have successfully controlled many pathogens and reduced the need for field spraying.[101–103]

In the field, chemical control of pathosystems has achieved economically acceptable results, and now plays an important role in disease management for small grain crops, especially in the southern region, providing enough advantages to cover the 18% share of the production cost that spraying represents.[101–106]

For other crops, the difficulty of accurately diagnosing plant diseases, plus the uncertainty of payback thresholds and insufficient technical support from the rural extension service, have resulted in a lack of criteria regarding the use of fungicides.[24,107] Often, growers do not let up on fungicide sprayings, even when the sprayings are not necessary, and sometimes non-registered fungicides have been used in cases of wrong diagnosis, such as benomyl sprayings applied in attempts to control pathogens such as *Xanthomonas fragariae* on strawberry fields.[108] The lack of information about pesticide deposition on leaves and the use of inappropriate spray nozzles also contribute to misguided sprayings, often at high dip volumes. As a result, the runoff of pesticides in common bean and tomato plants was found to be 49–88% and 44–70%, respectively, of the total sprayed amount; the runoff drips onto the soil and reaches non-target organisms.[109]

Chemical control has been used largely to counter low resistance to diseases, such as with potato and tomato crops, where the use of fungicides increased 46.3% and 41.2%, respectively, from 1984 to 1990, even with new products that are sprayed at lower dosages.[90] On average, 15 to 30 fungicide sprayings have been used to control diseases caused by alien pathogens such as *A. solani* and *P. infestans*, accounting for 30% of crop production costs. The high spraying frequency has been justified by the high market value of these crops, especially such varieties as Bintje potatoes, which are preventively sprayed in 3- to 5-day intervals, or even daily during weather that is conducive to late blight infection.[110]

In 1990, potato and tomato crops were responsible for 41% of the sales of fungicides in Brazil.[91] These crops are usually planted close to rivers, on the outskirts of urban areas, and the frequent sprayings pose a risk of environmental pollution. After the solanaceous, the citrus and coffee crops are the next largest consumers of fungicides, accounting for 19.6% and 11.3% of all fungicides sold, respectively. Because they represent a large cropped area, spraying of citrus and coffee crops also presents a high risk of environmental pollution.

In the center-west and southeastern center-pivot-irrigated areas, 200,000 ha are cropped to common beans during the dry and cold season, from April to September, when on average three fungicide sprayings are done, at a cost of $90/ha, representing a 12% share of the crop production cost.[111] About 100,000 ha of these crops are sprayed with irrigation water.[112] Although "chemigation," as it is called, is a forbidden practice, and there is no registered fungicide or nematicide that is approved for spraying in irrigation water, it is commonly practiced by growers.[113]

Careless farmers have used highly diluted solutions for the control of foliar diseases, thereby reducing fungicide efficiency since the chemicals then reach the soil instead of staying on the leaves,[114,115] although chemigation may also address the control of soilborne pathogens, such as *S. sclerotiorum*. In contrast, dosages of up to 250% above the recommended level have been used in chemigation,[116] increasing the risks of environmental damage and inducing resistance to fungicides. Efficiencies that can be achieved with the correct use of fungicides through irrigation water have been evidenced by the research sector, but chemigation is done without following recommended criteria and poses a threat to the environment.[112,114,117]

In general, fungicide residues have been below the tolerance limits, except on vegetables and fruits. Residues were found in 63% of the vegetables marketed in Rio de Janeiro state, and 24% of these samples exceeded tolerance levels by as much as 50%.[118] In the city of São Paulo, 15% of the samples examined from 1994 to 1998 had residues of unregistered fungicides, which because of their status do not have official tolerance patterns.[119] Residues remaining on pesticide containers have also been a major problem. About 0.3% of the contents remain as residue on empty containers, amounting to 76 tons of fungicides in 2000. Despite guidelines and incentives aimed at the proper storage and recycling of empty containers, these guidelines are frequently ignored.[99,120]

The emergence of fungicide-resistant isolates is another consequence of arbitrary sprayings, and was recorded especially for benomyl, iprodione, triadimenol, and metalaxyl.[121–126] Resistance to antibiotics, despite their sporadic use, was also detected.[127]

Cases of pesticide poisoning are seldom diagnosed by medical doctors and are treated improperly at home. The Brazilian Ministry of Health recorded 4135 poisoning cases from agricultural pesticides in 1999, and the World Health Organization estimates that each reported case of toxification corresponds to another 50 that remain undetected.[128,129] Considering that fungicides correspond to 17% of the pesticides sold in Brazil, it is estimated that they caused 35,000 cases of intoxication in 1999.

On small farms, workers generally do not receive appropriate instructions for the handling of toxic compounds, nor do they understand that pesticides can be absorbed in lethal amounts through the skin, as contact with the substances does not sting or burn. Because of the hot and humid weather in almost all of Brazil, many farmers hesitate to use protection equipment and remain exposed to poisoning, of which 80 to 99% of all cases are caused by the skin contact.[130]

5.8 Trends and needs in Brazilian agriculture

To satisfy the food needs of the growing population, and to keep export growth in line with forecasts, Brazil must take a great interest in preventing the entrance of new alien pathogens.[131] The current challenge to the Brazilian scientific community is to develop efficient methods of pathogen detection, as well as to improve the control and management of alien pathogens already established in the country. Government and growers are partners in this process, interacting with the research sector in the search for better solutions.

Efforts to minimize disease intensities and yield losses have helped refine control measures; a combination of disease resistance, improved cultivation techniques, and chemical control has yielded the best results. In some instances cultivation practices are used as pathogen control measures, such as no-tillage systems for *S. sclerotiorum* and *H. glycines*.[31,44,132]

The risk of alien pathogen introduction is always present, and it will no doubt rise as free trade grows between Brazil and its neighbor countries that are partners in the MERCOSUL.* Under the coordination of COSAVE (South Cone Plant Protection Committee), a common list of A1 and A2 quarantined pathogens was approved in 1996; the list is currently being updated in order to determine tolerance patterns that will support the trade of agricultural commodities.[133]

The intrinsic and extrinsic impact (on other sectors besides agriculture) of excessive use of fungicides can be measured by the increased demand for organic foods,[98] a market that has grown 10% annually. At the same time as the general occurrence of pathogens makes the transition from traditional cropping systems to organic systems more challenging, the expanding organic foods market makes it important to reduce dependence on chemical inputs.

The agricultural sustainability and competitive advantage of agribusiness are assured as Brazilian agriculture moves toward integrated disease management.[31,33,132,134] Disease management on an ecological basis requires biological knowledge of targeted pathogens. Although several methods are available for the control of many diseases, the knowledge of each pathosystem is crucial, and such knowledge takes time to acquire for both newly introduced and established pathogens. Improvements in the nation's plant inspection system and the implementation of laws that regulate plant transport and marketing will be necessary to complement Brazil's plant defense requirements.

Acknowledgments

I wish to thank Adalberto C. Café Filho (Universidade de Brasília), Aloísio Sartorato (Embrapa Arroz e Feijão), Carlos A. Casela (Embrapa Milho e Sorgo), Edson A. Pozza (Universidade Federal de Lavras), Luiz C.B. Nasser (Embrapa Cerrados), João Carlos Fazuoli and Marcos Machado (Instituto Agronômico de Campinas), Mirtes F. Lima (Embrapa Trópico Semi-Árido), Silvana V. de Paula (Embrapa Recursos Genéticos e Biotecnologia), and Leonardo B. Giordano and Cláudia S.C. Silva (Embrapa Hortaliças), for the information they kindly provided. Special thanks go to Jamil Macedo (Embrapa's Secretariat for International Cooperation), Steven Garcia (Universidade de Brasília), and André N. Dusi, Carlos A. Lopes, Gilmar P. Henz, and Leonardo Boiteux (Embrapa Hortaliças) for revisions and criticism that greatly improved this chapter.

References

1. Mendes, M.A. S., et al., *Fungos em plantas no Brasil*, Embrapa/SPI-Embrapa/Cenargen, Brasília, 1998, 556.
2. Kitajima, E.W., Ávila, A.C., and Resende, R.O., Taxonomia de vírus de plantas, *Fitopatol. Bras.*, 22, 5, 1997.
3. Manso, E.C., et al., *Catálogo de Nematóides Fitoparasitos Encontrados Asociados a Diferentes Tipos de Plantas no Brasi*l, Embrapa/SPI, 1994, 448.
4. Bedendo, I.P., Protozoários, in *Manual de Fitopatologia Princípios e Conceitos*, 1, Bergamin Filho, A., Kimati, H., and Amorin, L., Eds, 3rd ed., Ceres, São Paulo, 1995, 161.

* South American Common Market, which includes Brazil, Argentina, Paraguay, and Uruguay. Chile is also a partner on plant health issues. These five integrate the South Cone countries.

5. Marques, A.D. A., et al., A quarentena e o intercâmbio vegetal no Brasil: a atuação do CENARGEN, *Pesq. Agrop. Bras*., 30, 143, 1995.
6. Malavolta Jr., V., Almeida, I.M.G., and Sugimori, M.H., Ocorrência de *Xanthomonas campestris* pv. *viticola* em videira no Brasil, *Summa Phytopathol*., 25, 262, 1999.
7. Fábrega, A.R., et al., Effect of *Diaporthe phaseolorum* var. *sojae* infection on seed quality of soybean genotypes, *Fitopatol. Bras*., 25, 624, 2000.
8. Luz, W.C., Diagnose e controle das doenças de espiga de milho no Brasil, Embrapa/CNPT, Passo Fundo, 1995, 28.
9. Meneguzzo, J., et al., Efeito da podridão do cacho na composição do mosto de *Vitis vinifera* cv. Gewurztraminer, *Fitopatol. Bras*., 24, 45, 1999.
10. Yorinori, J.T., et al., Doenças da soja e seu controle, in *Cultura da soja nos Cerrados*, in Arantes, N.E. and Souza, P.I.M., Eds., POTAFOS, Piracicaba, 1993, 333.
11. Nasser, L.C.B., Importância da semente sadia, *Cultivar,* 12, 54, 2000.
12. Zadoks, J.C., The concept of thresholds: warning, action and damage thresholds, in *Crop Loss Assessment and Pest Management*, Teng, P.S., Ed., APS Press, St. Paul, 1987, 168.
13. Kimati, H., et al., Eds., *Manual de Fitopatologia, Doenças das Plantas Cultivadas,* 2, Ceres, Piracicaba, 1997, 774.
14. Menten, J.O.M., Prejuízos causados por patógenos associados às sementes, in *Patógenos em sementes: detecção, danos e controle químico*, Menten, J.O.M., Ed., ESALQ/FEALQ, Piracicaba, 1991, 115.
15. Lopes, C.A. and Ventura, J.A., Detecção e caracterização do olho-preto (*Fusarium solani*), uma nova ameaça à bataticultura brasileira, *Fitopatol. Bras*., 21, 513, 1996.
16. Bergamin Filho A. and Kimati, H., Importância das doenças de plantas, in *Manual de Fitopatologia, Princípios e Conceitos,* 1, Bergamin Filho, A., Kimati, H., and Amorin, L., Eds., 3rd ed., Ceres, São Paulo, 1995, 13.
17. Alves, E.R.A., Lopes, M., and Contini, E., O empobrecimento da agricultura Brasileira, *Revista de Política Agrícola*, 8, 5, 1999.
18. IBGE, *IBGE's website*, http://www.ibge.gov.br, Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro, 15/01/2001.
19. Schumann, G.L., *Plant Diseases: Their Biology and Social Impact*, APS Press, St. Paul, 1991, 335.
20. Echandi, E., et al., *Crop protection in Brazil, Uruguay Bolívia, Ecuador and Dominican Republic*, USAID/University of California, Berkeley, 1972, 67.
21. Miyasaka, S. and Medina, J.C., Eds., *A soja no Brasil*, Campinas, ITAL, Campinas, 1981, 1062.
22. Nunes, E.P. and Contini, E., Dimensão do complexo agroindustrial brasileiro, *Agroanalysis*, Rio de Janeiro, Fundação Getúlio Vargas, 12, 54 2000.
23. Silva, J.R. (Coord.) *Caracterização da Sigatoka Negra* (*Mycosphaerella fijensis* Morelet) como Praga Quarentenária A2, Ministério da Agricultura e do Abastecimento, Brasília, 1999.
24. Olinger, G., Ascensão e decadência da extensão rural no Brasil, EPAGRI, Florianópolis, 1996, 523.
25. Roberto, S.R., et al., Guerra às pragas, *Agroanalysis*, 6, 58, 2000.
26. Sawakazi, E., et al., Reação de cultivares de milho à mancha de Phaeosphaeria no estado de São Paulo, *Pesq. Agrop. Bras.*, 32, 585, 1997.
27. Paradela Filho, O., et al., Primeira constatação em cafeeiro no Brasil da *Xylella fastidiosa* causadora da clorose variegada dos citros, *Laranja*, 16, 135, 1995.
28. Carvalho, V.L. and Chalfoun, S.M., *Doenças do cafeeiro. Diagnose e controle*, Boletim Técnico 58, Belo Horizonte, EPAMIG, 2000, 44.
29. Vieira, C., *Doenças e pragas do feijoeiro*, UFV, Viçosa, 1988, 231.
30. Lobo Jr., M. Lopes, C.A., and Silva, W.L.C., Sclerotinia rot losses in processing tomatoes grown under centre pivot irrigation in central Brazil, *Plant Pathol*., 49, 51, 1999.
31. Café Filho, A.C. and Lobo Jr., M., Manejo de fatores físicos e culturais para controle de patógenos de solo, *Revisão Anual de Patologia de Plantas,* 8, 267, 2000.
32. Reis, E.M., et al., Grain losses caused by the infection of wheat heads by *Giberella zeae* in southern Brazil, from 1984 to 1994, *Summa Phytopathol*., 22, 123, 1996.
33. Reis, E.M., et al., Efeito de práticas culturais na severidade de manchas foliares do trigo e sua relação com a incidência de fungos patogênicos na semente colhida, *Fitopatol. Bras*., 22, 407, 1997.

34. Lopes, C.A., Ocorrência de doenças em plantas sob cultivo irrigado — plasticultura, *Summa Phytopathol.*, 22, 81, 1996.
35. Flores, M.X. and Nascimento, J.C., *Desenvolvimento sustentável e competitividade na agricultura brasileira,* EMBRAPA/SEA, Brasília, 1992, 30.
36. Bergamin Filho, A. and Amorin, L., *Doenças de Plantas Tropicais: Epidemiologia e Controle Econômico, Ceres,* Piracicaba, 1996, 299.
37. Coffee Business, *Anuário Estatístico do Café,* 3rd ed., Coffee Business, Rio de Janeiro, 1997, 107.
38. Zambolim, L., et al., Café (*Coffea arabica* L.), in *Controle de doenças de plantas*, 1, Vale, F.X. R. and Zambolim, L., Eds., Ministério da Agricultura e do Abastecimento, Brasília, 1997, 83.
39. Yorinori, J.T., Detecção do agente causador do cancro da haste em sementes de soja, in *Patógenos em Sementes: Detecção, Danos e Controle Químico*, Menten, J.O.M., Ed., ESALQ/FEALQ, Piracicaba, 1991, 109.
40. EMBRAPA, Soja: recomendações técnicas para o Mato Grosso do Sul e Mato Grosso, Embrapa Agropecuária Oeste, Dourados, 1999, 158.
41. Lima, R.D., Ferraz, S., and Santos, J.M., Ocorrência de *Heterodera* sp. em soja no Triângulo Mineiro, in *Congr. Bras. Nemat.*, Lavras, Sociedade Brasileira de Nematologia, 1992, 82.
42. Mendes, M.L. and Dickson, D.W., Detection of *Heterodera glycines* in Brazil, *Plant Dis.* 77, 499, 1993.
43. Yorinori, J.T., Principais doenças da soja no Brasil, *Summa Phytopathol.*, 21, 70, 1995.
44. Yorinori, J.T., Galerani, P.R., and Garcia, A., *Manejo da cultura para o controle do nematóide da soja,* Embrapa/CNPSo, Londrina, Documentos 83, 26, 1994.
45. Pereira, O.A.P., Análise da situação atual de doenças de milho no Brasil e disponibilidade de germoplasma resistente, *Summa Phytopathol.*, 21, 67, 1995.
46. Fernandes, F.T., The current status of maize diseases in Brazil, in *Diagnosing maize diseases in Latin America,* Brasilia/Ithaca, Casela, C.R., Renfro, B., and Krattiger, A.E., Eds., ISAAA/EMBRAPA, 1998, 5.
47. Brasil, E.M. and Carvalho, Y., Comportamento de híbridos de milho em relação a *Phaeosphaeria maydis* em diferentes épocas de plantio, *Pesq. Agrop. Bras.*, 33, 1977, 1998.
48. Oliveira, E., Manejo de doenças causadas por patógenos transmitidos por insetos, *Fitopatol. Bras.*, 22 (S), 229, 1997.
49. Waquil, J.M., et al., Efeito na produção e incidência de viroses em híbridos comerciais de milho, *Fitopatol. Bras.*, 21, 460, 1996.
50. Fernandes, F.T. and Oliveira, E., A ascensão de uma doença, *Cultivar*, 6, 14, 2000.
51. Rava, C.A. and Sartorato, A., Eds., *Principais doenças do feijoeiro comum e seu controle*, EMBRAPA SPI/CNPAF, Brasília, 1994, 300.
52. Saccá, A.R., Algumas moléstias novas e outras pouco conhecidas da batatinha (*Solanum tuberosum), Bol. Secret. Agr. Ind. Com. do Est. de São Paulo*, 21, 741, 1920.
53. Chaves, G.M., Estudos sobre *Sclerotinia sclerotiorum* (Lib.) De Bary, *Experimentiae*, 4, 1, 1964.
54. Nasser, L.C. B., personal communication, 2001.
55. Mitsuda, T. and Charchar, M.J.D., Mofo branco do feijoeiro (*Sclerotinia sclerotiorum*) nos Cerrados: estudos recentes e métodos de controle, in *Semin. Progr. Pesq. Agríc. Cerrados*, EMBRAPA/CPAC/EMPRA/JICA, Brasília, 1992, 75.
56. Lobo Jr., M., personal communication, 2001.
57. Arancibia Carvajal, R., *Dinâmica populacional em campo, agressividade, detecção em sementes de feijão e determinação dos efeitos de temperatura e herbicidas na germinação carpogênica de Sclerotinia sclerotiorum* (Lib.) de Bary, M.Sc. Dissertation, Universidade de Brasília, Brasília, 1999, 103.
58. Prabhu, A.S., Filippi, M.C., and Ribeiro, A.S., Doencas e seu controle, in *A Cultura do Arroz no Brasil*, in Vieira, N.R.A., Santos, A.B., and Sant'ana, E.P., Eds., Embrapa Arroz e Feijão, Santo Antonio de Goiás, 1999, 262.
59. Anjos, J.R.N., et al., Ocorrência de brusone (*Pyricularia grisea*) em trigo e centeio na região dos Cerrados do Brasil Central, *Pesq. Agrop. Bras*., 31, 79, 1996.
60. Reis, E.M., Medeiros, C.A., and Casa, R.T., Control of leaf blights of wheat by elimination of the inoculum source, *Proc. Helmintho sp. Blights of Wheat: Spot Blotch, Leaf Blight and Leaf Blotch*, El Batan, CIMMYT/UCL/BADC, 1997, 327.
61. Lopes, C.A. and Buso, J.A., *Cultivo da Batata* (Solanum tuberosum L.), Instruções Técnicas da Embrapa Hortaliças, 11, Embrapa Hortaliças, Brasília, 1997, 36.

62. Melo, P.C.T., Agroindústria de tomate em expansão, *Hort. Bras*., 17, cover note, 1999.
63. Zambolim, L., Vale, F.X.R., and Costa, H., Eds., *Controle de Doenças de Plantas*. Hortaliças, 1, Universidade Federal de Viçosa, Viçosa, 2000, 444.
64. Zambolim, L., Vale, F.X.R., and Costa, H., Eds., *Controle de Doenças de Plantas*. *Hortaliças*, 2, Universidade Federal de Viçosa, Viçosa, 2000, 879.
65. Kurosawa, C., Ocorrência de *Oidiopsis taurica* (Lev) Salmon em tomateiro, *Summa Phytopathol.*, 13, 20, 1987.
66. Faria, J.C., et al., Situação atual das geminiviroses no Brasil, *Fitopatol. Bras.*, 25, 125, 2000.
67. FNP Consultoria & Comercio, *Agrianual 2001. Anuário da Agricultura Brasileir*a, FNP Consultoria & Comercio, São Paulo, 2000, 545.
68. Leite, R.M., et al., Hospedeiros alternativos de *Xylella fastidiosa* entre plantas invasoras de pomares de ameixeira com escaldadura da folha, *Fitopatol. Bras.*, 22, 54, 1997.
69. Bleicher, J., Doenças em rosáceas de caroço, in *Manual de Fitopatologia, Doenças das Plantas Cultivadas*, vol.2., Kimati et al., Eds., 3rd ed., Ceres, São Paulo, 1995, 621.
70. Boneti, J.I.S. and Katsurayama, Y., A sarna da macieira, *Corr. Agric.*, 2, 11, 1995.
71. Warwick, D.R.N., Influência dos fitopatógenos na cadeia produtiva do coqueiro, *Fitopatol. Bras*., 25 (S), 296, 2000.
72. Lima, M.F., personal communication, 2001.
73. Feitchenberger, E., Manejo integrado das principais doenças dos citros no Brasil. *Fitopatol. Bras*., 25 (S), 288, 2000.
74. Laranjeira, F.F., et al., Constatação da Clorose Variegada dos Citros em Sergipe, *Fitopatol. Bras*. 21 (S), 521, 1996.
75. Tubelis, A., et al., Difusão da Clorose Variegada dos Citros em pomares comerciais de laranja no Brasil, *Laranja*, 14, 239, 1993.
76. FUNDECITRUS, *Fundecitrus website*, http://www.fundecitrus.com.br, Fundo de Defesa da Citricultura, 19/01/2001.
77. Siqueira, A.S., Producing for the world, *Brazil Now* (special edition), 14, 2000.
78. GCONCI, *GCONCI website*, http://www.gconci.com.br, Grupo de Consultores em Citros, 19/12/2000.
79. Pereira, J.C.R., et al., Ocorrência da Sigatoka Negra no Brasil, *Fitopatol. Bras.,* 23 (S), 477, 1998.
80. Hanada, R.E., Gasparotto, L., and Pereira, J.C.R., Sobrevivência de conídios de *Mycosphaerella fijensis* em diferentes materiais, *Fitopatol. Bras*., 25 (S), 380, 2000.
81. Pereira, J.C.R., Gasparotto, L., and Cordeiro, Z.J.M., Ataque às folhas, *Cultivar HF,* 1, 34, 2000.
82. Pereira, J.C.R., et al., *Sigatoka Negra*, Embrapa Amazônia Ocidental/CESV-AM/DFA-AM/IDAM, 1998.
83. Brasil, Ministério da Agricultura, Instrução Normativa 38, *Diário Oficial da União*, October 14, 1999.
84. Yorinori, J.T. and Klingelfuss, L.H., Novas raças de *Cercospora sojina* em soja, *Fitopatol. Bras.*, 24, 509, 1999.
85. Costa, I.A., Ocorrência de *Ditylenchus dipsaci* em alho-semente na região de Curitibanos, in *Proc. Simp. Bras. Ditylenchus dipsaci,* 1, Sociedade de Olericultura do Brasil, Itajaí, 1983, 9.
86. Embrapa, *Global conference on ergot of sorghum*, Embrapa Milho e Sorgo/INTSORMIL, Sete Lagoas, 1999, 208.
87. Machado, J.C., Nasser, L.C.B., and Menten, J.O.M., Patologia de sementes, *Cultivar,* 1, 62, 1999.
88. Nasser, L.C.B., Machado, J.C., and Menten, J.O.M., *Jornal do Confea*, May, Brasília, 1999, 14.
89. Pierobom, C.R., Médico ou monstro, *Seed News,* 1, 14, 2000
90. Nascimento, W.M., Situação da produção e importação de sementes de hortaliças no Brasil, in *Encontro Sobre Produção e Qualidade de Sementes de Hortaliças*, in Nascimento, W.M. and Pessoa, H.B.S.V., Eds., CNPH/EMBRAPA, Brasília, 1991, 1.
91. Maringoni, A.C., Virulência de isolados de *Xanthomonas campestris* pv. *phaseoli*, *Pesq. Agrop. Bras*., 33, 861, 1998.
92. Carbonell, S.A., et al., Raças fisiológicas de *Colletotrichum lindemuthianum* e reação de cultivares e linhagens de feijoeiro no estado de São Paulo, *Fitopatol. Bras*., 24, 60, 1999.
93. Santos, S.C. and Rios, G.P., Identificação de raças fisiológicas de *Uromyces appendiculatus* nos Estados de Goiás, Rio Grande do Sul e de Santa Catarina, *Fitopatol. Bras.*, 25, 607, 2000.

94. Bonatto, E.R., Soja que resiste, *Cultivar*, 13, 16, 2000.
95. Castro, M.E.A., et al., Variabilidade patogênica de *Alternaria solani,* agente causal da pinta-preta do tomateiro, *Summa Phytopathol*., 26, 24, 2000.
96. Canteri, M.G., Jaccoud Filho, D.S., and Molin, R., Ataque por cima, *Cultivar*, 12, 16, 2000.
97. Brasil, Ministério da Agricultura, Portaria 71, *Diário Oficial da União,* Seção 1/9, February 23, Brasília, 1999.
98. Spadotto, C.A. and Bettiol, W., Uso de fungicidas e potencial de contaminação ambiental no Brasil, *Summa Phytopathol.*, 23, 195, 1997.
99. ANDEF, Vendas de 1988 a 1998, http://www.andef.com.br/Dentro/banc_te3.htm, Associação Nacional de Defesa Vegetal, São Paulo, 15/01/2001.
100. Ahlgrimm, P., Perdendo mercado, *Agroanalysis*, 9, 22, 2000.
101. Machado, J.C., *Tratamento de sementes no controle de doenças*, LAPS/UFLA/FAEPE, Lavras, 2000, 138.
102. Goulart, A.C.P. and Melo Filho, G.A.M., *Quanto custa tratar as sementes de soja, milho e algodão com fungicidas*?, Embrapa Agropecuária Oeste, Dourados, Documentos, 11, 2000, 23.
103. Goulart, A.C.P., Tratamento de sementes de soja com fungicidas para o controle de patógenos, *Fitopatol. Bras.*, 23, 137, 1998.
104. Goulart, A.C.P., et al., Controle de doenças da parte aérea do trigo pela aplicação de fungicidas, *Summa Phytopathol.*, 24, 160, 1998.
105. Picinini, E.C., et al., Impacto econômico do uso do fungicida propiconazole na cultura do trigo, *Fitopatol. Bras.*, 21, 362, 1996.
106. Reis, E.M., Contribuição de defensivos agrícolas na produção de plantas, *Summa Phytopathol.*, 21, 80, 1995.
107. Torres, G., Doenças de hortaliças, o tratamento adequado, *Informe Agropec.*, 17, 3, 1995.
108. Furlanetto, C., et al., Doenças do morangueiro e aspectos da produção no Distrito Federal, *Hort. Bras.*, 14, 218, 1996.
109. Chaim, A., et al., Avaliação de perdas de pulverização em culturas de feijão e de tomate, EMBRAPA/CNPMA, Jaguariúna, 1999, 28.
110. Nazareno, N.X.R., et al., Controle da requeima da batata através do monitoramento de variáveis climáticas, *Fitopatol. Bras.*, 24, 170, 1999.
111. EMBRAPA Base de dados da socioeconomia da Embrapa Arroz e Feijão, Embrapa Arroz e Feijão, Goiânia, 2001.
112. Sartorato, A. and Rava, C.R., Controle químico da mancha angular do feijoeiro comum com aplicação de fungicidas via pivô central, *Summa Phytopathol.*, 24, 253, 1998.
113. Dolabella, R.H.C., Aspectos ambientais e agronômicos da agricultura irrigada na bacia do rio Jardim, Distrito Federal, in *Proc. Int. Symp. on Tropical Savannas. Biodiversidade e Produção Sustentável de Alimentos e Fibras nos Trópicos*, 1, Pereira, R.C., and Nasser, L.C.B., Eds., Embrapa/CPAC, Brasília, 1996, 249.
114. EMBRAPA, *Pesquisa condena uso de fungicida via pivô*, Noticiário n° 019, Embrapa/CPAC, Planaltina, 2, 1991.
115. Vieira, R.F., Introdução à quimigação, in Quimigação. Aplicação de Produtos Químicos e Biológicos via Irrigação, Costa, E.F., Vieira, R.F., and Viana, P.A., Eds., EMBRAPA/SPI, Brasília, 1994, 13.
116. Oliveira, S.H.F., et al., Efeito de fungicidas sobre microrganismos não alvo, *Summa Phytopathol.*, 19, 62, 1993.
117. Oliveira, S.H.F., Recco, C.A., and Oliveira, D.A., Avaliação comparativa da fungicação e aplicação convencional de fungicidas no controle de *Sclerotinia sclerotiorum, Summa Phytopathol.*, 21, 249, 1995.
118. Reis, M.R.C.S. and Caldas, L.Q.A., Dithiocarbamate residues found on vegetables and fruit marketed in the State of Rio de Janeiro, Brazil, *Ciência e Cultura*, 43, 216, 1991.
119. Gorenstein, O., Uma abordagem sobre resíduos de agrotóxicos em alimentos frescos, *Inform. Econôm.,* 30, 3, 37, 2000.
120. Alencar, J.A., et al., Descarte de embalagens de agrotóxicos, *Pesticidas*, 8, 9, 1998.
121. Ghini, R., Ocorrência de resistência em linhagens de *Botrytis cinerea*, no estado de São Paulo, *Fitopatol. Bras.*, 21, 285, 1996.

122. Ghini, R., Ocorrência e sensibilidade colateral de linhagens de *Botrytis cinerea* resistentes a benzimidazóis em berinjela, em Registro, SP, *Summa Phytopathol*., 16, 36, 1990.
123. Fancelli, M.I. and Kimati, H., Ocorrência de linhagens de *Alternaria dauci* resistentes ao fungicida iprodione no estado de São Paulo, *Summa Phytopathol*., 14, 51, 1988.
124. Reis, E.M., et al., Sensibilidade de *Drechslera teres* ao fungicida triadimenol usado em tratamento de sementes de cevada, *Fitopatol. Bras*., 22, 539, 1997.
125. Ghini, R. and Kimati, H., *Resistência de fungos a fungicidas*, Embrapa Meio Ambiente, Jaguariúna, 2000, 78.
126. Nogueira, E.M.C., Toledo, A.C.D., and Sordi, I.M.P., *Plasmopora viticola* resistente ao metalaxyl no estado de São Paulo, *Fitopatol. Bras.* 13, 103, 1988.
127. Silva, V.L.S. and Lopes, C.A., Populações epifíticas de *Pseudomonas syringae* pv. *tomato* em cultivo comercial de tomateiro industrial, *Fitopatol Bras.*, 20, 179, 1995.
128. FIOCRUZ, Ministério da Saúde, http://www.fiocruz.br/cict/sinitox/apresent.htm, Fundação Osvaldo Cruz/Sistema Nacional de Informações Tóxico-Farmacológicas, Rio de Janeiro, 20/12/2000.
129. World Health Organization, WHO website, http://www.who.org, Geneva, 20/12/2000.
130. Camargo Filho, W.P., *Artigos sobre defensivos agrícolas (agrotóxicos)*, São Paulo, Instituto de Economia Agrícola, 1986, 64.
131. Quirino, T.R., Rodrigues, G.S., and Irias, L.J.M., *Ambiente, sustentabilidade e pesquisa: tendências da agricultura brasileira até 2005*. EMBRAPA/CNPTIA, Jaguariúna, 1997, 21.
132. Hall, R. and Nasser, L.C.B., Practice and precept in cultural management of bean diseases, *Can. J. Plant Path.*, 18, 176, 1996.
133. COSAVE, Comitê de Sanidade Vegetal do Cone Sul, *Quarantine Pest Data Sheets*, http://www.cosave.org.py/listafichasing.htm, 10/01/2001.
134. Ventura, J.A., Manejo integrado de doenças de plantas, *Fitopatol. Bras.*, 24, 226, 1999

section four

British Isles

chapter six

Alien plants in the British Isles

Mark Williamson

Contents

6.1 Introduction

There has only been one attempt[49] to estimate the cost, species by species, of a large set of native and introduced plants in the British Isles. That attempt was based primarily on the cost of herbicides and is very useful as far as it goes, but clearly it does not estimate the other appreciable costs of some species. This chapter will examine various ways in which such costs might be estimated.

The approach here is based on two programs of work with which I have been involved. The first is the study of the impacts of alien species and how to measure them.[43,70] The second was the Economics section of the Global Invasive Species Programme (GISP).[45] It is important to note that economics is not accounting, even though economic assessments will normally include a cost-benefit analysis. So the GISP Economics Programme produced few cost estimates, and none that should be taken too seriously. The same applies to the cost figures in this chapter. Although I give some numbers, the importance and effect of

0-8493-0836-4/02/$0.00+$1.50

alien invasive plants in the British Isles is given more reliably by an understanding of how costs arise and the policy options for containing them, rather than by concentrating on narrowly based figures.

This chapter deals with the British Isles; that is, the large islands of Britain and Ireland and numerous smaller associated islands. Politically, that involves two sovereign states: the Republic of Ireland, and the United Kingdom of Great Britain and Northern Ireland. For biological purposes they are usually treated together. Britain and its associated islands comprise about 229,000 km^2, 131,000 of them in England. The population of Britain is about 54 million. (All the population figures given here are based on the 1991 censuses.) Most of that population is in England; there are nearly 5 million in Scotland and about 2.8 million in Wales. Ireland is considerably smaller than Britain at about 84,000 km^2, and it has a population of slightly more than 5 million. The total area considered here is about 313,000 km^2. England comprises only about 40% of the land area, but it accounts for more than 75% of the economy of the British Isles.

As a preliminary, it is desirable to know how many plant species of different status are thought to grow in the British Isles, which is more problematic than the invasion literature might lead one to expect. It is also necessary to clarify how impact may be measured and the relationship of that to cost. I will deal with those two points first and then consider 30 particular invasive alien plant species. Only after that will I consider the distribution of impact and cost over the British flora, with a view to getting an overall understanding of the impact of alien plants in the British isles.

6.2 The number of British alien plant taxa

Both the number of British native plants and that of alien plants are uncertain. There are further doubts about the ecological status of some taxa. I'll describe these uncertainties and show the effects they have on numbers.

With native plants, the troubles come mostly from microspecies and hybrids. Hybrids are perfectly satisfactory taxa, recognizable and nameable, such as the cord grass hybrid *Spartina* × *townsendii*. The × indicates it is a known hybrid, in this case between the native *S. maritima* and the alien, American *S. alterniflora*. *S.* × *townsendii*, like many hybrids, can only reproduce vegetatively, but it is the parent of *S. anglica* (discussed below as one of 30 interesting species), which is, again like many hybrids, fertile. But most hybrids fail to form populations, fail to establish, and occur only near their parents. Counts of British species usually omit hybrids, but as there are around 400 of them listed in the floras, including them makes a large difference to the counts of taxa. There is also the question of whether to count crosses between natives and aliens, such as *S.* × *townsendii*, as native or alien; most floras, oddly in my view, call them native if they have arisen in the British Isles. They are non-indigenous species in the sense of not having been in Britain before agriculture.

Microspecies and critical species are common in the British flora. Critical species are those where identifications need to be verified by an expert in the group, but may nevertheless be perfectly good species in all senses. They are just difficult to identify. All microspecies are critical, but are in groups that are often apomictic, so the definition of a species is unclear. Stace[59] estimates there are 400 microspecies in *Rubus fruticosus* agg. (blackberries), 250 in *Hieracium* (hawkweeds), and almost all are native. There are ordinary, non-critical species in those genera too. In *Taraxacum* (dandelions), 226 microspecies are recognized: 39 are endemic, 76 are described as other natives, and 111 are considered alien. With modern genetics techniques, many more could be distinguished. Generally none of these are included in counts comparing the British flora with others. With around 900 native microspecies, counting them in the total of native species would make a huge difference to comparisons.

Even so, there is doubt about the number of what I will call in this context "native macrospecies." There have been three authoritative floras in the past 15 years, and counts from them produce 1311,[8,68,69] 1225,[58,63] and 1552[59] macrospecies. Taking the highest of those and the counts of hybrids and microspecies gives 2852 native species. But one could argue that the figure should be as low as 1225. I would suggest that about 1500 provides a sensible basis for comparisons. This figure is not far from the 1407 natives picked out by the Ecological Flora Database.[24,74]

The next uncertainty is whether all of those species are in fact native. Some of them may well be aliens. Almost all native species had to invade the British Isles since the last glaciation, so those known to be growing in the forested landscape of the Mesolithic, before agriculture, roughly 5000 to 10,000 years ago, are called native. It is customary to call native those present in the late glacial period, notably some species of disturbed ground, even though some may well have died out and been reintroduced with agriculture. But there are many species for which there is no fossil or historic record and which might be native or not. In the floras, roughly 10% of the species have labels indicating uncertainty, such as "probably native" or "possibly introduced."

The pair of complementary catalogues of alien plants[9,54] list 49 species as "accepted with reservations as native." One standard flora[58] lists seven of these as unqualified native, while another[8] lists 11, but there is only one species common to both sets, *Centaurea cyanus*, the cornflower. That is native on the basis of only one well-stratified pollen grain, of more grains that could have been washed down, and from its occurrence in post-glacial, pre-agricultural deposits on the mainland of Europe. Salisbury[55] was remarkably indignant about this sort of procedure: "Hence the presence of seeds, still less of pollen grains, of a species afford little if any evidence as to its status, whether casual or more or less naturalised. To assert, because of the presence of the pollen of a species in prehistoric deposits, that it is 'native' is at once misjudged, misleading and well-nigh meaningless." That is too strong a position, but caution is needed.

As the number of native species is uncertain, so is the number of aliens that would be called archaeophytes on continental Europe, those introduced before ca. 1500 AD.[50] Yet it is essential to include archaeophytes when estimating the cost of aliens, as their impact is much the same as neophytes (those introduced after ca. 1500). *Aegopodium podagraria*, ground elder, and *Avena fatua*, wild oats, are two notable archaeophytes in the list of 30 below.

Most neophytes have a date when they were first introduced into the British Isles, or first found outside cultivation, or both. But species first found relatively recently may still be labeled native. An example is *Gladiolus illyricus*, wild gladiolus. It is found in a few places in Hampshire in the extreme south of England, but these locations are nevertheless about 400 km north of European mainland records. For a species with a showy flower in a county full of naturalists, the first date of 1856[40] and its disjunct distribution suggest to me that it may well be alien. It is said that it "has the look of a genuinely wild species,"[40] but Webb[66] showed how unreliable such a criterion is.

But even when species are clearly introduced, difficulties with the terms *casual*, *persistent*, *established*, and others lead to very different counts of the number of alien species. Table 6.1 gives counts I published some years ago,[68] counts that underpin the tens rule[69,72,73]: that 10% of plant taxa imported into the British Isles become at least casual, while 10% of the casuals become established. Of the established, about 10% become pests, that is, economically significant. Table 6.1 shows some of the different usages of "established"; the tens rule works with "fully established" rather than "locally established." Local floras, perhaps not surprisingly, seem generally to follow "locally established," as

Table 6.1 The Number of British Plant Aliens by Status

Severe pests	11
All pests	39
Widely naturalised	56
Fully naturalised	196
Subtotal including pests	**210**
(established, sensu Williamson and Fitter[69,73])	
Locally naturalised	348
(established as used in some county floras	
Subtotal, all above	**558**
Garden outcasts	223
Casuals	898
Subtotal, all above	**1642**
(introduced, sensu Williamson and Fitter[69,73])	
Other imports	10,821
Grand total	**12,507**

From Williamson, M., Invaders, weeds and the risk from genetically manipulated organisms, *Experientia*, 49, 219–224, 1993.

Table 6.2 Counts of Plant Taxa in the British Isles

Source	Natives[a]	Established aliens	Established as % native	All aliens	All aliens as % native
British Counts					
Ecological Flora Database[24,74]	1407	196	14	—	—
Williamson[68] (Table 1)	—	210 or 558	—	1642	—
Vitousek et al.[63]	1255	945	75	—	—
Alien catalogues[9,54]	—	945	—	3467	—
Stace[59] (Weber/Pyšek count)	1552	725	47	—	—
Clapham et al.[8,68]	1311	193	15	—	—
County Counts					
Cumbria[30]	951	—	—	469	33
14 counties etc., mean[14,38]	878.8	—	—	449.5	34
5 northern vice-counties[27]					
Cheshire	868	225	26	363	42
S. Lancs.	831	266	32	685	82
W. Lancs.	922	229	25	657	72
Durham	1000	430	43	656	66
Northumberland	949	279	29	626	66

[a] "Natives" mostly exclude hybrids and microspecies, but the usage is not consistent.

can be seen in Table 6.2. But the proportions in the set of county floras from the north of England are very significantly different, showing that different standards are being used.

Various counts of the numbers of aliens in the whole of the British Isles are given in Table 6.2. There are three counts of around 200 for fully established, going up to 945 for established in the weakest sense. That sense is, in the limit, a single plant thriving: "at least one colony either reproducing by seed or vigorously spreading vegetatively."[9] What is fairly certain is that the 745 or so species that are only locally or weakly established present negligible costs of any sort.

The number of casuals is far higher than the number of established species, whatever criterion is used. The set of casuals includes a great many garden escapes and occasional planted specimens. Some of them nevertheless have important costs, namely those that are volunteers, so-called, in crops. Volunteers come from previous crops on the same site. Oil seed rape *Brassica napus* and potato *Solanum tuberosum* (both hybrids as crops) rank eighth and twelfth in the herbicide costs estimated by Prus,[49] ahead of all species in the 30 interesting species considered below except for the two *Avenas* (wild oats) and *Veronica persica* (common field speedwell). Although the database I used in elaborating the tens rule[68] had only 1642 casual and established alien species in total, by searching for every record on single plants and other extreme casuals, the alien catalogues[9,54] raised the number to 3467. The numbers established, using whatever number you take from the previous paragraph, need to be subtracted from the total number of aliens to give the number of casuals. But with the exception of the volunteers, the cost of these casuals will be totally negligible.

So what is the proportion of the British flora that is alien? Lonsdale,[38] using some significantly heterogeneous data brought together by Crawley,[14] thought it was 31%, while Vitousek et al.[63] made it 75% (expressed as 43% of the total). Using traditional figures of about 1500 native good species and about 200 fully established aliens, the result is 12%. Using the highest totals above, 2900 natives (including hybrids, critical, and microspecies) and 3500 aliens seen in the wild since 1930 gives 55%. To each their own. My own view is that the lowest of those three figures, 12%, gives the best feel for the noticeable impact of aliens in British vegetation. It is also quite close to the 9% that Lonsdale[38] estimated for the rest of Europe.

6.3 From impact to cost

The possible types of impacts of aliens and the ways in which they might be measured are both large.[43] The Lonsdale equation

$$I = R \times A \times E$$

where I is the overall impact, R the range size, A the abundance, and E the effect per unit, brings some order. R and A are fairly straightforward, but E is still fairly complex.

Nevertheless, the Lonsdale equation is about as complicated as present measuring techniques usually allow. It would be desirable to add the extra dimensions of species interaction, community structure, and so on, but for the present they are normally measured as the effect E of the invasive species. It is rare for the data to be good enough to measure multivariate effects, but when they are, the results are interesting.[67] There are no such data available for British alien terrestrial plants as a set.

In theory, each impact could be converted into an estimate of cost; or, even better, a functional relationship could be found between variation in the impact and variation in cost. Again, this is not possible with present data for most British alien plants. The Lonsdale equation does, however, allow us to say that when one of its three components is negligible, then the total impact and so the total cost will also be negligible. That simple rule, as will be seen, applies to a surprisingly large number of alien plants in Britain.

For some British aliens, I was able[70] to find five quantifiable measures, of which two were the first two components of the Lonsdale equation: range and abundance. The other three related to weediness: weediness as perceived by a panel of scientists, weediness as measured by the cost of herbicides, and weediness as measured by the incidence of weeds in an agricultural survey. The correlations between these were only moderate,[70] showing they were indeed measuring different aspects of impact. Different aspects of cost should, similarly, be measured by different things. How this might be done is best treated by considering individual species.

6.4 Thirty interesting aliens

6.4.1 Generalities

In order to describe in general the cost and impact of British non-indigenous plants, I have picked 30 for more detailed discussion. These are the 20 listed by Crawley[14] as "The 'top twenty' British alien plant species" along with 10 others that have a major impact in some measure. Coincidentally, they include 10 that are not regarded in the alien catalogue[9] as naturalized and another 10 that are not spreading according to the data from the Sample Survey.[42] The catalogue definition of naturalized is "Established extensively amongst native vegetation so as to appear native."[9] As a first approximation, only species naturalized in that sense will have an important environmental impact, even though those not so naturalized are often conspicuous. It is more common to use "naturalized" just to mean "established,[53] and the two usages cause some confusion. Economic impact can be important whether a plant is naturalized in the alien catalogue[9] sense or not; arable weeds such as *Avena sterilis* (wild oat) and *Veronica persica* (common field speedwell) are examples of the latter.

The major and consistent estimate of cost for these 30 species is what I call the Prus cost.[49] This is stated both as the cost in pounds sterling per year and as its natural logarithm, e.g., 13.816 or £1 million. Prus calculated his weed cost for each species from three main variables: the value of herbicide sales, the cost of application, and the cost of cultivation. He derived these for each species by an ingenious use of government statistics, manufacturers' information, and surveys of farmers, foresters, nurserymen, and head gardeners. The result is a cost of control not just of agricultural weeds, but of all species in the British flora. Nevertheless, it is fundamentally an estimate derived from herbicide costs.

Estimates of rate of spread in the accounts below, called Sample Survey[42] estimates and explained more fully in Section 6.5.3, are derived from comparisons of surveys in 1952–60 and 1987–88, shown graphically in Figure 6.1.

6.4.2 Species accounts

1. *Acer pseudoplatanus*, sycamore. This is a native European tree species, and it is surprising that it failed to reach England after the last glaciation. It is often said to have been a Roman introduction, but that is probably wrong. Jones[33] found records for Scotland from the 15th century, possibly earlier, but from England only from the 16th century, and it seems not to have been established in the wild before the 18th century. That is consistent with its absence in the archaeological record. Now it behaves like a native and disperses readily, though it is not spreading, having filled its range. "Ubiquitous in mixed and deciduous woodland, parkland, as a planted street tree and in shelter belts and hedgerows"[75]; "a Johnny-come-lately out nativing the natives in almost any situation, shading out native species."[40]

It is, however, difficult to estimate sycamore's impact in ways other than range. It is commonly of concern in nature reserves[70] and in forestry. It is probably the main source of the "wrong sort of leaves" that delay trains in the autumn. The Prus[49] cost estimate is

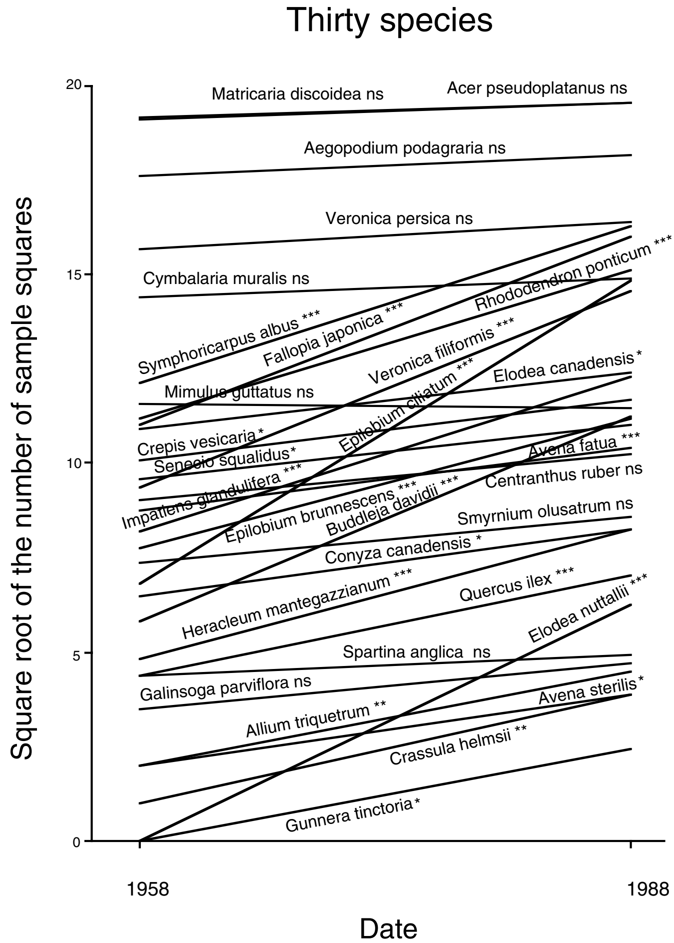

Figure 6.1 The change in recorded number of sample hectads for the 30 species. The Atlas survey[44] was done between 1952 and 1960 but was more or less complete by 1958; the Sample Survey[42] was done in 1987 and 1988. Note the square root scale of the ordinate.

10.71 = £44,802, which is very low, showing that it usually is controlled mechanically. Against that should be put the considerable benefits of the species. It forms a straight, handsome tree in exposed and in polluted sites. So it provides shelter for upland farms and near the sea, and it adorns towns and other places. Entomologists have mixed feelings about it. Any total cost figure, particularly one that allowed credit for the benefits, would, on my present information, just be a wild guess.

2. *Aegopodium podagraria*, ground elder. This perennial herb is an ancient introduction, apparently brought in by the Romans, possibly as a pot-herb, possibly for medical reasons (it is also called goutweed). It is now a major garden weed in the British Isles and relatively rare away from gardens. Wilmore[75] describes those non-garden habitats: "widespread colonist of waste ground, disused gardens, roadside verges and other marginal land." It is not spreading, and is far from ubiquitous in Scotland and Ireland. The total cost in time and effort for gardeners must be considerable. It is often said that the only satisfactory way to get rid of bad infestations is to dig them out completely. In practice, infestations in orchards and such places are usually left, or are cut along with the grass. Small populations can be eliminated by painting glyphosate on the emerging leaves in spring;

I've done it. Variegated forms are still sold to gardeners, but the benefits of this plant must be negligible in comparison. The Prus[49] cost estimate is 10.71, or £44,802 per year, which suggests a total cost of between £100,000 and £1 million per year.

3. *Allium triquetrum*, three-cornered leek or white bluebell, is a weed of rough, waste, and cultivated ground, including copses, hedgerows, and waysides.[59] It is a perennial herb that grows to about 45 cm. It was found almost exclusively in southwestern England until quite recently, but now seems to be spreading fast and diffusely to Ireland, Wales, the Isle of Man, and southern England. Although it can be quite abundant locally, the significant impact of this species is as a weed of bulb fields (of daffodils, etc.) in Cornwall and the Isles of Scilly. There it has been a serious weed since the 19th century. It is regarded as impossible to eradicate, and "most islanders have abandoned any attempt at control."[39] Therefore its cost would have to be estimated from the loss through slower growth that it causes the growers. I have found no data on this. As a national cost it would seem to be negligible. The Prus[49] cost estimate is 8.82 = £6768, which is small.

4. *Avena fatua*, wild oat, is an ancient invader dating from the Bronze Age, 3000 years ago or so. Nevertheless it remains largely a weed of lowland England, although it has been spreading for a surprisingly long time. In the northwest of England the first historical record for Cheshire was 1805, for south Lancashire 1840, and for west Lancashire 1900,[27] despite there being Bronze Age records from Cheshire. But it has also become more abundant in recent decades because of the difficulty of controlling a grass weed in a grass crop such as wheat with herbicides. It is a "weed of arable and waste ground and also a wool and bird-seed alien."[75] The Prus[49] cost estimate of 17.88 = £58,235,168 is by far the largest in the 30 species. But it is not the most expensive in his list, being exceeded by *Alopecurus myosuroides*, or black-grass, and *Galium aparine*, cleavers or goose-grass, which are both native, and *Matricaria recutita*, scented mayweed, which Prus[49] (following Webb[66]) regarded as a neophyte from the 16th century, but which most floras call native.

5. *Avena sterilis*, winter wild-oat, is a much more recent invader than *A. fatua*, having been introduced during World War I, and has a much more restricted distribution in central England. It is found in similar places to *A. fatua*, but usually on heavy soils and replacing it there.[59] Nevertheless, where it occurs it is a major weed of cereals, giving a Prus[49] cost estimate of 16.36 = £12,736,724 which represents a major national cost. It has not been found much outside of crops, and not in native vegetation.[54]

6. *Buddleja davidii* is usually known as buddleia, although the Botanical Society[19] name is butterfly-bush, a name that partly explains its popularity with gardeners. It is a shrub that grows to 2 m or more, with sprays (pyramidal panicles) of typically lilac-colored flowers (also purple or white) in late summer. As a naturalized alien, it is found chiefly in marginal and derelict habitats: waste ground, walls, banks, and scrub[58], where the costs and benefits may be evenly balanced. However, it is still spreading (the fourth fastest of the set of 30 in 1958–88; see Figure 6.1) and may yet become an environmental threat. For instance, Everett[22] says, "I am watching the march of Buddleja along the Kennet and Avon Canal, where it is ousting fen and water-margin natives such as Comfrey [*Symphytum officinale*] (foodplant of local Scarlet Tiger moths [*Callimorpha dominula*]), Meadowsweet [*Filipendula ulmaria*] and willows [*Salix* spp.]…a stone's throw from the River Kennet proposed Special Area of Conservation," and she goes on to point out that it could fairly easily be eliminated now, but soon will not be easy to eliminate. No action is being taken, and the species is being recommended by some conservationists for its value as a feeding source for adult butterflies. This is a familiar sort of story to invasion biologists, but it would be hard to claim there is an appreciable cost now. The Prus[49] cost estimate is 7.54 = £1881, which is negligible.

7. *Centranthus ruber*, or red valerian, is a garden plant, an erect perennial growing to 80 cm that escapes to colonize walls, disused railway land, and other waste places. It is

a 17th century introduction, so it is not surprising that it is no longer spreading. It is unwanted in some places, leading to the Prus[49] cost estimate of 9.74 = £16,984, but its national impact and cost are trivial, even though it can occur in native vegetation.[9]

8. *Conyza canadensis*, Canadian fleabane, is an annual herb introduced in the 17th century. It is a "quite widespread plant of urban derelict land, waste ground, disused railway land and marginal areas which seems to be increasing its range and abundance [in Yorkshire] in recent years"[75] and may possibly have been spreading nationally in 1958–88 (Figure 6.1). The Prus[49] cost estimate of 9.74 = £16,984 shows that it is sometimes unwanted. It is interesting as a plant that is more of a pest where native than where introduced.[15,51] In the British Isles, where it is not found among native vegetation[9] and is not a serious weed of cultivation, the total cost is trivial.

9. *Crassula helmsii*, New Zealand pigmyweed, is an herb grown by aquarists and discarded or planted in ponds; it is "well naturalised in many places in south England, rapidly spreading."[59] Clement and Foster[9] describe it as abundant and a threat (though they neglect to use the word "naturalized"). Its history in Britain and maps of its known distribution in 1969, 1979, 1989, and 1998 are given by Leach and Dawson.[35] Some ineffective attempts to control it with herbicide have been described.[6] The Prus[49] cost estimate is only 0.63 = £2, a derisory figure in light of its very recent spread. It seems unlikely that control will be effective except very locally, and its cost, potentially large, should be estimated from the environmental cost of changed habitat and reductions in other species. I know of no way of doing this that I would believe.

10. *Crepis vesicaria*, beaked hawk's-beard, is an herb of grassy places, waysides, walls, and rough ground.[59] It is not found in native vegetation[9] and probably is no longer spreading, which is not surprising for an 18th century introduction. The Prus[49] cost estimate of 9.19 = £9799 is trivial, and it is in the Crawley[14] top 20 merely from being a commonly seen plant. There can be no appreciable national cost.

11. *Cymbalaria muralis*, ivy-leaved toadflax, is a common herb on English walls and was introduced in the 17th century. A "locally abundant plant of walls, disused quarry areas, builders' rubble, derelict sites and marginal land."[75] Considering where it grows, the Prus[49] cost estimate of 9.74 = £16,984 is surprisingly high. It is not found in native vegetation and can be a pleasant adornment of walls, a minor benefit. Boyd Watt[3] gives the history of its introduction as a garden plant, and he notes that it is a prolific flowerer and deserves the name used in some parts of "mother of thousands." I would put its national net cost at zero.

12, 13. *Elodea canadensis*, or Canadian waterweed, and *Elodea nuttallii*, Nuttall's waterweed, are both found in streams, dikes, and canals, and other slow-moving or still water bodies.[75]

The history of the spread of these two pond-weed species is given by Simpson.[57] Briefly, *E. canadensis* was first recorded in 1836, increased rapidly, and often became a pest. But it declined in abundance if not range from the 1880s. It can still be locally abundant or dominant in some stretches[75] and is no longer spreading. *E. nuttallii* was only recorded in 1966 and is still spreading. It has often replaced *E. canadensis*, and although it can form large and extensive beds, it has rarely been regarded as a pest. The economic cost of these two species is essentially confined to the mid-19th century; the present cost is negligible at a national scale. Environmentally, there may even be some benefit now from increased habitat heterogeneity and water oxygenation. The Prus[49] cost estimate for *E. canadensis* is 9.74 = £16,984, just about worth noting, but for *E. nuttallii* it is only a derisory 3.09 = £22, reflecting its recent spread and confusion with *E. canadensis*. Together their total national cost must be less than £100,000 annually.

14. *Epilobium brunnescens*, New Zealand willowherb, is a prostrate perennial herb that (like 29, *Veronica filiformis*) was introduced as a rock garden plant, first noted as a casual

in 1908. There was confusion about its name for some time, there being many epilobia in New Zealand, and it was called *nerterioides*[44,55] and, earlier, *pedunculare*. New aliens are not infrequently difficult to identify. It is now found on "damp stony or marshy ground, often in upland terrain, as well as being a noticeable garden weed,"[75] still spreading (Figure 6.1) and occurring sometimes in natural vegetation. The Prus[49] cost estimate of 9.19 = £9799 presumably reflects the herb's behavior in gardens. Outside, it just seems yet another minor if common addition to the flora that is of no consequence, although of some interest. The Prus[49] cost is probably the right order of magnitude for the total cost.

15. *Epilobium ciliatum*, American willowherb, is another *Epilobium* with a changing name; it used to be called *E. adenocaulon*, and I would not be surprised if it changed its name again, as it is a member of a critical group. There seem to have been two important introductions, possibly of different genotypes (or even species). The introduction in Leicestershire before 1891 established but scarcely spread. That in Surrey was before 1930 and spread steadily[48] in all directions, including over Leicestershire. It was the fastest-spreading alien in 1958–88 (Figure 6.1) and is very common in some areas. It is a perennial herb, a "weed species of disturbed ground, woodland edges, disused railway land, urban waste ground and often frequent on damper stream or canal sides."[75] The Prus[49] cost estimate of 9.19 = £9799 is the same as that for *E. brunnescens*, but since it is a less serious garden weed and a less serious weed in natural vegetation, I would put the total cost as less, even though it is the more abundant species.

16. *Fallopia japonica*, or Japanese knotweed, another perennial herb, was introduced as a garden flower and won prizes as such.[2] Nowadays it is much disliked, even feared, particularly in cities as a "widespread aggressive colonist of waste ground, disused cemeteries, railway land, disturbed woodland herb layers and sometimes damper, rich organic soils."[75] It is also one of the two colonizing plants named in the Wildlife and Countryside Act of 1981; the other is 19, *Heracleum mantegazzianum*. There is even a Japanese Knotweed Alliance (JKA).[72] The problem with this plant comes from its rhizomes, which can go to 2 m in depth. They are difficult to kill with herbicide, and the plant can regenerate from small fragments (as little as 0.7 g), so digging may cause more harm than good. The Prus[49] cost estimate of 10.71 = £44,802 is not large. For the city of Swansea in Wales, the JKA estimates that £1/m^2 for the cost of spraying glyphosate and £8/m^2 for landscaping would come to £9.5 million. But would any sensible authority pay that if it realised how ineffective glyphosate is with this plant? What Swansea's planning department has actually spent is £140,000 over 6 years for treating established populations.[72] The Loughborough group[4,6,17] seem to me to show that endless sums can be spent on ineffective control. The JKA would like to try classical biological control. This has never been used against a plant in the British Isles and would have to be extremely specific,[69] as there are closely related native species.

Although *F. japonica* is undoubtedly a major problem in some places, there are those who say the scope of the general problem it poses is exaggerated. Dickson[18] writes from personal knowledge that "Japanese Knotweed was already very common in the Glasgow area forty years ago.... If it is a problem now it was a problem then," and he argues against major attempts to control aliens in urban sites (and strongly for controlling aliens that may invade "vegetation of outstanding interest," such as 24, *Rhododendron ponticum*). Gilbert[25] finds merit in Japanese knotweed as a habitat for grass snakes (*Natrix natrix*) and otters (*Lutra lutra*) and as actually improving the habitat for spring woodland flowers in the Sheffield area.

Clearly, any estimate of total cost is greatly affected by perception and by whether the money is being well spent. My guess is that the cost of controlling *F. japonica* effectively, where it really needs to be controlled, could be as much as £1 million a year. It is doubtful if the cost of developing and testing biological control would be justified; the use of better herbicide regimes[26] seems a more cost-effective, and politically acceptable, route.

17. *Galinsoga parviflora*, gallant soldier, is another noticeable perennial herb invader that is a "well naturalized weed of cultivated and waste ground"[59] and a garden weed in some places. It is not a threat to semi-natural vegetation and is no longer spreading. The Prus[49] cost estimate of 12.73 = £337,729 reflects its image with gardeners and seems high for another daisy-flowered weed with an amusing English name.

18. *Gunnera tinctoria*, giant rhubarb, is a spectacular herb that produces leaves almost 2 m across and stands 1.5 m high. It is "planted by lakes etc. and often self-sown where long-established; naturalized in scattered places through much of lowland British Isles."[59] The Prus[49] cost estimate of 0.63 = £2 reflecting the smallness of the problem in general from this species, but it is spreading (Figure 6.1) and is a problem in some semi-natural grassland, especially in the west of Ireland,[31] where it can occur as stands suppressing all other plants. It is not known if control will be needed, how difficult it would be, or what it would cost. I include it here as an example of the early stages of an invasive alien that could conceivably become costly in the future.

19. *Heracleum mantegazzianum*, giant hogweed, is another impressive perennial herb with a reputation (not really deserved according to Dickson,[18] but correct according to Wade et al.[64]) of causing serious dermatitis. The Prus[49] cost estimate of 9.74 = £16,984 is quite low. But this plant is one of the two named in the Wildlife and Countryside Act of 1981. It is "common along industrial river corridors and in wetland areas, also found locally along motorway verges and in waste ground and tall ruderal grassland."[75] The spread and management of this species and the next have been studied and modeled by the Durham group.[10,65] While they conclude that successful management depends on understanding population structure, and they model such structure fairly successfully, they make no cost estimates.

20. *Impatiens glandulifera*, Himalayan balsam, is an annual herb, the tallest such plant in the British flora at 2 m. It is an "aggressive colonist of river and canal banks, sewage works, waste ground and damp carr woodland."[75] For its history and spread, see Section 6.5.3. But the Prus[49] cost estimate is only 9.74 = £16,984, as it is neither an agricultural nor a garden weed. As was noted in the previous species, the management has been modeled,[10,65] but without estimating costs. As an annual, it might be thought it would be easy either to pull up the plant or to cut off the flowering stems, particularly as there is only a small seed bank. Most seeds germinate within a year. In practice, such measures usually give only temporary relief. Estimating the cost requires estimating the value of the biodiversity in the woodland. In some cases, it might be possible to put a value on the pheasant (*Phasanius colchicus*) shooting lost, but valuing the biodiversity in a nature reserve such as Askham Bog near York is still an essentially subjective process. But clearly the cost must be several times the Prus cost, suggesting maybe £100,000 to £500,000 a year, but all such figures are very foggy.

21. *Matricaria discoidea*, pineapple-weed, a small annual herb, is a "virtually ubiquitous species of waste ground, path edges, gardens, muddy gateways of arable and pasture fields, disused railway land, and marginal land and verges,"[75] but it does also occur a bit in the body of arable fields. The Prus[49] cost estimate is 13.59 = £798,108, which implies that some farmers find it weedy. It is no longer spreading and is not found in native vegetation. Its characteristic habitat is bare ground unusable by other species, so to that extent it is a neutral addition to British biodiversity. It is difficult to see in what way this species can inflict a real cost of nearly £1 million.

22. *Mimulus guttatus*, monkey flower, is a low-growing, but often prolifically flowering, perennial herb. It lives in "stream flush zones, pond edges, marshy grassland and sometimes acidic wetland zones on moorland."[75] It seems not to threaten biodiversity or

anything else, and its flowers can liven up otherwise rather drab habitats. It has completed its spread in Britain. The Prus cost estimate of 9.19 = £9,799 is trivial, and there seems no reason to add to it.

23. *Quercus ilex*, evergreen oak or holm oak, is a fine tree. "Introduced; much planted for ornament, and often for shelter in east England; self-sown in south and central England, Wales, south Ireland and the Channel Islands."[59] The Prus[49] cost estimate of 7.54 = £1881 shows that herbicide would not usually be used to control this species. As a fine tree it brings many benefits, but it has costs, too: "This species is locally becoming a threat to native vegetation,"[9] but then so are some native trees. The net cost is probably near zero, regardless of how these effects are valued.

24. *Rhododendron ponticum*, rhododendron, is an "evergreen shrub in woodlands, ornamental parkland and large gardens on acidic or semi-acidic soils"[75] and can grow to 5 m. It has been much planted in woodland, particularly in Victorian times (19th century), to give cover for pheasants (*Phasanius colchicus*) and for its profuse flowers. The British stock came primarily from southern Spain, and much of it is hybrid, crossed particularly with *R. catawbiense* but also with *R. maxima*, both from the Appalachian Mountains in the eastern United States.[41] The hybrids may be important in allowing the taxon to thrive in harsher climates. In westerly parts of Britain, rhododendron can be a very serious problem, forming dense monocultures and shading out all other species. In the east it is much more rarely a pest. It is still readily available from nurseries, and there is often no reason why it should not be grown in gardens. Nevertheless, it is probably the major alien environmental weed in the British Isles.

The extent of the problem has been described[16,29,62] in many places. It is a problem in forestry, for national parks and conservation bodies, for the National Trust (which owns and manages buildings and land of historical and environmental importance), and for land owners in general. The Prus[49] cost estimate of 10.71 = £44,802 is a serious underestimate of the cost of rhododendron. This is because much of the control is mechanical, either by machines or by hand cutting. Hand cutting may be necessary on difficult terrain and is often done by volunteers. Gritten[29] estimated the total cost in the Snowdonia National Park in north Wales at £45 million. As there are less than 45,000 ha of woodland in Gwynedd[37] (the county containing the park), that would imply a cost of many thousands of pounds per affected hectare, even allowing for some spread beyond woodlands, but it is not clear how the figure has been derived. On National Trust property, rhododendron bashing is second only to bracken bashing as hard labor by volunteers. (Bracken is the native fern *Pteridium aquilinum*, and bashing means attacking in any physical way.) Costing that effort is difficult.

One place where rhododendron is a threat to biodiversity is on the island of Lundy in the Bristol Channel. This is the only locality for the endemic Lundy cabbage *Coincya wrightii* (Brassicaceae), whose closest relatives are in Spain.[12] Lundy cabbage is the only food plant for the flea beetle *Psylliodes luridipennis*. *C. wrightii* is confined to 2500 m of the east coast of Lundy. Its range is restricted by grazing mammals and exposure to southwesterly storms, so it occurs mostly on cliffs and in gullies. These are now being invaded by rhododendron, and the whole population of *C. wrightii* would probably eventually be shaded out[13] without control measures. But clearing rhododendron from cliffs is dangerous work, requiring skilled climbers and stringent safety controls. In 1997 it took 226 volunteer-hours to clear one hectare.[13] It may be possible to eliminate rhododendron from cliffsides and clifftops along with 5 metres of the cliff edge by 2006 with 105 days work, or a cost of £26,880 overall[11] at commercial rates. That works out to almost £60,000 per ha, reflecting both the difficulty of the terrain and the high cost of labor when paid for. Even so, it may be an underestimate, because glyphosate, as applied, has not stopped regeneration, and other herbicides have yet to be tried.

The National Trust (for England, Wales, and Northern Ireland) and the National Trust for Scotland (NTS) have kindly provided me with some figures. In Scotland, on the island of Arran a 40-ha plot was managed at a cost, not counting volunteers, of £20,000. That is £500 per hectare with free labor. Although the NTS is the largest charitable conservation organization in Scotland, it has only about 500 ha of rhododendron needing control, which leads to an estimate of £250,000 plus the value of volunteer labor, but that represents total cost, not annual cost.

The National Trust owns about 250,000 ha in all, of which about 25,000 ha is woodland managed by the Trust. Rhododendron has been controlled on about 1000 ha in the past 10 years, though less than half of that involved dense rhododendron scrub. The cost averages around £2000 to £2500 per hectare, with a range from £200 to £4000, again not including the value of volunteer labor, but including both initial mechanical clearing and the labor and chemical costs of herbicide treatment of stumps and of regenerating leaves. That comes to at least £200,000 a year in direct costs.

The extent of the rhododendron problem has not been quantified, so it is not possible to extrapolate from these figures a total cost in the British Isles, either to what is being spent or to what should be spent. But clearly the figures would run into millions, if not tens of millions, of pounds. Indeed, if Gritten[29] were to be believed, it would be hundreds of millions.

25. *Senecio squalidus,* Oxford ragwort, is well named, as it seems it is a species that arose in Oxford Botanic Gardens. It is non-indigenous rather than an alien, as is 27, *Spartina anglica*. Some time in the 17th century, material from the hybrid swarm, on Mount Etna in Sicily, between *Senecio aethensis* and *S. chrysanthemifolius*,[1] was brought to Oxford and cultivated. By the 1790s it was growing on walls in Oxford.[34] The current feral species "originated in cultivation"[1] and is fairly certainly the consequence of evolution and adaptation in the Botanic Gardens. Unusually for an invasive species, it is self-incompatible. Possibly the "genetic flexibility" in the system was crucial to its success;[32] only four S alleles have been found, S being the incompatibility locus.

Oxford ragwort's spread has been rather irregular and not all that fast, partly along railways that give it suitable habitat. It is still spreading in Ireland and Scotland. Salisbury[55] claims that *squalidus* refers to the habitat, but that is not so. A common English name for it in the early 19th century was inelegant ragwort, which gives probably the best translation of the Latin and refers to the disposition of the ray florets. "One ca'n't help one's petals getting a little untidy."[5] The name Oxford ragwort seems to date from 1886.[20] Nowadays it is found in "waste ground, disused railway land, canal towpaths, waysides and derelict land generally."[75] The Prus[49] cost estimate 9.19 = £9799 is surprisingly high for a species that is neither found in native vegetation nor a pest. I would be reluctant to put its cost at anything but zero. But it is a most interesting plant biologically.

26. *Smyrnium olusatrum*, alexanders, is a biennial herb, the only biennial in this list of 30 species. It is "fully naturalized on cliffs and banks, by roads and ditches and in waste places, mostly near the sea"[59] and is not spreading. Some of its habitats are natural, but it seems not to threaten biodiversity. The Prus[49] cost estimate is trivial at 7.54 = £1881 and seems a fair estimate of its cost.

27. *Spartina anglica*, common cord-grass, is a perennial grass of mud flats and is, like 25, *Senecio squalidus*, non-indigenous but not alien. Most floras call it native, though the Joint Nature Conservation Committee editors[21] disagree, as do I. Its history is well known,[69] and, as stated above, it is the fertile allotetraploid derived from the sterile diploid *S. townsendii*, which is itself derived from the cross between the native *S. maritima* and the alien, American *S. alterniflora* (which was the female[23] parent). All these are tidal mud species. *S. anglica* is useful for reclaiming mud flats, but it is a serious problem in that it blocks channels. It is no longer spreading in the British Isles. Much of the present distri-

bution comes from planting, and it is in fact declining in the south of England. The Prus[49] cost estimate is 4.98 = £145, showing that herbicide is not used to control this species. Millions are spent controlling *S. anglica* overseas, for example in Tasmania and in Washington state in the United States, but not in the British Isles. In view of its balance of costs and benefits, I would put the net cost in the British Isles as near zero.

28. *Symphoricarpos albus*, snowberry, is a low shrub that is often planted for cover. It "occurs in woodland, scrub, thickets, ornamental parkland, churchyards, hedgerows, wasteland and large gardens"[75] and spreads vegetatively quite vigorously. The Prus[49] cost estimate of 4.98 = £145 shows that herbicide is not used to control this species. Indeed, it is only a problem when planted where its spread is unwanted. The cost of this species must be near zero.

29. *Veronica filiformis*, slender speedwell, was introduced as a rock garden plant and soon became invasive of lawns.[69] Some gardeners dislike it and try to control it. It was still spreading quite rapidly between 1958 and 1988 (Figure 6.1). In 1996, I wrote that "Each April the lawns of the campus at the University of York turn blue with the flowers of *Veronica filiformis*,"[6] but this is no longer true. It has died back and now occurs only in small patches, and that seems to be true elsewhere as well. It has been found "in shorter mown grassland and verges, soft turf banks, and sometimes stream sides"[75] and, I would add, in longer grass, as in my orchard. It is not found in native vegetation. The Prus[49] cost estimate is 9.74 = £16,984, which is nothing much but does show that some gardeners want pure grass lawns.

30. *Veronica persica*, common field speedwell, is a small annual herb This plant is a well-known and widespread agricultural weed. In Yorkshire it is found in "arable land, waste ground, roadside verges, disused railway land and gardens."[75] It is not spreading, having reached its geographical limits, nor is it found naturalized among native vegetation.[9] The Prus[49] cost estimate is 17.41 = £36,397,112, a large figure that reflects its importance as an agricultural weed.

6.4.3 Species summary

Remarkably few of these 30 species are widespread and serious pests. *Avena fatua*, *A. sterilis*, and *Veronica persica* are important agricultural weeds. *Aegopodium podagraria* is a serious garden weed. *Acer pseudoplatanus*, *Impatiens glandulifera*, and *Rhododendron ponticum* can be major pests in woodland. *Fallopia japonica* and *Heracleum mantegazzianum* are the only two named in the Wildlife and Countryside Act of 1981 and are serious pests in some places, particularly riversides and urban areas. That completes the list of those that are, at present, of national importance in terms of impact and cost (just nine species).

With the Prus[49] costs, only nine again score higher than 10, that is, have an estimated annual cost of more than £22,000. The bulk of the Prus costs, 98.6% of them, comes with the two *Avena* spp. and *Veronica persica*, the arable weeds. The total Prus cost for those three is £107 million. But, as noted under *Avena fatua*, some native species cost even more. The remaining species with a Prus cost of more than 10 are those listed in the previous paragraph less *Heracleum mantegazzianum* and *Impatiens glandulifera*, but with the addition of *Matricaria discoidea* (occasionally a minor agricultural weed) and *Galinsoga parviflora* (a garden weed of restricted distribution).

Nevertheless, many of the species are spreading quite fast, as will be quantified in Section 6.5.3. For instance, *Crassula helmsii*, a weed of ponds and similar areas, is causing concern because of its unconstrained origin from aquarists and the difficulty of controlling it, while *Buddleja davidii*, at the moment one of the numerous aliens of waste and derelict land, may become an important environmental weed as it spreads.

Many other species, both among the 30 and in general, can be difficult pests in some circumstances. The importance of these need to be looked at in the flora as a whole, and by comparing native and alien species, which is the focus of the next section.

6.5 Overall estimates of impact and cost

By most measures, the cost or impact, species for species, is about the same for natives and aliens[70] in the British Isles. Here I consider in varying detail the distribution of such impacts over the British flora.

6.5.1 Abundance

Abundance is a basic element in the impact of any species. Unfortunately, abundance is difficult to measure with plants because of the variety of life forms and phenotypic plasticity, while vegetative reproduction can cause problems in deciding what unit to use. The use of biomass, which might seem to be the obvious common measure, presents great difficulties, because so much of it is underground.

The only extensive published survey I have been able to find that relates to abundance is the one done by what was the Unit of Comparative Plant Ecology,[28] the Sheffield Survey II. This survey recorded the presence and absence of each species in 1-m^2 quadrats. It also recorded presence in 10-cm squares within the quadrats. That finer measure has been called abundance[61] but is really gregariousness.[28,71] The 1-m^2 samples were taken in a way that can be "loosely described as a stratified random sampling scheme."[70]

It is well known that plotting the logarithm of abundance against the rank of the species gives a lightning-strike curve. This is often called a diversity dominance curve. That such a curve is shown by the Sheffield Survey (Figure 6.2) is consistent with my view that it primarily measured abundance. In Figure 6.2, I have distinguished three categories:[28] native, planted, and introduced. The planted category includes both those not native to the Sheffield region but British natives, and Sheffield region natives whose abundance has been increased by planting. It can be seen that all three categories follow the same distribution. There is no significant difference among them. Overall, aliens in Britain have the same abundance distribution as natives, and so to that extent incur the same cost, species for species.

6.5.2 Range size

The range of aliens is the one collective character that distinguishes them from natives; aliens have, statistically, smaller range sizes.[70] This is so whether casuals are included or not, although casuals, as might be expected, have smaller range sizes than established species. When considering cost, casuals can be almost entirely disregarded.

The distribution of range sizes typically follows a logit-normal[74] distribution. When plotted as a diversity dominance curve, this usually gives a simple convex curve, as can be seen for British natives in Figure 6.3. As far as I know, such a plot has not been published before, and I call it an area dominance curve. It is the plot of the logarithm of the range of each species against its rank. The data in Figure 6.3 are the occurrence in hectads (10-km × 10-km grid squares) for the species in the Ecological Flora Database.[24]

It can be seen in Figure 6.3 that British aliens, non-natives, are much less widely distributed and have a distinct turn-up in the curve at the left-hand side. That is, they show a curve more like a typical abundance (diversity dominance) curve. The reason probably is that many of them are still spreading, as will be discussed in the next section. But whatever the reason, Figure 6. 3 shows non-native British plants have, on average, a

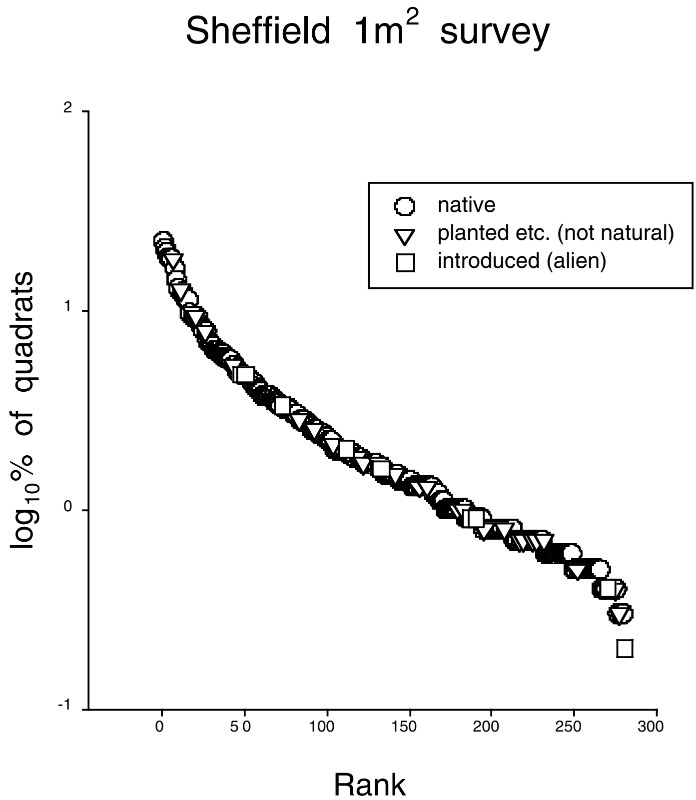

Figure 6.2 Dominance diversity plot for 1-m^2 quadrat records of the Sheffield Survey II,[28] showing native, planted, and introduced species simultaneously. Note the logarithmic scale of the ordinate.

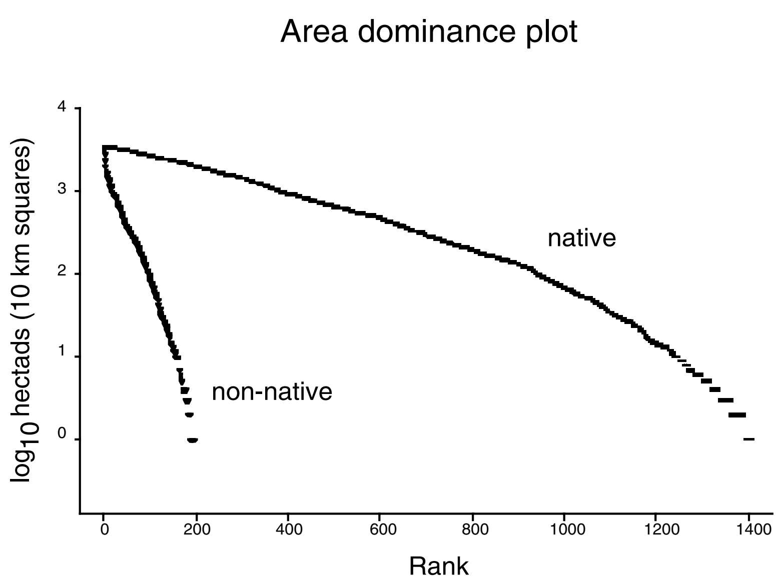

Figure 6.3 The first published example of an area diversity plot. This is for hectad records from the Ecological Flora Database.[24] Note the logarithmic scale of the ordinate.

much more restricted range than natives, and thus a lesser impact by this measure. This will tend to make their costs less than those of comparable native species.

6.5.3 Rate of spread

One reason why aliens are more narrowly distributed than natives could be that they are still spreading and haven't yet reached their full range. This spread was shown for three *Impatiens* species by fitting logistic curves to vice-county records.[47,69] (Vice-counties are subdivisions of civil counties to give roughly equal areas; they average 2200 km^2.) All three *Impatiens* spread from the first half of the 19th century and were expected to reach their full range early in the 21st century. If that were typical, and since most aliens were introduced in the 19th century or later, many aliens would still be spreading and so have misleadingly small recorded ranges. Only those introduced early or that had fast rates of spread, faster than the *Impatiens* spp., would be expected to have reached their maximum range.

A period of 100 to 200 years for an alien plant to reach its range limits in the British Isles is not surprising. Forest trees after the last glacial took 1000 to 2000 years.[69] The order-of-magnitude difference shows the magnitude and effectiveness of human dispersal, usually accidental, in spreading aliens. But the period is sufficiently long to make it difficult to compare rates of spread among different alien plant species.

There has been only one pair of extensive surveys that allow testing of whether aliens have been spreading. The pair were the distribution surveys of 1952–60[44], the Atlas, and 1987–88, the Sample Survey.[42] The latter was intended partly to assess the changes in roughly 30 years and was deliberately a sample survey. Both surveys were based on the 10-km × 10-km squares, or hectads, of transverse mercator grids. The first survey tried to be complete. The second was based on systematic sampling of one such square in every three in both dimensions, hence one in nine (except in coastal regions). The samples were taken systematically, so they were in a sense a sample survey of areas of 900 km^2, about 40% of the area of a vice-county. Within each sampled hectad, only three tetrads (2 km × 2 km) were studied, but intensively. The effect was that the second survey was marginally the more efficient, except for very rare or local species.

Unfortunately, the organizers[51,52] of the Sample Survey and the editors of the report became overconcerned about the statistical validity of the comparison between the two surveys. There are, of course, sampling errors and biases in both. There are in any large survey, and more effort was made to control them in the second survey than the first. But records of readily found and recognizable taxa, the bulk of the flora, can certainly be compared. The statements "a statistical comparison is considered to be inappropriate" and "differences between the two datasets … make valid comparisons extremely problematical"[42] seem quite unnecessarily cautious. This gloom seems to be the result of wishing both surveys to be comprehensive rather than samples. Treating both as sample surveys[36] allows many statistical comparisons.

The Sample Survey[42] maps show, for each of 1553 taxa (330 of them aliens), in which of the sampling survey hectads the taxon was recorded in the first (Atlas) survey, in the second (Sample) survey, or in both. There are 318 such hectads in Britain, 110 in Ireland, 428 in total. From those maps it is straightforward, if tedious, to count the records in each survey. For taxa that have not changed their distribution, if the surveys had been equally efficient, then the relationship of the two totals would not be significantly different from 1:1 and can be tested by χ^2, a process familiar to all who have done some simple Mendelian breeding.

That was done for the 30 species discussed above, as was indicated in some of the accounts. The results for the set are shown in Figure 6.1, where I have dated them 1958 (when the field work was largely complete) and 1988 (when the Sample Survey was finished). It can be seen that 10 species have a non-significant spread, have probably not changed range, in this 30-year period; six are significant at the 5% level (*); two at the 1% level(**); and twelve at the 0.1% level(***). The main oddity is *Avena fatua*, an ancient invader, but one that is apparently spreading with *** significance. It is known to have become more common through the change of agricultural practices, and the map is perhaps better interpreted as meaning just that: a marked change in abundance leading to an increase in records.

The four that have spread fastest in this 30-year period are, in rank order, *Epilobium ciliatum*, *Heracleum mantegazzianum*, *Elodea nuttallii*, and *Buddleja davidii*, as can be seen, more or less, in Figure 6.1. But such figures cannot be used to rank the species in order of their spreading potential. The ones that are not now spreading may include the fastest spreaders, ones that have reached their ecological limit relatively quickly. An arbitrary 30-year period for species that have been introduced at widely different times cannot be used to get comparative figures on dispersal ability. From the six most widespread but not statistically significant species, it is possible to examine the 1:1 assumption. That could also be expressed as 50:50. Taking the apparent change in these six species gives 48.8 to 51.2, which is probably a measure of the efficiency of the two surveys and is so close to 50:50 as not to affect the significance levels importantly.

So I would count 16 of the species as definitely expanding their ranges between 1958 and 1988: the 11 *** species other than *Avena fatua*, both ** species, and, in the * set, *Avena sterilis*, *Gunnera tinctoria* (both spreading from small ranges), and *Senecio squalidus* (clearly still spreading in Scotland and Ireland). That leaves just three that are significant at the 5% * level that may or may not really be spreading, as the statistical test is a crude one. They are *Crepis vesicaria*, *Conyza canadensis* (which may well be starting to invade Ireland), and *Elodea canadensis*. Out of the 26 for which I feel confident of their status, 16, or 62%, are spreading. The range of the rest seems more or less static. Natives, in contrast, are both spreading and shrinking their ranges as different species respond individually to climatic and land-use changes.[60]

If around three-fifths of British established aliens are still spreading, it means that comparisons of geographical range are biased against them, and that estimates of present costs underplay what the future cost will be.

6.5.4 Perceived weediness, abundance as weeds, and cost of control

I have used three measures of the impact of plants as weeds[70]: the perception of 49 species of annuals by a set of scientists[46]; the rank of incidence of dicotyledonous weeds[56] on English farms, which appears to be a measure of abundance; and an estimate of the economic cost, a weed cost, for all the British flora.[49] All three show the same major pattern when considering aliens: There are no important differences in the distribution of impact of natives and aliens. So here I only want to present the third, the Prus[49] cost (see Figure 6.4).

Figure 6.4 shows the results for all species with a score more than 10; that is, with a cost of more than £exp(10) = £22,026. Prus[49] modeled his full results with a three-parameter function

$$p(x) = 1 - (\exp(-(x/b)^c)^d)$$

where p(.) is the cumulative probability function, x the weed cost of each species, and b, c, and d the three parameters. This Prus function has the shape of a dominance diversity curve. The exponential nature of this function, or equivalently the logarithmic abscissa of Figure 6.4, explains why so few species contribute nearly all the cost, as was noted above in Section 6.4.2, 4, *Avena fatua*, and Section 6.4.3.

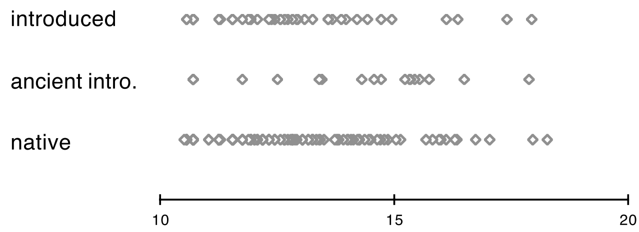

Figure 6.4 The distribution of Prus[49] weed costs over a score of 10 for three categories of British plants. Note that the abscissa is in natural logarithms of pounds sterling.

6.6 Conclusion

The impact of British non-indigenous plants or aliens is, species for species, much the same as the impact of British natives when the impact is measured by abundance or by weediness. The range of British aliens is, as a statistical distribution, less than that of natives, but this is partly because many aliens are still spreading and have yet to reach the limits of their distribution. Costing these impacts is difficult, but there seems little doubt that major costs come from nine or fewer species. The weed control costs of Prus come to more than £100 million, these being essentially agricultural costs. The environmental costs are very much more uncertain, but seem likely to be less. This suggests a total cost of aliens in the British Isles of £200 million to £300 million. Other indications are that the adverse costs of native species are about twice that. Aliens are costly; natives are more so.

Acknowledgments

I am very grateful to Richard Abbott (University of St. Andrews), Humphry Bowen (Dorset), Wendy Bunny (National Trust), Steve Compton (Leeds University), Alastair Fitter (University of York), Allan Hall (University of York), John Harvey (National Trust), Ray Hawes (National Trust), Stephen Jury (University of Reading), Duncan Stevenson (National Trust for Scotland), and Michael Usher (Scottish National Heritage) for much advice and information.

References

1. Abbott, R.J., et al., Hybrid origin of the Oxford Ragwort, *Senecio squalidus* L., *Watsonia*, 23, 123–138, 2000.
2. Bailey, J.P. and Conolly, A.P., Prize-winners to pariahs — A history of Japanese knotweed s.l. (Polygonaceae) in the British Isles, *Watsonia*, 23, 93–110, 2000.
3. Boyd Watt, H., Notes on the introduction and distribution of *Cymbalaria muralis* Gaertn., Mey. & Scherb. in Scotland (written ca. 1932). *Proceedings of the Botanical Society of the British Isles* 2, 123–125, 1957.
4. Brock, J.H., et al., The invasive nature of *Fallopia japonica* is enhanced by vegetative regeneration from stem tissues, in *Plant Invasions: General Aspects and Special Problems*, Pyšek, P., et al., Eds., SPB Academic Publishing, Amsterdam, 1995, pp. 131–139.
5. Carroll, L., *Through the Looking-Glass and What Alice Found There*, Macmillan, London, 1872.
6. Child, L.E. and Spencer-Jones, D., Treatment of *Crassula helmsii* — a case study, in *Plant Invasions: General Aspects and Special Problems*, Pyšek, P., et al., Eds., SPB Academic Publishing, Amsterdam, 1995, pp. 195–202.

7. Child, L., Wade, M., and Wagner, M., Cost effective control of *Fallopia japonica* using combination treatments, in *Plant Invasions: Ecological Mechanisms and Human Response*, Starfinger, U., et al., Eds., Backhuys, Leiden, 1998, pp. 143–154.
8. Clapham, A.R., Tutin, T.G., and Moore, D.M., *Flora of the British Isles, 3rd ed.*, Cambridge University Press, Cambridge, 1987.
9. Clement E.J. and Foster, M.C., *Alien Plants of the British Isles*, Botanical Society of the British Isles, London, 1994.
10. Collingham, Y.C., et al., Predicting the spatial distribution of non-indigenous riparian weeds: issues of spatial scale and extent, *Journal of Applied Ecology* 37 (Suppl 1), 13–27, 2000.
11. Compton, S.G. and Key, R.S., Species Action Plan: Lundy Cabbage (*Coincya wrightii*) and its associated insects, English Nature, Peterborough, 1998.
12. Compton, S.G. and Key, R.S., *Coincya wrightii* (O.E. Schultze) Stace (*Rhyncosinapis wrightii* [O.E. Schultze] Dandy ex A.R. Clapham), Biological flora of the British Isles, *Journal of Ecology*, 88, 535–547, 2000.
13. Compton, S.G., et al., Control of *Rhododendron ponticum* on Lundy in relation to the conservation of the endemic plant Lundy cabbage *Coincya wrightii*, English Nature Research Reports 263, 1–67, 1998.
14. Crawley, M.J., What makes a community invasible? Symposia of the British Ecological Society 26, 429–453, 1987.
15. Crompton, C.W., et al., Preliminary inventory of Canadian weeds, Technical Bulletin 1988–9E, Agriculture Canada, Ottawa, 1988.
16. Cross, J.R., Rhododendron ponticum L. (Biological Flora of the British Isles). *Journal of Ecology*, 63, 345–364, 1975.
17. de Waal, L.C., Treatment of *Fallopia japonica* near water — a case study, in *Plant Invasions: General Aspects and Special Problems*, Pyšek, P., et al., Eds., SPB Academic Publishing, Amsterdam, pp. 203–212, 1995.
18. Dickson J.H., Plant introductions in Scotland, in *Species History in Scotland*, Lambert, R.A., Ed., Scottish Cultural Press, Edinburgh, 1998, pp. 38–44.
19. Dony, J.G., Jury, S.L., and Perring, F.H., English Names of Wild Flowers, 2nd ed., Botanical Society of the British Isles, London, 1986.
20. Druce, G.C., The Flora of Oxfordshire, Parker, Oxford, 1886.
21. Eno, N.C., Clark, R.A., and Sanderson, W.G., Eds., Non-native Marine Species in British Waters: a Review and Directory, Joint Nature Conservation Committee, Peterborough, 1997.
22. Everett, S., Conservation news: Introductions and genetic conservation, *British Wildlife*, 11, 450–451, 2000.
23. Ferris, C., King, R.A., and Gray, A.J., Molecular evidence for the maternal parentage in the hybrid origin of *Spartina anglica*, *Molecular Ecology*, 6, 185–187, 1997.
24. Fitter, A.H. and Peat, H.J., The ecological flora database, *Journal of Ecology*, 82, 415–425, 1994.
25. Gilbert, O., Japanese knotweed — what problem? *Urban Wildlife News*, 11(3), 1–2, 1994.
26. Green, D., Japanese knotweed, letter in The Times (London), July 1, 2000.
27. Greenwood, E.F., Vascular plants: a game of chance? in *Ecology and Landscape Development: A History of the Mersey Basin*, Greenwood, E.F., Ed., Liverpool University Press, Liverpool, 1999, pp. 195–211.
28. Grime, J.P., Hodgson, J.G., and Hunt, R., *Comparative Plant Ecology*, Unwin Hyman, London, 1988.
29. Gritten, R.H., *Rhododendron ponticum* and some other invasive plants in the Snowdonia National Park, in *Plant Invasions: General Aspects and Special Problems*, Pyšek, P., et al., Eds., SPB Academic Publishing, Amsterdam, 1995, pp. 213–219.
30. Halliday, G., *A Flora of Cumbria*, Centre for North-West Regional Studies, University of Lancaster, Lancaster, 1997.
31. Hickey, B. and Osborne, B., Effect of *Gunnera tinctoria* (Molina) Mirbel on semi-natural grassland habitats in the west of Ireland, in *Plant Invasions: Ecological Mechanisms and Human Responses*, Starfinger, U., et al., Eds., Backhuys, Leiden, 1998, pp. 195–208.
32. Hiscock, S.J., Genetic control of self-incompatibility in *Senecio squalidus* L. (Asteraceae): a successful colonizing species, *Heredity*, 85, 10–19, 2000.

33. Jones, E.W., Acer. Biological flora of the British Isles, *Journal of Ecology*, 32, 215–252, 1945.
34. Kent, D.H., *Senecio squalidus* L. in the British Isles — 1, early records (to 1877), *Proceedings of the Botanical Society of the British Isles*, 2, 115–118, 1956.
35. Leach, J. and Dawson, H., *Crassula helmsii* in the British Isles — an unwelcome invader, *British Wildlife*, 10, 234–239, 1999.
36. Le Duc, M.G., Hill, M.O., and Sparks, T.H., A method for predicting the probability of species occurrence using data from systematic surveys, *Watsonia*, 19, 97–105, 1992.
37. Locke, G.M.L., Census of Woodlands and Trees 1979–82, Forestry Commission Bulletin 63, HMSO, London, 1987.
38. Lonsdale, W.M., Global patterns of plant invasions and the concept of invasibility, *Ecology*, 80, 1522–1536, 1999.
39. Lousley, J.E., *The Flora of the Isles of Scilly*, David & Charles, Newton Abbott, 1971.
40. Mabey, R., *Flora Britannica*, Sinclair-Stephenson, London, 1996.
41. Milne, R.I. and Abbott, R.J., Origin and evolution of invasive naturalized material of *Rhododendron ponticum* L. in the British Isles, *Molecular Ecology*, 9, 541–556, 2000.
42. Palmer, M.A. and Bratton, J.H., Eds., A Sample Survey of the Flora of Britain and Ireland. UK Nature Conservation 8. Based on a 1990 report for the Nature Conservancy Council by T.C.G. Rich and E.R. Woodruff, Joint Nature Conservation Committee, Peterborough, 1995.
43. Parker, I.M., et al., Impact: toward a framework for understanding the ecological effects of invaders, *Biological Invasions*, 1, 3–19, 1999.
44. Perring, F.H. and Walters, S.M., *Atlas of the British Flora*, Thomas Nelson and Sons, London and Edinburgh, 1962.
45. Perrings, C., Williamson, M., and Dalmazzone, S., Eds., *The Economics of Biological Invasions*, Edward Elgar, Cheltenham, 2000.
46. Perrins, J., Williamson, M., and Fitter, A., A survey of differing views of weed classification: implications for regulation of introductions, *Biological Conservation*, 60, 47–56. 1992.
47. Perrins, J., Fitter, A., and Williamson, M., Population biology and rates of invasion of three introduced *Impatiens* species in the British Isles, *Journal of Biogeography*, 20, 33–44, 1993.
48. Preston, C.D., The spread of *Epilobium ciliatum* Raf. in the British Isles, *Watsonia*, 17, 279–288, 1988.
49. Prus, J.L., New methods of risk assessment for the release of transgenic plants, Ph.D. thesis, Cranfield University, 1996.
50. Pyšek, P., On the terminology used in plant invasion studies, in *Plant Invasions: General Aspects and Special Problems*, Pyšek, P., et al., Eds., SPB Academic Publishing, Amsterdam, 1995, pp. 71–81.
51. Rich, T.C.G., Squaring the circles — bias in distribution maps, *British Wildlife*, 9, 213–219, 1998.
52. Rich, T.C.G. and Woodruff, E.R., Recording bias in botanical surveys, *Watsonia*, 19, 73–95, 1992.
53. Richardson, D.M., et al., Naturalization and invasion of alien plants: concepts and definitions, *Diversity and Distributions*, 6, 93–107, 2000.
54. Ryves, T.B., Clement, E.J., and Foster, M.C., *Alien Grasses of the British Isles*, Botanical Society of the British Isles, London, 1996.
55. Salisbury, E., Weeds and aliens, New Naturalist 43, Collins, London, 1961.
56. Schering Agriculture, *Weed Guide*, rev. ed., Schering Agriculture, Nottingham, 1986.
57. Simpson, D.A., A short history of the introduction and spread of *Elodea* Michx in the British Isles, *Watsonia*, 15, 1–9, 1984.
58. Stace, C., *New Flora of the British Isles*, Cambridge University Press, Cambridge, 1991.
59. Stace, C., *New Flora of the British Isles, 2nd ed*., Cambridge University Press, Cambridge, 1997.
60. Thompson, K., Predicting the fate of temperate species in response to human disturbance and global change, in *Biodiversity, Temperate Ecosystems, and Global Change*, Boyle, T.J.B. and Boyle, C.E.B., Eds., Springer-Verlag, Berlin, 1994, pp. 61–76.
61. Thompson, K., Hodgson, J.G., and Gaston, K.J., Abundance-range size relationships in the herbaceous flora of central England, *Journal of Ecology*, 86, 439–448, 1998.
62. Usher, M.B., Invasibility and wildlife conservation: invasive species on nature reserves, Philosophical Transactions of the Royal Society B 314, 695–710, 1986.

63. Vitousek, P.M., et al., Introduced species: A significant component of human-caused global change, *New Zealand Journal of Ecology*, 21, 1–16, 1997.
64. Wade, M., et al., *Heracleum mantegazzianum*: a problem for river managers in the Republic of Ireland and the United Kingdom, in *Plant Invasions: Studies from North America and Europe*, Brock, J.H., et al., Eds., Backhuys, Leiden, 1997, pp. 139–151.
65. Wadsworth, R.A., et al., Simulating the spread and management of alien riparian weeds: are they out of control? *Journal of Applied Ecology*, 37 (Suppl. 1), 28–38, 2000.
66. Webb, D.A., What are the criteria for presuming native status? *Watsonia*, 15, 231–236, 1985.
67. Williamson, M., Are communities ever stable? *Symposia of the British Ecological Society*, 26, 353–371, 1987.
68. Williamson, M., Invaders, weeds and the risk from genetically manipulated organisms, *Experientia*, 49, 219–224, 1993.
69. Williamson, M., *Biological Invasions*, Chapman & Hall, London, 1996.
70. Williamson, M., Measuring the impact of plant invaders in Britain, in *Plant Invasions: Ecological Mechanisms and Human Responses*, Starfinger, U., et al., Eds., Backhuys, Leiden, 1998, pp. 57–68.
71. Williamson, M., Can the impacts of invasive species be predicted?, in *Weed Risk Assessment*, Groves, R.H. et al., Eds. CSIRO Publishing, Collingwood Victoria, Australia, 2001, pp. 20–33.
72. Williamson, M. and Brown, K.C., The analysis and modelling of British invasions, Philosophical Transactions of the Royal Society B 314, 505–522, 1986.
73. Williamson, M. and Fitter, A., The varying success of invaders, *Ecology*, 77, 1661–1666, 1996.
74. Williamson, M. and Gaston, K.J., A simple transformation for sets of range sizes, *Ecography*, 22, 674–680, 1999.
75. Wilmore, G.T.D., *Alien Plants of Yorkshire*, Yorkshire Naturalists Union, Doncaster, 2000.
76. http://www.cabi.org/bioscience/Japanese_Knotweed_alliance.htm

chapter seven

Economic and environmental costs of alien vertebrate species in Britain

Piran C.L. White and Stephen Harris

Contents

0-8493-0836-4/02/$0.00+$1.50

7.1 Alien species, alien populations, and the process of invasions

Alien or introduced species are non-indigenous species that have been imported, have bred, and have become established in a particular region, either accidentally or deliberately. There are a variety of reasons for introduction of species to a new area, such as sport (shooting, fishing, hunting), amenity or ornament, food, domestication as pets, or importation for utilitarian purposes (livestock, fur, food). Manchester and Bullock provide examples of reasons for the introduction of selected alien species in the United Kingdom.[1] Whereas most invertebrate and microbe introductions worldwide have been accidental, most vertebrate and plant introductions have been intentional.[2] For vertebrates, there are a few exceptions to this generalization, most notably where species have been imported initially to satisfy demands for utilitarian or ornamental purposes.

Although some vertebrate species were introduced to Britain as early as the Iron and Bronze Ages, the majority of vertebrate introductions occurred during the late 19th and early 20th centuries. During that time there was a considerable interest in, and fashion for, "acclimatization," the history of which has been documented by Lever.[3]

There have been many attempts to understand the process of invasion by alien species, focusing on determinants of invasion success, the rate of spread of alien species, and the susceptibility of different environments to invasion. Williamson summarized the various issues surrounding biological invasions as a conceptual framework, in which he separated the invasion process into four stages: arrival and establishment, spread, equilibrium and effects, and implications.[4] In this chapter, we are concerned primarily with the third and fourth stages of the process, although — as will be illustrated by examples later on — these are affected considerably by earlier stages. The effect of invasion pressure (the number of individuals being introduced and the number of introductions) on the likelihood of establishment is of particular relevance, as are population parameters such as the intrinsic rate of increase and dispersal ability and, in some cases, climatic or habitat matching.

Two general rules that have emerged from the empirical observations of invasions by alien vertebrates are that islands are more susceptible to successful invasion than continental regions, and that simple communities with fewer species are more susceptible than more diverse communities.[5] Britain is an island with a relatively low-diversity vertebrate fauna, and should therefore be more susceptible to successful invasions by vertebrates than many other countries. Such invasions may be the result of natural processes (e.g. range expansion, such as that by the collared dove, *Streptopelia decaocto*, across Europe from the 1930s), but it is deliberate or accidental introductions by humans that concern us here.

For some indigenous species, their numbers in a particular region may have been enhanced, or perhaps replaced altogether, by translocations of non-indigenous individuals or populations. This may be the result of accidental escapes or deliberate releases of animals in captivity, or it may be deliberate reinforcement for the purposes of nature conservation. For the red squirrel (*Sciurus vulgaris*), red kite (*Milvus milvus*), white-tailed eagle (*Haliaeetus albicilla*), goshawk (*Accipiter gentilis*), and capercaillie (*Tetrao urogallus*), this has been done primarily for nature conservation purposes. For the goshawk and the capercaillie, additional reasons for enhancement were falconry and sport shooting, respectively. The capercaillie actually became extinct in the second half of the 18th century, but was reestablished successfully during the 19th century.[6] The goshawk and white-tailed eagle also show the same pattern of extinction followed by reestablishment, and the populations of all three species now extant in Britain are therefore completely distinct from the original native wild stock. The same is true for the reindeer (*Rangifer tarandus*) population on the Cairngorm plateau in Scotland. Other examples of the influence of introduced animals on native stocks are the release of 400,000 mallards each year for shooting[7] and the reinforcement of Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) populations for fisheries purposes.

Although the definition of an alien species *per se* is relatively clear-cut, the influence of alien animals may extend to populations of native species. Introductions of new genes are likely to have a significant impact on the genetic diversity of the native fauna, which is therefore a very important conservation issue in its own right. However, for the purposes of this review, we will confine ourselves to discussions of the impacts of alien species *per se*.

The impacts of alien species or populations within countries or regions can be grouped into five main categories: consumption of other species via predation or herbivory, competition with other species, introduction or maintenance of disease, interbreeding with native populations or species, and disturbance of the environment (physical or chemical). These impacts will in turn lead to reductions in global biodiversity through either species loss or interbreeding, and this is probably the most serious long-term effect of alien species. Impacts may be of either ecological or economic concern, or both, depending on which other species and environments are affected.

In this chapter, rather than focusing on individual case studies, we will take an impact-based approach to alien species in Britain and illustrate each type of impact with reference to particular introductions. In the main part of the chapter, we will concentrate on the costs imposed by alien species. However, alien species may also bring benefits, and these will be discussed in the final section. In the next section, we will first provide a species-based overview of the vertebrate introductions that have occurred in Britain. Where we use the term "introduced species," we take this to be the same as "alien species," and where appropriate we distinguish "introduced populations" in the same way, that is, as populations of native species that are in fact made up of introduced individuals. When we refer to Britain, we mean the mainland and surrounding small islands, but we exclude the Isle of Man, the Channel Islands, and Ireland. We also restrict our account to land vertebrates, by which we mean those animals that spend at least part of their life on the land surface above mean sea level. This therefore excludes the cetaceans that inhabit British waters and the leatherback turtle (*Dermochelys coriacea*), which is a seasonal visitor to British waters in the North Atlantic, where it feeds almost exclusively on jellyfish.[8] It also excludes all entirely marine species of fish. However, it includes seals and fish that migrate between fresh and marine waters.

7.2 Overview of alien vertebrate introductions in Britain

7.2.1 Mammals

A total of 22 mammal species that have been introduced and bred in Britain are currently extant in the wild (Table 7.1), and a further eight are now extinct. These figures do not include vagrant species. Sixty-five mammal species exist in breeding populations in Britain at the moment, and introduced species therefore represent 34% of the current mammal fauna in terms of species richness. If the terrestrial mammals only are included (this excludes two species of seals and 15 species of bats), introduced species account for 46% of the mammal species currently extant in Britain (Table 7.2).

Table 7.1 History, Population Status, and Significant Environmental and Economic Costs of Extant Introduced Vertebrates in Britain

Common name	Scientific name	Date of introduction	Population estimate	Population change	Reference(s) for population data	Economic costs	Environmental costs
Mammals							
Red-necked wallaby	*Macropus rufogriseus*	1850s on	26	–1	164		
Lesser white-toothed shrew	*Crocidura suaveolens*	Iron Age or earlier	14,000	0	9		
Rabbit	*Oryctolagus cuniculus*	Norman (1066-1154)	37,500,000	2	9	**Fh E**	**Fh**
Brown hare	*Lepus europaeus*	Roman	817,500	0	165		Fh
Grey squirrel	*Sciurus carolinensis*	1876 to 1930	2,520,000	2	9	**Fh** E	Fp **C**
Orkney and Guernsey voles	*Microtus arvalis*	Neolithic/Bronze Age	1,000,000	–2	9		
Harvest mouse	*Micromys minutus*	Post-glacial	1,425,000	–2	9		
House mouse	*Mus domesticus*	Iron Age or earlier	5,192,000	–2	9	**Fh D E**	
Common rat	*Rattus norvegicus*	1728-9	6,790,000	–2	9	**Fh D** E	
Ship rat	*Rattus rattus*	Roman (3 AD)	1300	–2	9	**D**	
Fat dormouse	*Glis glis*	1902	10,000	1	9	Fh E	
Feral ferret	*Mustela furo*	Norman or 14th C.	2500	0	9	D	**H**
Mink	*Mustela vison*	1930s	110,000	–1	9	Fh D	**Fp**
Feral cat	*Felis catus*	Norman	813,000	0	9		**Fp H**
Sika deer	*Cervus nippon*	1860s on	11,500	2	9	**Fh** D	Fh **H**
Fallow deer	*Dama dama*	Roman/Norman	100,000	0	9	**Fh** D **E**	**Fh**
Reeves' muntjac	*Muntiacus reevesi*	Early 1900s	40,000	2	9		**Fh**
Chinese water deer	*Hydropotes inermis*	1915	650	1	9		
Père David's deer	*Elaphurus davidianus*	1963 on	30	0	164		
Feral goat	*Capra hircus*	Neolithic	3565	0	9		Fh
Feral sheep	*Ovis aries*	Neolithic	2100	0	9		

Feral pig	*Sus scrofa*	1800s on	200	1	129	E	Fh E
Birds							
Night heron	*Nycticorax nycticorax*	1868 on	<50	0	166		
Black swan	*Cygnus atratus*	Unknown	<30	0	166		
Pink-footed goose	*Anser brachyrhynchus*	Unknown	<100	0	166		
White-fronted goose	*Anser albifrons*	Unknown	<100	0	166		
Bar-headed goose	*Anser indicus*	Unknown	85	0	167		
Snow goose	*Anser caerulescens*	Unknown	<200	0	166, 167		
Canada goose	*Branta canadensis*	Early 1700s	59,500	2	11	**Fh E**	Fh
Egyptian goose	*Alopochen aegyptiacus*	Late 1700s on	906	2	11, 167		
Muscovy duck	*Cairina moschata*	1980s	<300	0	11		
Wood duck	*Aix sponsa*	1870s	<100	2	11, 166		
Mandarin duck	*Aix galericulata*	1745	7000	2	11		
Red-crested pochard	*Netta rufina*	1937 on	<100	2	11		
Ruddy duck	*Oxyura jamaicensis*	1950s	3625	2	11, 166		**H**
Red-legged partridge	*Alectoris rufa*	1770 on	c. 350,000[a]	2	11, 168		
Pheasant	*Phasianus colchicus*	1040s on	3,100,000[b]	0	11		**H**
Golden pheasant	*Chrysolophus pictus*	1890s on	1500	2	11		H
Lady Amherst's pheasant	*Chrysolophus amherstiae*	1828 on	150	–2	11		

Table 7.1 (Continued) History, Population Status, and Significant Environmental and Economic Costs of Extant Introduced Vertebrates in Britain

Common name	Scientific name	Date of introduction	Population estimate	Population change	Reference(s) for population data	Economic costs	Environmental costs
Ring-necked parakeet	*Psittacula krameri*	1969 on	1508	2	169		
Monk parakeet	*Myiopsitta monachus*	1899 on	<20	0	166		
Eagle owl	*Bubo bubo*	Unknown	<10	0	166		
Little owl	*Athene noctua*	1842 and 1870s on	18,000	–1	11		
Reptiles							
Red-eared terrapin	*Trachemys scripta elegans*	1990s	<100	0	8		
Wall lizard	*Podarcis muralis*	1930s	800	1	8		
Aesculapian snake	*Elapha longissima*	1970	<100	0	8		
Amphibians							
Midwife toad	*Alytes obstetricans*	1900s on	<3000?	0	8		
Edible frog	*Rana esculenta*	1837 on	15,000	1	105		H
Marsh frog	*Rana ridibunda*	1935 on	15000	1	105		**Fp** H
Alpine newt	*Triturus alpestris*	1920s	<1000?	0	8		
African clawed toad	*Xenopus laevis*	1960 on	<500?	0	8		
Italian crested newt	*Triturus carnifex*	Thought to be 1960s	<2000?	0	8		H
Yellow-bellied toad	*Bombina variegata*	1954	<200?	0	8		
Bullfrog	*Rana catesbeiana*	1980s	<200?	1	8		
Fish							
Rainbow trout	*Oncorhynchus mykiss*	Unknown	<10	1	14		**Fp**
Pink salmon	*Oncorhynchus gorbuscha*	Unknown	Vagrant	–1	14		

Brook charr	*Salvelinus fontinalis*	Unknown	<10	1	14		H
Carp	*Cyprinus carpio*	Unknown	>30	1	14	E	**Fp** H
Sunbleak	*Leucaspius delineatus*	1990	<10	1	15		
Goldfish	*Carassius auratus*	Unknown	>30	1	14		
Bitterling	*Rhodeus sericeus*	Unknown	<30	1	14		
Ike	*Leuciscus idus*	Unknown	<30	1	14		H
Wels	*Siluris glanis*	Unknown	<10	1	14	Fp	Fp
Large-mouth bass	*Micropterus salmoides*	Unknown	<30	–1	14		
Pumpkinseed	*Lepomis gibbosus*	Unknown	<30	1	14		Fp
Rock bass	*Ambloplites rupestris*	Unknown	<30	–1	14		
Zander	*Stizostedion lucioperca*	1878	>30	1	14	**Fp**	**Fp**

[a] 2 million captive-bred birds are released each year for shooting.[168]

[b] 25 million captive-bred birds are released each year for shooting.[48]

Environmental and economic costs are represented as follows: Fh, feeding (herbivory); Fp, feeding (predation); C, competition; D, maintenance or introduction of disease; H, hybridization; E, environmental disturbance.

Significant impacts are highlighted in bold.

For mammals, birds, reptiles, and amphibians, population estimates are the estimated number of animals at the start of the breeding season.

For reptiles and amphibians, figures are our estimates based on information provided in the references listed.

For fish, population estimates are the number of stocks.

For mammals, amphibians, and reptiles, population changes in terms of numbers and/or range between 1965 and 1995 are represented as follows: 2, strong evidence of increase; 1, suggestions of increase; 0, probably stable; –1, suggestions of decrease; –2, strong evidence of decrease.

For birds, population changes in terms of numbers and/or range between 1968 and 1972 and 1988 and 1991 are represented as follows: 2, >25% increase; 1, 10–25% increase; 0, increase or decrease of <10%; –1, 10–25% decrease; –2, >25% decrease.

For fish, population changes in terms of numbers and/or range are represented as follows: 1, evidence of increase; 0, probably stable; –1, evidence of decrease.

Fish taxonomy is based on Wheeler.[171]

Table 7.2 Number of Extant Native and Introduced Species in Britain of the Different Vertebrate Groups

Vertebrate group	Total number of species	Number of introduced species	Introduced species as percentage of total species
Mammals			
All species	65	22	33.8
Terrestrial species	48	22	45.8
Birds	219	21	9.6
Reptiles	9	3	33.3
Amphibians	14	8	57.1
Fish	56	13	23.2

Terrestrial species of mammal exclude seals and bats.

Sources of reference are given in the text and in Table 7.1.

Some mammal species were introduced in the Iron and Bronze Ages and during the Neolithic period, and a few species were introduced by the Romans and the Normans. Introductions of mammals at these times were either accidental, such as the house mouse (*Mus domesticus*) and the common rat (*Rattus norvegicus*), or deliberate, with the introduced animals being used for wool and meat. This was the case for feral sheep (*Ovis aries*) and feral goats (*Capra hircus*) during the Neolithic period and for rabbits (*Oryctolagus cuniculus*) in Norman times. However, rabbits were initially farmed in warrens, and substantial increases in wild populations did not occur until the mid-18th century in England, and not until the early 19th century in parts of Scotland and Wales.[9]

The majority of mammalian introductions occurred in the late 19th and early 20th centuries, as part of the vogue for acclimatization. Many wild populations of alien species originated from escapes. In the majority of cases, these were escapes of species originally imported for ornamental purposes on private estates — for example, Reeves' muntjac (*Muntiacus reevesi*), sika deer (*Cervus nippon*), fallow deer (*Dama dama*), Chinese water deer (*Hydropotes inermis*), and the red-necked wallaby (*Macropus rufogriseus*). In some cases, escapes were of species originally imported for fur, such as mink (*Mustela vison*) and muskrat (*Ondatra zibethica*). Finally, a few species were liberated deliberately into the wild, such as the grey squirrel (*Sciurus carolinensis*) and the fat dormouse (*Glis glis*).

7.2.2 Birds

The categorization of birds as native or introduced species is more difficult than that of mammals, owing to their greater mobility and therefore uncertainty about what constitutes their "native" range. The situation is complicated further because many native populations, especially waterfowl, have been reinforced by introduced individuals. The British Ornithologists' Union Records Committee's British List does not classify species as "native" or "introduced," referring instead to species occurring in an "apparently natural state" (category A) and species that have derived populations from introduced stock (category C).[10] Species can therefore occur in more than one category in the British List. For the purposes of this chapter, we have included those species for which the bulk of the population stems from introduced animals, and excluded those species which are commonly regarded as native, although they may also have been reinforced by some introduced animals. We have also excluded any feral domesticated species that are originally descended from native wild stock (such as feral pigeon or rock dove, *Columba livia*) and those species for which there is no current evidence of breeding.

Twenty-one of the breeding bird species presently extant in Britain were introduced (Table 7.1), and at least another four previously introduced species are now extinct. A total of 219 bird species were recorded as breeding in Britain during 1988–91.[11] Introduced species therefore currently represent 10% of the British breeding bird fauna in terms of species richness (Table 7.2), although only eight of the extant species have populations in excess of 1000 individuals. In total, almost 300 non-native bird species have been recorded in the wild in Britain, and approximately 50 of these species have bred at some point,[10,12] but in most cases, populations have not become established.

The earliest introduction to the British bird fauna, the pheasant (*Phasianus colchicus*), was introduced from the Norman times onward initially as a source of food and later fulfilled the additional role of a hunting quarry. The red-legged partridge (*Alectoris rufa*) was also introduced generally as a game bird from the 1820s on, although Lever records that the species was first brought to Britain in 1673.[6] At least part of the reason for the introduction of the little owl (*Athene noctua*) was for biological control of small mammals. However, such functional introductions are the exception rather the rule for birds, and all of the other introduced species are the result of escapes from private collections or deliberate introductions for ornamental purposes.

7.2.3 Reptiles

Three reptile species known to be currently extant in Britain were introduced (Table 7.1), with at least a further three previously introduced but now extinct. The British reptile fauna totals nine species,[8,13] so introduced species constitute 33% of the British reptile fauna in terms of species richness (Table 7.2).

Of the six recorded introductions of reptiles to the British mainland fauna, the wall lizard (*Podarcis muralis*) and the green lizard (*Lacerta viridis*) were introduced deliberately for ornamental purposes. Three of the other species — dice snake (*Natrix tessellata*), European pond tortoise (*Emys orbicularis*), and red-eared terrapin (*Trachemys scripta elegans*) — may also have been introduced deliberately on some occasions. However, since all three have been regularly kept as pets at various times, the main form of introduction for all three species, in common with the Aesculapian snake (*Elapha longissima*), was probably escape.

7.2.4 Amphibians

Eight amphibian species known to be currently extant in Britain were introduced (Table 7.1), with at least five previously introduced species now being extinct. The British amphibian fauna totals 14 species,[8,13] so introduced species constitute 57% of the British amphibian fauna in terms of species richness (Table 7.2).

Most introductions of amphibians to Britain have been deliberate. These include the Italian crested (*Triturus carnifex*) and Alpine (*Triturus alpestris*) newts, African clawed toad (*Xenopus laevis*), edible frog (*Rana esculenta*), marsh frog (*Rana ridibunda*), yellow-bellied toad (*Bombina variegata*), fire-bellied toad (*Bombina bombina*), and painted frog (D*iscoglossus pictus*), the latter two species now being extinct.[6,8] However, it is probable that some populations may also have originated from accidental releases.

7.2.5 Fish

Thirteen fish species currently found in Britain are introduced (Table 7.1), and at least three further introduced species recorded by Lever[6] are now extinct. A total of 55 fish species are listed by Maitland and Lyle as being found as wild populations in Britain at present.[14] To this list should be added *Leucaspius delineatus*, which was introduced in the 1990s as a result of importation for ornamental purposes and subsequent escapes.[15] Introduced species therefore represent 23% of the fish fauna in terms of species richness (Table 7.2). Of all the vertebrate groups, fish have been the most influenced by introductions. This has been due to a combination of introductions of native species outside their natural range, and enhancement of populations of native species with captive-bred fish from a variety of origins, often unknown. As a result, the native ranges of many fish species are now obscured, and their genetic diversity irreparably compromised. The impact of introduced species on fish has therefore been greater than on any other vertebrate group in Britain and represents a major conservation issue.

The main reasons for the deliberate introduction of fish species are either for sport fisheries or for ornamental purposes, or in some cases both. Examples of species introduced predominantly for sporting reasons include the rainbow trout (*Oncorhynchus mykiss*), brook charr (*Salvelinus fontinalis*), carp (*Cyprinus carpio*), ide (*Leuciscus idus*), wels (*Siluris glanis*), and zander (*Stizostedion lucioperca*). An example of an ornamental introduction is the goldfish (*Carassius auratus*), some wild populations of which have emanated from fish breeding in lakes on private estates.[6] A few species appear to have been introduced accidentally. These include bitterling (*Rhodeus sericeus*), which was introduced due to its use as live bait for perch (*Perca fluviatilis*), as well as *Tilapia zillii*, a type of cichlid, and the guppy (*Poecilia reticulata*), which are both escapees from pet shops. Lever recorded that populations of *Tilapia zillii* and guppy were breeding in a stretch of the St. Helens Canal in Lancashire in the early 1960s, where the water was artificially heated by local industrial discharge.[6] Koi carp (*Cyprinus carpio*) are stocked extensively for anglers, with the present British rod-caught record standing at almost 20 kg.[16] The keeping of koi as pets is widespread as well, and it is likely that escapees from angling or pet stocks have also bred successfully in the wild, although there are no records of this at present.

7.3 Economic impacts of introduced species

7.3.1 Consumption of other species or crops

Of the introduced mammal species, the rabbit, grey squirrel, fallow deer, sika deer, and Reeves' muntjac are those which currently have the most significant economic impact on forestry or agricultural crops *in situ* via their feeding behavior. Putman and Moore presented data on the number of inquiries concerning deer damage to agricultural interests in lowland Britain received by the Wildlife and Storage Biology section of the Agricultural Development and Advisory Service (Ministry of Agriculture, Fisheries and Food) over the period January 1987 to March 1989.[17] These data are complicated by the fact that the native deer species, red (*Cervus elaphus*) and roe (*Capreolus capreolus*) deer, may also cause significant damage in certain areas. Most inquiries came from eastern England, where fallow deer were the primary concern, and the second highest number came from southwestern England, where inquiries were split fairly evenly between fallow, red, and roe deer. Overall, the number of inquiries relating to fallow deer was higher than for any other deer species, in particular in relation to damage to grass, cereal, and root crops. However, only 1% of all wildlife-related inquiries concerned deer of any species, which suggests that damage to lowland agriculture by deer is relatively insignificant compared with other wildlife species. In a separate study, Doney and Packer carried out a survey of deer damage to farms in four regions of England.[18] Damage to cereal crops was reported on 44% of the farms where deer were present. However, 85% of the respondents estimated the level of financial loss due to deer damage to be less than £500 per year. Langbein, in a survey of an area of southwestern Britain, found a median annual deer damage level of £500, equivalent to £4.50/ha/year, over all properties, equivalent to £10.30/ha/year if only those properties which actually reported damage were included.[19]

Deer can cause damage to coniferous plantation forestry by browsing the leading shoots of young trees, by stripping the bark from mature trees, or by rubbing their antlers against trees (known as "thrashing" or "fraying").[20] The most common forms of financial loss are the loss of incremental growth due to leader browsing, and the downgrading of wood produced, which can occur due to forking, marking, and staining as a consequence of fungal invasion, a problem that is exacerbated by deer damage to the bark.[21-23] Gill et al. calculated that the loss of value may be up to 8.4% in a worst-case scenario, where 25% of all stems are forked.[23] Ward calculated that the total yield was not significantly reduced until at least 40% of trees were forked.[24] However, his model also showed that loss of annual increment could not be tolerated if it exceeded 1 year. With 1 year of growth lost, the net present value of a stand was reduced from £512/ha to £112/ha, and a further year's growth loss resulted in a net loss of £288/ha. Deer damage therefore has the potential to be economically significant in forestry, and sika and fallow deer contribute to this damage. Reeves' muntjac is not a significant pest of commercial forestry, but it may be a pest in short-rotation coppice. Although the area of land under short-rotation coppice for biomass energy is small and localized at present, this may become a more significant problem in the future.

The relative impact of rabbit grazing has changed over time in correspondence with the impact of myxomatosis on the British rabbit population. Rabbit numbers in Britain were at their peak in the early 1950s. Following the introduction of myxomatosis in 1953, rabbit numbers fell by 99% within a few years. However, in less than two decades, evolutionary changes in the virus and the rabbits meant that an intermediate state developed whereby only around 50% of rabbits died from the disease,[25,26] and by the 1980s, this mortality rate had fallen further to about 20%.[27] Surveys in Scotland showed that 55.9% of farms had serious rabbit infestations prior to 1953, and that this was reduced to 1.5% by 1969–70 and then increased to 26.5% by 1991.[28] Rees et al. estimated that rabbit grazing caused national losses to cereal crops of about £40 million in the early 1980s,[29] and Kolb estimated that rabbits caused £11.8 million worth of damage to agriculture as a whole in Scotland during 1990–91.[28]

McKillop et al. used enclosures to quantify the effects of rabbit grazing on winter wheat at densities of between zero and 40 rabbits per ha.[30] They found that there was a yield loss of 1% per rabbit per ha. This was exhibited in terms of a reduction in the number of ears produced and the number of grains per ear rather than any difference in height at harvest. They calculated that this equated to a loss of about £6.50 per rabbit. Bell et al. quantified the effect of rabbit grazing on winter cereals under experimental conditions with varying rabbit densities up to 77 rabbits per ha to represent natural variation in the population throughout the year.[31] They found yield losses could be as high as 35% and that some cultivars of wheat were more susceptible than others. This equates to a loss of approximately 0.5% per rabbit per ha. Bell et al. also found the level of damage was not linearly related to rabbit density and suggested there may be a threshold damage level that determines the ultimate effect of grazing on crop yield.[31] These experiments are inevitably artificial, in that rabbits in enclosures do not have access to any alternative food sources, which would probably reduce their overall impact on the crops, especially at high rabbit densities when they would switch to more profitable food items. Nevertheless, the figures do represent maximum likely impacts on yield.

Bark-stripping damage by grey squirrels, whereby squirrels peel off the outer bark from trees before scraping off and eating the sap-filled phloem tissue beneath,[32] is of major concern to foresters and woodland managers. Trees can be killed outright when stripping results in complete ring-barking of the stem. Stripping high up can also lead to the death of the crown and deformation of the tree. Less extensive stripping can assist fungal penetration, either possibly killing the tree or, more commonly, significantly downgrading any timber products. Damage usually takes place in May, June, and July, with less frequent occurrences in August.[33] Broad-leaved trees are most commonly affected, the most vulnerable being sycamore (*Acer pseudoplatanus*, itself an introduced species), followed by beech (*Fagus sylvatica*), oak (*Quercus robur*), and ash (*Fraxinus excelsior*).[33] There is much inter-annual variability in damage levels, and damage itself is usually concentrated in small areas within stands. In a study of beech stands in southern England, Rowe found that 87% of stands reported squirrel damage, but that in 52% of these, less than 20% of the stems had been attacked.[34]

Damage also occurs on coniferous trees,[35] and this is of significant economic concern. Generally, less than 5% of damaged trees are killed.[36] Damage can occur on trees of all ages up to about 60 years old, but trees 10 to 40 years old are the most vulnerable. Squirrels also seem to concentrate on the most actively growing trees, such as those on the edges of a stand or the dominant trees within a stand.[33] However, despite a considerable investment in squirrel control in British forests, there have been no attempts to quantify the economic importance of squirrel damage. When red squirrels were common, they were also extensively culled to limit damage to commercial timber crops. There are no data to show that current economic losses to grey squirrels are significantly higher than previous losses to red squirrels, although it is likely that they are, since grey squirrels live at higher densities.[37]

The major economic impact in Britain caused by the feeding behavior of common rats and house mice is on stored food. It has been recorded that 53% of farm grain stores[38] and 33% of commercial grain stores can be infested with common rats.[39] Even at relatively high densities, rats and mice do not consume large quantities of food. However, damage to wheat sacks by common rats is much more important economically than direct consumption of the stored wheat.[40] Both common rats and house mice can also contaminate significant amounts of stored food with their excretory products. For example, a house mouse produces 50 or more droppings per day, which are costly to remove from stored foods.[41] Yet despite the continuing problem of damage to stored foodstuffs caused by commensal rodents, there have been no detailed analyses of the economic costs and benefits of rodent control.

Although ship rats (*Rattus rattus*) were previously important in Britain due to their consumption and contamination of stored foodstuffs, their very low population level now means that these impacts are economically insignificant. Similarly, from the 1950s until their extinction in 1989, coypu (*Myocastor coypus*) had a significant impact on root crops, in particular sugar beet.[42]

The only other introduced mammal that causes significant damage via consumption of crops is the fat dormouse. This species can cause significant visual damage to forestry crops locally in the Chilterns where it occurs. The favoured species is larch (*Larix decidua*), but pine (*Pinus* spp.), spruce (*Picea* spp.), and beech may also be damaged. The fat dormouse may also cause localized damage to orchard crops.[43] However, there has been no economic assessment of this damage, and it is very localized.

Mink have often been cited as an agricultural pest, and predation on domestic poultry, gamebirds, and fish stocks has been widely reported.[6,44-46] However, the occurrence of prey items derived from domestic animals in the diet of mink is rare, even in studies that have involved mink populations living in the vicinity of farm buildings and game-rearing pens.[47] Tapper reported evidence of up to 180 killings in a single night in pens containing 400 gamebirds,[48] and such mass killings can have an economic impact on commercial shoots if carried out near the time of release.[49] However, such incidents appear rare, and studies using scat analysis have shown that poultry and gamebirds generally make up less than 1% of the mink's diet, with the highest recorded being 5.4%.[50-52] Since a typical domestic laying hen was worth about £2 in the mid-1980s, Harrison and Symes concluded that most incidents of mink predation did not represent a serious financial loss.[49]

Fish farms and fisheries provide a potentially abundant food source for mink, and Chanin and Linn found that salmonid fish accounted for 34.2% of the mink's diet.[51] However, salmonids only feature significantly in the diet of mink feeding on rivers where salmon and trout make up a large proportion of the biomass. Mink damage was reported from 5% of fish farms in England and Wales during the mid-1980s,[49] but mink tend to take fish less than 20 cm long, and overall their commercial impact on fish farming is negligible.[53]

Of the introduced bird species, the Canada goose (*Branta canadensis*) is the only species that currently has a significant economic impact, because of its consumption and trampling of crops and its consumption, trampling, and fouling of amenity grassland. There are no data quantifying the level of this impact nationally, but localized damage can be significant. Simpson reported instances of crop damage in the U.K. costing £15,000, with 20% yield losses on winter cereals continuously grazed by Canada geese.[54] The population of Canada geese has increased dramatically in recent years, showing a proportional increase of 640% since the mid-1960s,[55] with populations in the U.K. currently growing at 8% per year.[56] This population increase has been accompanied by an increase in complaints of Canada goose damage to agricultural crops and grasslands.[55] The ring-necked parakeet (*Psittacula krameri*) has the potential to be a serious pest of orchards in southeastern England. At present the population is at too low a level for this to be a widespread problem, but damage may already be locally significant, and it is important not to become complacent.[57]

The only introduced reptiles, amphibians, or fish that have a potential economic impact through their consumption of other species or crops are the wels and the zander. These species can damage fisheries due to their predation on other smaller fish, such as roach (*Rutilus rutilus*), bream (*Abramis brama*), and ruffe (*Gymnocephalus cernuus*).[58] However, this problem is localized, and there have been no attempts to cost it in financial terms.

7.3.2 Competition with other species

No introduced mammals, birds, reptiles, amphibians, or fish in Britain currently have a significant economic cost due to their competing with native vertebrate species.

7.3.3 Introduction or maintenance of disease

The most important introduced mammals with significant potential for causing or maintaining disease with economic consequences in humans or other species are the commensal rodents: the ship rat, the common rat, and the house mouse. Although ship rats were historically very important due to their role as a vector in the transmission of bubonic plague (Black Death, *Pasteurella pestis*) in the Middle Ages, their low population densities today mean that this is no longer a significant threat. The major disease problems from introduced mammals in Britain therefore now come from the common rat and the house mouse.

Common rats can act as hosts of a number of diseases that can affect humans and livestock.[59] In Britain, the major ones are salmonellosis, leptospirosis, cryptosporidiosis, toxoplasmosis, and yersiniosis. *Salmonella* is commonly transmitted from pigs and poultry to humans, but house mice are often claimed to be the initial reservoir,[60] and Davies and Wray reported *Salmonella* in 48.7% of house mice from poultry units.[61] However, Pocock et al. tested 341 samples (fecal and intestinal samples from mice and environmental samples) from four mixed-agriculture farms in North Yorkshire and found none to be positive for *Salmonella*.[62] This study, along with that of Healing and Greenwood,[63] suggests that it is probably infected poultry that initially infect the mice with *Salmonella*, although the mice can then serve as a reservoir for reinfection of the poultry.[61,64]

Yersiniosis in humans is characterised by acute infection (pseudoappendicitis and enterocolitis) and immunological complaints. *Yersinia* spp. are typically isolated from 3 to 10% of wild rodents tested.[62] Pocock et al. isolated *Yersinia* from 23 of 354 samples tested across four mixed-agriculture farms and found prevalences in fecal and intestinal samples of 3.4% and 9.3%, respectively.

Cryptosporidiosis is another infectious gastroenteritis that affects humans and other mammals. Quy et al. found an average prevalence of 24% for *Cryptosporidium parvum* in farmland rat populations, and suggested that rats would be able to act as a potential source of infection for the disease on both livestock and arable farms.[65]

In all these studies, while it can be demonstrated that commensal rodents are acting as reservoirs for disease to a greater or lesser degree, the importance of their role in the transmission process, relative to other modes of transmission, is difficult to quantify. Thus it is currently impossible to quantify the relative economic impact of introduced rodents as reservoirs of disease in Britain.

Sika and fallow deer, feral ferrets (*Mustela furo*), mink, and common rats have been shown to carry bovine tuberculosis (TB) in Britain.[66] However, the prevalence of bovine TB found in these species (with the exception of ferrets) is significantly lower than that found in the badgers (*Meles meles*), a native species that is believed to represent the main wildlife reservoir for the disease.[66] The role of these introduced species in the maintenance of bovine TB, as well as other native species in which the disease has been found, remains unclear and is the subject of current research.

Sika, fallow, and muntjac deer, as well as feral sheep, feral goats, and common rats, can also act as carriers of foot-and-mouth disease, which broke out in British livestock during 2001. However, wildlife did not appear to play a significant role in this outbreak, either as a reservoir or as a vector of infection. There is no evidence that foot-and-mouth disease has reached the feral pig (*Sus scrofa*) population in southeastern England. If it did, this could constitute a much more significant threat than any other wildlife species, but the lack of a carrier state for the disease means that any long-term problem is unlikely.[67]

There is no evidence that any introduced species of bird, reptile, amphibian, or fish in Britain is acting as a significant reservoir of an economically significant infectious disease.

7.3.4 Interbreeding with native species

Sika deer hybridizing with red deer may reduce income from stalking due to smaller trophy heads and carcasses. However, there appear to be no other cases for mammals, birds, reptiles, amphibians, or fish where introduced species have led to significant negative economic impacts through their interbreeding with native species.

7.3.5 Disturbance of the environment

From the 1950s until their extinction in 1989, coypu caused local damage to agricultural interests in East Anglia due to their burrowing into the banks of watercourses, which at times caused flooding.[42] Grey squirrels, house mice, and fat dormice can all cause economic damage to property by disturbance to their environment. Fat dormice chew through electric cables, roofing felt, and ceiling plaster.[43] Grey squirrels also enter the loft spaces of buildings[68] and will do similar damage. The actions of grey squirrels, house mice, and fat dormice can occasionally cause fires. These can be financially devastating to individual properties, although the overall economic impact of such damage is probably negligible.

The feeding activities on agricultural crops of introduced mammals such as rabbits, fallow deer, common rats, and, locally, feral pigs and birds such as Canada geese can also cause physical damage to the crops, which will have economic implications. It is extremely difficult to isolate the effects of such disturbance from actual consumption of the crops, so the economic significance of this is unknown.

7.4 Environmental impacts of introduced species

7.4.1 Consumption of other species

The majority of introduced mammal species have significant environmental impacts through the effects of their consumption of other species. Fallow and sika deer, Reeves' muntjac, brown hares (*Lepus europaeus*), rabbits, feral goats, feral pigs, and, historically, coypu can all have significant environmental impacts on vegetation species through their herbivory. Whether or not grazing and browsing are viewed as beneficial or detrimental depends on the management objectives in place for a particular location at a particular time.

The rapid increase in numbers of both native and introduced deer in Britain (and in the rest of western Europe) since the 1950s has led to increasing impacts of deer browsing and grazing on conservation. Light deer grazing can be beneficial, in that it helps to maintain early successional stages of grassland and therefore hold back woodland encroachment.[69] Heavy browsing effectively reduces the structural diversity of woodland by removing the middle layer of regenerating trees, creating wood pasture conditions. While this is detrimental to small mammals and invertebrates, it may be beneficial to some birds of conservation importance such as wood warblers (*Phylloscopus sibilatrix*), pied flycatchers (*Ficedula hypoleuca*), and redstarts (*Phoenicurus phoenicurus*).[70]

These examples indicate that deer grazing can sometimes be beneficial to conservation. However, conservation objectives are generally geared toward woodland regeneration, when the effects of deer grazing and browsing will be detrimental. Heavy browsing of young saplings by deer at densities as low as five deer per km^2 can inhibit regeneration of natural woodland.[70-73]

Coppice woodland is particularly vulnerable to deer damage. Coppice regrowth can be delayed or occasionally completely inhibited by deer browsing, even at relatively low deer densities,[74-76] and this can sometimes have consequences for invertebrate conservation.[75] New plantations may also suffer quite heavy damage levels. Key et al. found damage rates by fallow deer to terminal shoots in broad-leaved plantations of up to 93%, with considerable variation between tree species.[77] In addition to impacts on trees, introduced deer may also damage the ground flora. This is particularly the case for Reeves' muntjac, and Cooke has attributed declines in bluebell (*Endymion nonscripta*) and dog's mercury (*Mercurialis perennis*) in Monk's Wood National Nature Reserve to grazing from Reeves' muntjac.[78,79]

Rabbits can also have a significant impact on woodland regeneration,[80] but more important, rabbit grazing can cause shifts in entire plant communities in grassland.[81-83] Because rabbits have such a dramatic effect on vegetation composition, failure to exclude rabbits can have considerable influence on the success of land restoration projects.[84]

Coypu had a dramatic impact on native vegetation in the Norfolk Broads in East Anglia when numbers were high during the late 1950s and early 1960s, and again in the late 1970s and early 1980s. They caused a massive decline in reed-swamp in the Broads[85] and a big reduction in their favored food plants, including cowbane (*Circuta virosa*).[86] The effects of herbivory by coypu have contributed to very significant changes in the Norfolk Broads ecosystem. However, the precise contribution made by coypu is impossible to quantify, since their effects were also linked with changes in phosphate and nitrate pollution, boating activity, a greater frequency of avian and fish diseases, and changes in management of the rivers and marshes.[87]

Brown hares may damage saplings or shrubs during hard winters when grazing is not available,[88] feral goats may damage young trees due to browsing and bark stripping,[89] and feral pigs may adversely affect the ground flora of woodlands. However, because of the generally much lower densities of these species, and the very localized populations of feral goats and pigs, the overall conservation impacts they have are relatively insignificant compared with those of the introduced deer species and rabbits.

Other introduced mammal species have had major impacts on native species via predation. Mink have been implicated in the decline of water voles (*Arvicola terrestris*), coots (*Fulica atra*), moorhens (*Gallinula chloropus*), and various nesting seabirds. The decline of water voles was the most dramatic of any native British mammal during the 20th century.[9] Many authors have cited mink predation as a cause of local reductions in water vole populations in various habitat types.[90-94] However, other factors, such as habitat change and increased drainage, have also contributed to the decline in water voles, and they were probably already in decline prior to the establishment of mink in many catchments.[95]

Many authors have recorded that mink predate on waterfowl,[6,51,96] but populations of most duck species are increasing, and there is little evidence to link mink predation with any specific population declines. For example, Halliwell and Macdonald found no significant correlations between mink and moorhen abundance in the Upper Thames catchment.[93] However, Ferreras and Macdonald, studying the same geographic area, found that mink took between 16 and 27% of adult moorhens and 46 to 79% of moorhen broods, and 30 to 51% of adult coots and 50 to 85% of coot broods.[97] The presence of mink was also found to adversely affect the breeding success of coots, although for moorhens the evidence was less clear.

The impact of mink on coastal ground-nesting bird populations is especially severe on islands. Observations have shown that where mink gain access to smaller colonies (fewer than 200 pairs) of terns or gulls, it is unusual for any chicks to fledge.[98] For example, between 1989 and 1995, mink predation of nests and chicks caused widespread breeding failures of whole colonies of black-headed gulls (*Larus ridibundus*), common gulls (*Larus canus*), and common terns (*Sterna hirundo*) on small islands off the west coast of Scotland.[99] Mink predation has also had an adverse effect on populations of ground-nesting birds on the islands of Harris and Lewis and in the Sound of Harris.[100,101] In the latter instance, lapwing (*Vanellus vanellus*) and redshank (*Tringa totanus*) were particularly badly affected.

Other introduced mammals that are reported as preying on native species include feral cats (*Felis catus*) and grey squirrels. The major impact of feral cats is on birds. In one village in central England, birds accounted for 35% of the prey items brought to homes by cats, and cats killed about one third of the total breeding population of house sparrows (*Passer domesticus*).[102] Although the cats in these instances were domestic pets rather than feral cats, the species can nevertheless have a considerable impact on native wild bird populations. However, there are few data available to quantify this impact accurately, and the distinction between feral cats and house cats is a blurred one. Although grey squirrels also predate bird eggs and nestlings, there is no evidence that the overall effect of this is significant.[68]

Among the birds, the only introduced species that has any proven significant environmental impact via its consumption of native plants is the Canada goose. Overgrazing by Canada geese can have a significant adverse effect on reed beds, salt marshes, and other vegetation.[103] It has been suggested that pheasants could have been instrumental in butterfly population declines, although any effects have not been proven.

The only introduced amphibian that could potentially have widespread impacts on native species via predation is the marsh frog. Observations in southeastern England in the late 1970s and early 1980s suggested that there had been no detrimental impact on native frogs and toads.[104] However, uncertainty remains, since the habitat where populations of marsh frogs first became established may not be that favorable to native amphibians. Moreover, the warm summers of the late 1980s encouraged populations to increase, and it is estimated that there are now about 30,000 adult green frogs (marsh frogs and edible frogs) in approximately 80 populations in southeastern England.[105] There may therefore be a more real threat to native amphibians if these populations continue to increase and spread.

Predation of native species is a cause for concern with some introduced fish species. Rainbow trout, carp, wels, pumpkinseed (*Lepomis gibbosus*), and zander all predate newt, frog, and toad larvae.[106] Wels and zander also feed on smaller fish such as bream and roach, and wels also feed on mammals and diving birds.[16] However, the overall impact of this predation on the populations of the native species is not known, and for fish such as bream and roach, the effects are more likely to be important from a fisheries rather than a conservation perspective.

7.4.2 Competition with other species

The only well-documented significant competitive interaction between an introduced and a native mammal is that between the grey and the red squirrel. There have been disagreements in the scientific literature regarding the extent to which direct competition from grey squirrels has caused the decline of the native red squirrel. However, the weight of evidence suggests that it is a major contributory factor in this decline. There is little evidence for direct interference competition between adult red and grey squirrels,[107] and the mechanism of exclusion seems to be feeding competition.[108] The red squirrel is a less efficient forager in deciduous woodland, and in this habitat especially, grey squirrels can live at higher densities and achieve faster breeding rates.[37,109,110] Removal of grey squirrels can lead to red squirrel recovery, and intensive trapping of grey squirrels since 1998 on the island of Anglesey off the coast of North Wales has allowed an expansion in both numbers and distribution of the red squirrel on the island.[111]

There are almost certainly other examples where introduced species are having an adverse competitive effect on native ones, but we probably do not know the impacts that most introduced species are having at present. The most abundant bird species is an introduced one, the pheasant, which has a huge biomass compared with all other bird species, and it is likely that it is having an ecological impact of some kind. However, good evidence for any other impacts is lacking at present.

7.4.3 Introduction or maintenance of disease

Introduced species frequently act as hosts for disease. However, apart from those diseases of economic importance discussed in a previous section, there is no evidence that introduced mammals, birds, reptiles, amphibians, or fish have brought in new diseases that are environmentally damaging. There is little evidence, either, that alien species are contributing significantly to problems of disease in native species, although there are some exceptions.* Some alien fish species introduced to supplement native stocks for angling

* Grey squirrels may act as reservoir hosts for parapoxvirus, which may have exacerbated red squirrel declines in some areas.[172]

are leading to increased disease,[1] and increased fish diseases due to artificial stocking may have had a role in changes to the ecosystem of the Norfolk Broads. There is also some evidence to suggest that parasitic worms spread by pheasants may be harmful to native grey partridges (*Perdix perdix*), exacerbating the damaging effects of modern agriculture on this species.[112]

7.4.4 Interbreeding with native species

Three introduced mammal species have adverse environmental impacts due to their interbreeding with native species. These are the feral ferret, the feral cat, and the sika deer. Feral ferrets are fully interfertile with the native European polecat (*Mustela putorius*)[113] and are widely kept. Escapes are frequent, and populations have been established on the Scottish islands of Mull, Lewis, Bute, and Arran, and also on the Isle of Man and the mainland. Native polecats have increased in abundance and have extended their range considerably since a low in the 1920s, spreading out from a stronghold in mid-Wales.[48,114] As native polecat populations continue to spread, they will undoubtedly encounter feral ferrets and interbreed with them. However, the extent to which this has happened already is unknown.

Feral cats are capable of interbreeding with native wildcats (*Felis silvestris*). Wildcats have shown a population recovery and have spread in northern Scotland since World War I, and the population now seems to be stable.[9] Problems of direct persecution and habitat loss have now been reversed or halted, and the major problem is hybridization with feral or domestic cats. Hybridization has probably been going on for several hundred years, and a small level of hybridization does not seem to have significant adverse effects on population persistence. Wildcats have traditionally been distinguished from feral cats on the basis of morphological characteristics.[115] However, recent work has suggested that rather than being a specific form, there may be a morphological cline among wild-living cats in Scotland, with the "wildcat" existing at one end of this cline.[116] This in turn suggests that species-level approaches to this particular hybridization problem may be ineffective, and that management is required at the population level instead.[117]

Sika deer are able to hybridize with native red deer, although where both species exist at good densities, or where they are reasonably well separated, hybridization seems rare.[118,119] Nevertheless, hybridization has occurred in many areas, and sika-like hybrids seem to be better competitors than red deer in dense woodland, and so may replace them in such habitats.[120] However, red deer populations in southwestern Scotland and most of England are not native, so hybridization of these populations may not be a significant conservation issue.[9]

The only hybridization problem within Britain caused by an alien bird species, as opposed to introduced populations, concerns pheasants. However, in this case the affected species was itself introduced. There has been a traditional emphasis on measuring the success of pheasant shoots according to their bag size, rather than the "quality" of the shoot, and this has led to as many as 20 million birds being reared in captivity and released each year to supplement the wild stock.[48] Captive-bred birds have much lower survival rates over the winter and a significantly lower breeding success, and the continual dilution of the wild stock therefore represents a risk to the persistence of the genes in the wild stock.[121] There is also some hybridization between pheasant and Lady Amherst's pheasant (*Chrysolophus amherstiae*).[6] However, stocks of the latter are very localized, and both species are introduced, so this does not represent a real conservation concern.

The other significant hybridization problem in birds associated with a species introduced to Britain concerns the ruddy duck (*Oxyura jamaicensis*). However, in this case the actual location of the problem itself is elsewhere in Europe. Ruddy ducks originating from the Wildfowl Trust's collection in Gloucestershire have spread abroad and have now been reported from 13 other European countries.[122] In Spain and Turkey, the ruddy duck hybridizes with native white-headed ducks (*Oxyura leucocephala*), resulting in fertile offspring, which may lead to genetic introgression and possibly even extinction of wild white-headed ducks in these countries.[123]

There are two potential hybridization problems between native and introduced species among the amphibians. The first is between the Italian crested newt and the native great crested newt (*Triturus cristatus*). Hybridization between the two species occurs freely around Newdigate in Surrey, where the only well-established Italian crested newt population occurs in Britain.[8] However, although the hybrids are viable, they have a low fertility. Beebee and Griffiths consider that these hybrids therefore amount to a dead end, and that this is one reason why the Italian crested newts have not spread significantly beyond the site occupied by this single population.[8]

The other amphibian hybridization problem is that between native pool frogs (*Rana lessonae*) and introduced marsh and edible frogs. It is by no means certain that the pool frog is in fact a native species,[124] but Beebee and Griffiths consider that there is sufficient evidence to indicate that it probably is.[8] Unfortunately, the last remaining population near Thetford in Norfolk declined to extinction in the mid-1990s.[8] Marsh, pool, and edible frogs are all capable of interbreeding.[8] Edible frogs are viable vertical hybrids of male pool and female marsh frogs. Matings between male and female edible frogs produce female marsh frogs, but few of these survive, because they have low viability and because the habitats occupied by edible and pool frog populations are not well suited to marsh frogs. For this latter reason, most matings of edible frogs that produce viable offspring are with pool frogs, and green frog populations in Britain — as elsewhere in western Europe — are therefore mixtures of edible and pool frogs.

A number of fish species hybridize with one another quite frequently in the wild,[16] and the extent of hybridization is increased considerably by movements of species, both native and introduced, by anglers. However, only three hybridizations out of 11 recorded to date involve introduced species, and one of these is between two introduced species rather than between a native and an introduced species. These hybridizations are native trout × brook charr, native bream × ide, and carp × native Crucian carp (*Carassius carassius*). There is some disagreement about the status of Crucian carp, but the most recent evidence suggests that it is a native species, having a natural range in the eastern and Midland counties, although it is only present elsewhere in Britain as a result of introduction.[125] In addition to these interspecies hybridizations, hybridization between farmed and wild fish of the same species may dilute the wild genetic stock and lead to long-term introgression of gene pools.[126]

7.4.5 Disturbance of the environment

Physical disturbance of the environment that is of environmental, rather than direct economic, importance can be caused by the rooting behavior of feral pigs. The effects of this seem to vary, and studies in different countries have shown both decreases and increases in ground flora diversity or regeneration as a result.[127,128] Where feral pigs occur in parts of Kent and East Sussex, their rooting behavior can damage the ground flora in native woodlands. However, they occur only in one area here and currently number only about 100 individuals,[129] so this damage is extremely localized. In the past, coypu and muskrat were both responsible for significant disturbance to riparian bank habitats by their burrowing behavior. However, both of these species are now extinct, the muskrat having been eradicated soon after the first escapes in the 1930s, and coypu by 1989.[130]

The action of common carp (and native Crucian carp) feeding on invertebrates in bottom muds and silts often increases water turbidity, and this can contribute to changes in freshwater ecosystems by reducing the amount of light available to aquatic plants.[16] However, these effects are often interlinked with increased nutrient loading and algal growth, so the relative importance of the carp themselves is difficult to quantify. Moreover, in studies of the eutrophication of whole ecosystems, other factors, such as phosphate and nitrate pollution and boating activity, appear to be the primary causes of changes in ecosystem structure and function.[87]

7.5 Analysis and conclusions

The proportion of alien species in the different vertebrate groups present in Britain varies quite widely (Table 7.2), being highest for amphibians (57%), then mammals (34%), then reptiles (33%) and fish (24%), and by far the lowest for birds (6%). If the non-terrestrial mammals are excluded, introduced species make up almost half (46%) of the terrestrial mammal fauna. The figures for terrestrial vertebrate groups are much higher than global averages, and reflect both the changing attitudes of humans toward introductions over time and the factors affecting the richness of native species.

There are two main factors that have combined to cause the relative paucity of native terrestrial vertebrate species (and partly in consequence, the high proportion of introduced species) in Britain today. The first factor is the island nature of Britain. This has ensured its relative inaccessibility to terrestrial species since the disappearance of the land bridge to continental Europe following the last glacial retreat, a fact that was exacerbated by the relatively short time period between the retreat of the ice sheet and the flooding of the English Channel.[131] The second factor is the effect of humans in causing extinctions of native species. In the next few sections we review the importance of these factors in shaping both the native and the introduced vertebrate fauna of Britain.

7.5.1 History of the British native vertebrate fauna

The native species in the current British terrestrial (land or freshwater) vertebrate fauna date back to 15,000 years ago, with the retreat of the last Ice Age in the Devensian period.[131] At the peak of glaciation, tundra-like conditions existed to the south of the ice sheet. It is extremely unlikely that any species of reptile or amphibian could have survived in such conditions.[8] However, it is possible that a few species of stenohaline fish (species that can survive only in fresh water) did survive in the ice-free conditions in the extreme south.[16] It is also possible that a few mammals — the mountain hare (*Lepus timidus*), the stoat (*Mustela erminea*), and the weasel (*Mustela nivalis*) — could have survived.[131] Nevertheless, it is clear that during this time the vertebrate fauna of what was to become Britain was extremely species-poor.

The glacial period ended abruptly about 13,000 years ago, and mean annual temperatures increased dramatically from –8° to +8°C. Another cooling followed almost immediately, reverting to tundra-like conditions again 10,000 years ago, but this was followed by another very rapid warming up to 8°C again.[132] At that time, Britain was still connected to continental Europe by a land bridge, across which flowed rivers that were either connected to or shared a flood plain with the Rhine.[16] However, as the climate warmed, the ice melted and sea levels rose, and Britain became isolated. This isolation may have occurred as early as 9500 years ago, but certainly by 7000 years ago.[131] Thus there was only a span of some 500 to 3000 years for terrestrial vertebrates and fish from warmer climates to the south to colonize Britain. For the mammals, excluding the bats and seals, only 33 species managed to cross into Britain before it was separated as an island. Recent genetic evidence suggests that some small mammal species colonized from the east (Scandinavia) rather than the south.[133] Britain's native mammal fauna is therefore probably a mixture of southern and eastern origins, and even fewer species may have colonized across the land bridge than was previously thought.

This short amount of time for colonization partly explains the relative paucity of Britain's native terrestrial vertebrate assemblage today, since it did not allow a full representation of the northwestern European vertebrate fauna to reach Britain. Thus the fish fauna of Britain (55 species) is much reduced compared with that of northwestern Europe (about 80 species) and Europe as a whole (about 215 species).[16] Euryhaline fish (those species that can live in both fresh and salt water) are relatively flexible in terms of their colonizing ability, and as the ice melted, these species were able to follow the coastline northward and colonize any fresh waters that were accessible from the sea. However, stenohaline fish had to colonize using rivers across the land bridge. The tight environmental constraints on movements of stenohaline fish, combined with the fact that the number of euryhaline species is relatively small, also gives rise to a gradient of native fish species richness across Britain. Areas farthest away from the former land bridge, such as Scotland, northern England, and Wales, have fewer native species than the south and east of England.[16]

Other terrestrial vertebrates are similarly constrained by the environment in terms of their colonization movements, and the native fauna of these groups is also relatively species-poor. For example, Britain's 23 reptiles and amphibians (which include 11 introduced species) compares with 22 species in the Netherlands, and more than 60 amphibians and twice as many reptiles in the whole of Europe.[8]

The mammal fauna in Britain was also formerly relatively species-poor, but the difference has been reduced by introductions, especially of larger mammals. In fact, the number of introductions *pro rata* is greater for mammals than for any other taxon. There are currently 63 species of mammals in Britain, excluding pinnipeds, compared with 188 in Europe.[134,135] The comparative figures for the different orders (with European figures in parentheses) are: insectivores, 8 (28); bats, 17 (35); primates, 0 (1); lagomorphs, 3 (8); rodents, 14 (68); carnivores, 11 (27), artiodactyls, 9 (20); and marsupials, 1 (1).

Due to their greater mobility and non-reliance on the land bridge for colonization, birds are significantly more abundant than other vertebrate groups in the British fauna, and there are proportionally more birds in the British fauna than are found globally.[136] The mobility of most bird species means that, given appropriate habitat and food resources, they are readily capable of extending their native ranges. For example, the crane (*Grus grus*) recolonized in 1981,[11] and further climate change may result in some bird species expanding their breeding ranges into parts of southern Britain from continental Europe in the future. Similar opportunities for natural colonization also exist in theory for bats. However, bats tend to be weaker fliers than birds, and populations of most species of bats are currently much reduced throughout northwestern Europe. Recolonization by species such as the extinct mouse-eared bat (*Myotis myotis*) will therefore remain very unlikely for the foreseeable future.

7.5.2 Human-induced extinctions of native species

Although natural conditions have therefore resulted in a relatively species-poor terrestrial vertebrate fauna, the effects have been exacerbated by extinctions, many of them as a direct result of the actions of humans. Previous extinctions of vertebrates and the influence of humans are most easily traced from the archaeological record for mammals.

Of the 33 mammals extant in Britain around the time of Britain's separation from continental Europe, 10 are now extinct. The reindeer and tarpan (*Equus ferus*) became extinct at about the time the land bridge disappeared. The root vole (*Microtus oeconomus*) became extinct about 5000 to 8000 years ago, elk (*Alces alces*) about 4000 years ago, and the aurochs (*Bos primigenius*) about 3000 years ago.[131] These early extinctions were all probably caused by a combination of habitat and climate change, although humans may also have contributed to the decline of the tarpan and the aurochs.[131] Brown bears (*Ursus arctos*) became extinct about 2000 years ago,[131] and this was probably the first extinction of a native species in which human persecution played the most significant role. Since that time, there have been further extinctions in which human persecution has been the overriding factor. These have been the lynx (*Lynx lynx*) about 2000 years ago (previously thought to be a much earlier extinction at about 4000 years ago), wild boar 700 years ago (although it was maintained after that date as a park animal), beaver (*Castor fiber*) 400 years ago, and the wolf (*Canis lupus*) 300 years ago. The numbers of species extinctions are similar for mammals and birds at 10 and 9, respectively,[136] but the proportion of mammals driven to extinction is relatively higher.

7.5.3 Effects of competition on the success of introduced terrestrial vertebrates

The relative species-paucity of native terrestrial vertebrate fauna, combined with the effects of extinctions, implies that Britain offers a potentially large number of vacant niches for introduced species to occupy. However, the fauna is dynamic, remnant species may increase in abundance in response to extinctions, and the influence of humans on these processes may also be considerable.

Maroo and Yalden compared the current mammal fauna of Britain with a hypothetical Mesolithic mammal fauna, which was based on extrapolation from pristine habitats existing elsewhere in Europe that would have provided similar conditions.[137] They calculated that there has been a 175-million-kg decrease in the biomass of wild mammals from 304 million kg in the Mesolithic to 129 million kg now. This is largely due to extinctions or large reductions in the abundance of large herbivores. Introduced species make up 68 million kg of the current fauna, and approximately 90% of this represents just one species, the rabbit. Of the mainland native mammals, only the badger, red fox (*Vulpes vulpes*), field vole (*Microtus agrestis*), mole (*Talpa europaea*), stoat, and water shrew (*Neomys fodiens*) have increased in abundance since the Mesolithic.

Therefore, although the success rates of alien species in Britain are not as high as those for Ireland, which was cut off by rising sea levels while still under ice and tundra,[4] they are nevertheless higher than expected. Moreover, analysis of the data on population trends shown in Table 7.1 indicates that of 67 introduced vertebrate species for which data are available, 30 are currently increasing in abundance or range, 25 are stable, and 12 are declining. Bird, amphibian, and fish invaders are doing especially well at present (23 increasing, 17 stable, and 5 decreasing), while mammalian invaders are doing less well on average (7 increasing, 8 stable, and 7 decreasing).

7.5.4 Costs of control and mitigation

The costs of damage *per se*, especially where the impacts are environmental rather than economic, are very difficult to quantify. Although there is some information on damage costs for a few introduced species, such information is missing for many more. However, the costs of control programs against a species can be used to give some idea of at least the minimum estimated current or potential future damage (Table 7.3).

In 1991–92, Forest Enterprise, the organization that manages state-owned forests in Britain, spent £1.22 million on shooting deer.[23] There are no more recent figures. In Scotland, between April 1999 and March 2000, 4041 sika deer were culled,[138] of which about 60% were culled by Forest Enterprise. The total cost of a Forest Enterprise ranger (including overhead, equipment, and capital costs such as vehicle maintenance and provision of larders for storing carcasses) is approximately £13 per hour. The effort required to cull

Table 7.3 Costs of Damage and Control for the Introduced Vertebrate Species in Britain that Currently Cause the Most Significant Environmental or Economic Costs

Common name	Scientific name	Summary of damage (Econ/Env)	Costs of economic damage	Costs of control
Mammals				
Rabbit	*Oryctolagus cuniculus*	**Econ, Env**	Agriculture: £100 million overall in 1980s (£5 per rabbit)[29] Cereal agriculture: £40 million losses to cereal production in 1980s in England[29] and £12 million in 1990/1 in Scotland[28]; 1% yield reduction per rabbit per ha and £6.50 cost per rabbit[30]; 0.5% yield reduction per rabbits per ha[31] Forestry: £90 million damage costs[146]	Overall: £30 million total management costs per year[146] Agriculture: >£1 million spent on rabbit control across the UK per year[147] Forestry: ranger costs incurred in rabbit control by Forest Enterprise throughout Britain in 1994/95[139] estimated at £529,000
House mouse	*Mus domesticus*	**Econ**	No estimates	Arable agriculture: 1,169,103 kg of rodenticide bait used in 1996[148] @ total cost of £10/kg = £11.2 million per year
Common rat	*Rattus norvegicus*	**Econ**	No estimates	Local authorities used 405,000 kg of rodenticide bait in 1997[149] @ total cost of c. £10/kg = £4.1 million per year
Grey squirrel	*Sciurus carolinensis*	**Econ, Env**	No estimates	Forestry: ranger costs incurred in grey squirrel control by Forest Enterprise throughout Britain in 1994/95[139] estimated at £334,000
Feral ferret	*Mustela furo*	Econ, **Env**	No estimates	No available data
Mink	*Mustela vison*	Econ, **Env**	No estimates	Little coordinated control, but Scottish Natural Heritage have earmarked £443,000 towards a £1.65 million scheme to eradicate mink from the western isles[170]
Feral cat	*Felis catus*	**Env**	N/A	Cats Protection spend £2 million[153] and RSPCA £600,000[152] annually on feral cat neutering and rescue

Table 7.3 (Continued) Costs of Damage and Control for the Introduced Vertebrate Species in Britain that Currently Cause the Most Significant Environmental or Economic Costs

Common name	Scientific name	Summary of damage (Econ/Env)	Costs of economic damage	Costs of control
Reeves' muntjac	*Muntiacus reevesi*	Econ, **Env**	No estimates	Forestry: 678 culled by Forest Enterprise throughout Britain in 1994/95[139] at estimated cost of £14,000
Fallow deer	*Dama dama*	**Econ, Env**	Agriculture: up to £10.30 per ha per year in southwestern England for all deer species[19]	Forestry: 2904 culled by Forest Enterprise throughout Britain in 1994/95[139] at estimated cost of £76,000
Sika deer	*Cervus nippon*	**Econ, Env**	Forestry: £400 per ha for 1-year growth delay due to deer browsing[24]	4041 sika deer culled in Scotland April 1999-May 2000 at total annual cost of £788,000 (based on 15 ranger hours per sika culled in high density populations @ £13 per hour). Venison return from this is £200,000, so net cost of sika deer cull in Scotland is £588,000 per year[138] Forestry: 1395 culled by Forest Enterprise throughout Britain in 1994/95[139] at estimated cost of £220,000
Birds				
Ruddy duck	*Oxyura jamaicensis*	**Env**	N/A	No coordinated control
Canada goose	*Branta canadensis*	**Econ,** Env	Cereal agriculture: instances of damage up to £15,000 and 20% yield losses[54]	Amenity grassland: cost of ameliorating trampling and fouling damage up to £40 per bird[56] 27 licenses issued for control (19 for shooting, 8 for egg oiling) May 1999-April 2000, resulting in 276 birds and 793 eggs destroyed[154]

Pheasant	*Phasianus colchicus*	**Env**	N/A	Shooting carried out for sport, which generates revenue, so no cost of control *per se*
Amphibians				
Marsh frog	*Rana ribidunda*	**Env**	N/A	No coordinated control
Fish				
Rainbow trout	*Oncorhynchus mykiss*	**Env**	N/A	No coordinated control
Carp	*Cyprinus carpio*	**Env**	N/A	No coordinated control
Zander	*Stizostedion lucioperca*	Econ, Env	No estimates	No coordinated control
Totals			**> £190 million**	**> £49 million**

Summary of damage taken from details in Table 7.1 (Econ, economic; Env, environmental), with nationally significant damage shown in bold type.

each sika deer will vary considerably with location, population density, and the methods of shooting (whether by day or by night under license). It will also vary with the age and sex of the deer. However, in high-density populations of between 20 and 70 sika deer per km^2, the average culling rate is approximately 1 sika culled per 15 man-hours. In densities between 2 and 20 deer per km^2, this rate would be considerably lower and in the range of 1 sika per 20 to 100 man-hours. However, since a smaller proportion of the total cull would be taken from lower-density populations, the rate of 1 per 15 man-hours can be used. Based on these figures, the cull of 4041 sika deer would have cost about £788,000. Although this is the total investment cost in control, about £200,000 of this was returned in the form of revenues from the sale of venison, so the net cost of sika control in Scotland in 1999–2000 would have been £588,000.[138]

To obtain data for control of the other introduced deer species, it is necessary to take an indirect approach. Toleman provided data on the total number of deer of different species culled in state-owned forests across Britain in the year 1994–95.[139] A total of 26,499 deer were culled, consisting of 9589 red, 1395 sika, 11,933 roe, 2904 fallow, and 678 muntjac. Rangers from Forest Enterprise spend 70% of their time carrying out crop protection duties, which includes management of wildlife in the forests. Of this 70%, an average of 59% is spent on deer culling, although this figure varies considerably between different regions. Gill et al. stated that Forest Enterprise spent an average of £238/km^2/year on rangers.[23] Including additional 100% overhead as above, the average total cost of rangers is £476/km^2/year. Based on these figures, the total cost of controlling deer over the 8350 km^2 of Forest Enterprise forests throughout Britain in 1994–95 was £1.64 million. It was probably between £2 million and £2.5 million in 2000–01.

Detailed data are not available on the relative effort put into the culling of different deer species in forestry. However, an approximation can be made using the above figures based on estimates of the relative time input required to kill an individual deer of each species. This will vary with factors such as population density, habitat, and the behavior of individual rangers, but it is still possible to make generalizations based on species ecology and behavior. For sika deer, it has been estimated that it takes an average of 15 hours to cull one deer.[138] Sika are particularly secretive deer, with a preference for dense forest and a relatively high effort required per cull. Red deer prefer slightly more open habitats and are less secretive and easier to cull. Youngson provides figures from which estimates of approximate mean rates of 7 hours per red deer (range 2.7–11.4) and 11 per sika (range 6.4–16.0) can be derived.[140,141] Fallow deer prefer slightly more open habitats than red and sika deer and often feed on rides within the forest. Roe deer are usually solitary or found in small groups, and they show a preference for younger forest stands, which makes them slightly more difficult to cull than either fallow or red. Muntjac are quite secretive, preferring dense cover, but are also inquisitive, and this, combined with high densities, means they are easier to cull than roe deer.

Generalizations of culling rates between species need to be treated with caution. Nevertheless, 5 hours per deer for roe, 2.5 for fallow, and 2 for muntjac in comparison with 15 for sika and 7 for red seem approximately realistic. Using these figures, together with data on regional variation in the number of deer culled and the area of Forest Enterprise-owned forests, the total amount spent on control of the different deer species in forestry in 1994–95 can be calculated. Averaging over the whole of Britain, in 1994–95, £705,000 was spent on red deer control in forestry, £220,000 on sika, £626,000 on roe, £76,000 on fallow, and £14,000 on muntjac. These figures will have risen since 1994–95, especially for sika (as confirmed by the more recent estimates above) and muntjac. Moreover, these estimates, based on labor costs and culling rates, gain support from the independently estimated data for sika culls above. In 1994–95, 3.3 times fewer sika were culled than in 1999–2000, and the calculated cost for 1994–95 was 3.5 times lower.

Deer damage to woodlands and forestry can also be prevented by fencing, but this is a costly option. It is also frequently ineffective in the long-term, since holes appear in the fence without continual maintenance, and deer can get over the tops of fences during times of heavy snowfall. Fencing can provide protection for between 10 and 15 years, depending on the level of maintenance.[142] Staines quoted a cost of around £3 million and a lifetime of around 20 years for 3500 km of deer fencing in Scotland, amounting to an annual outlay of around £500,000.[143] Four years later, Ratcliffe gave a cost of approximately £4/ha/year for deer fencing over 300,000 ha of Forestry Commission plantations in Scotland, giving an annual outlay of £1.2 million.[144] However, most of the large-scale deer fencing operations in forestry are primarily for protection against native red and roe deer, so it is impossible to determine the level of expenditure on fencing for introduced species from the data available. Deer fencing also carries indirect costs in that it results in increased mortality of capercaillie due to collisions with fences.[145] This represents a strong conservation reason, in addition to the financial ones, for fencing currently losing favor as a damage prevention strategy. Fences may also be used to exclude deer from nature reserves,[75] but the costs involved are minimal compared with those in forestry.

Total rabbit management costs in Britain currently amount to approximately £30 million per year.[146] This incorporates contracts for work, such as gassing at £400 per day, and also includes the installation and maintenance of 300 km of fencing at £3.40/m. Fencing alone therefore represents a total cost of £1.02 million. More than £1 million per year was spent on rabbit control by farmers during the period 1995–98.[147] The average expenditure per farmer on different control methods for rabbits was £190 on gassing, £100 on shooting, £90 on ferreting, £90 on cage trapping, and £550 on fencing.[147]

The cost of rabbit and grey squirrel control in forestry in terms of ranger time can be calculated on the same basis as for deer control, according to the proportion of ranger time spent on control of these species. In 1994–95, this was 13.3% for rabbits and 8.4% for grey squirrels.[139] Over all Forest Enterprise forests in Britain, ranger costs in 1994–95 were therefore £529,000 for rabbit control and £334,000 for grey squirrel control. These figures include overhead, capital, and general equipment costs associated with rangers, but exclude the cost of specific equipment such as fencing for rabbits. It should also be remembered that the costs of grey squirrel management are concentrated in the forests of England and, to a lesser extent, Wales.

Despite the huge industry associated with the production of rodenticides for the control of commensal rodents, no cost-benefit analysis of commensal rodent control in Britain has been published, and there are no data on the cost-effectiveness of rodenticide treatments. However, the amount of rodenticide used is considerable. In 1996, 1,169,103 kg of rodenticide bait was used by arable agriculture,[148] and 405,000 kg by local authorities.[149] The cost of rodenticide application will vary considerably according to whether it is done under contract or by individual farmers, the type of rodenticide used, and the extent of the rodent infestation. Catalogue prices for rodenticides range from £2/kg to £8/kg, averaging £4.50, and commercial pest control contracts vary widely from £200 to £2500, including bait. The amount of rodenticide bait used per contract also varies, but the average is probably about 20 kg. The lower limit for an average total cost including bait and application would be about £10/kg.[150] This gives a cost of commensal rodent control of at least £11.2 million for arable agriculture and £4.1 million for local authorities.

The most widespread form of feral cat control in Britain is trapping and neutering.[151] The cost of the Royal Society for the Prevention of Cruelty to Animals' feral cat neutering effort is £600,000 each year.[152] In addition, various other organizations invest considerable resources in neutering unwanted feral cats. The largest of these is Cats Protection, which spends £2 million each year rescuing and neutering cats, work that is mostly done by volunteers.[153]

There are no data specifically on the cost of Canada goose control. However, 27 licenses were issued for control (19 for shooting and 8 for egg oiling) between May 1999 and April 2000, which resulted in 276 birds and 793 eggs being destroyed.[154] There are also costs associated with amelioration of goose fouling of recreational areas, and one London park official estimated that the reinstatement of damaged grassland and cleaning of fouled paths cost £40 per bird.[56] For pheasants, shooting of birds is carried out for sport, which generates considerable revenue, so there are no costs of control *per se*.

There are no relevant data for introduced amphibians or fish, due to the lack of coordinated control efforts against these species, which also reflects the lower level of perceived damage caused by them.

7.5.5 British vertebrates and invasion theory

Some introduced species that have become naturalized in the British fauna have only done so through persistent introductions or reinforcement of existing populations. This emphasizes the fact that invasion pressure is an important element determining the success of introductions.[4] For example, although mink started escaping from fur farms in 1929, the first record of mink breeding in the wild was not until 1956.[9] The same is also true for pheasants, which are enhanced each year throughout much of lowland Britain by the release of captive-reared birds, as noted earlier. It is also the case for many fish populations. For example, carp, rainbow trout, and brook charr are among the many fisheries species for which populations are continually artificially enhanced.[16] The current British muntjac population stems from no more than eight different maternal lines, and in the two decades following the first releases in 1901, spread was slow and populations were small and very localized.[155] However, spread in the second part of the 20th century was much quicker due to further deliberate and accidental releases, frequently outside the existing main range.[155]

Of the extant introduced vertebrate species, 11 out of 22 mammals (50%), 3 out of 21 birds (14%), 1 out of 8 amphibians (25%), none of 3 reptiles, and 3 out of 13 fish (31%) have had significant environmental or economic costs. Overall, 18 out of 67 extant introduced vertebrate species (27%) have had either significant environmental or economic costs or both, and 9 out of 67 (13%) have had significant economic costs alone.

The tens rule suggests that 10% of introduced species that become established should become pests,[156,157] defined by Williamson as species that have a negative economic effect.[6] Williamson set a standard roughness, or margin of error, for the rule, which has been put at between 5% and 20%.[158] According to the economic definition of pests, the percentage of introduced vertebrates to Britain that have become pests following establishment falls within the upper limits predicted by the tens rule. However, with a wider definition of pests as species that cause negative economic or environmental effects (or that have negative total economic effects, that is, incorporating environmental values as well as financial ones), this proportion is well above that expected by the tens rule.

The proportion of alien vertebrates in Britain that have had significant negative environmental and economic impacts is high. This may reflect the low number of resident species, and hence the ability of an introduced species to build up large numbers. Nevertheless, the scale of the impacts of introduced species in Britain is very much less than that experienced by countries with more typical insular biotas, such as New Zealand.[122] Of all the alien vertebrates in Britain, only the rabbit and possibly the coypu have had a sufficient environmental impact to significantly affect ecosystem structure and function, and only the rabbit, house mouse, common rat, and grey squirrel impose major economic costs on a national scale. Moreover, the economic impact of the grey squirrel is mainly confined to central and southern England and Wales. This picture is very different from that for countries with a more typical "island" fauna. However, some introduced species that are currently increasing in numbers or range, such as Reeves' muntjac, the sika deer, the Canada goose, the marsh frog, and the zander, have the potential to impose much greater environmental and economic costs in the future if their populations continue to grow. Furthermore, natural increases could be exacerbated by habitat and landscape change, as well as by climate change.

7.5.6 Future habitat changes and the impact of introduced vertebrates

One of the key habitat changes in the short to medium term will be an increase in broad-leaved woodland cover, encouraged by agri-environment schemes and a move toward biomass energy sources. The regeneration and growth of such woodlands will provide improved and more abundant habitats for deer, with the likelihood of increased conflicts between both introduced and native deer and the environmental and economic interests associated with the woodlands. This is already the case for native red and introduced sika deer in Scotland. Future climate changes may be especially important for those species such as amphibians whose current range or breeding success is closely linked with prevailing environmental conditions.[8] However, it may also affect diversity and the balance between native and introduced species for groups less obviously affected by climate, such as birds.[159]

Although this chapter has been concerned with the environmental and economic costs of alien species, it must be remembered that native species may have impacts that are just as severe, especially from an economic perspective. For example, although muntjac, fallow, and sika deer can cause economic damage to forests and woodlands, native red and roe deer cause far greater damage. Similarly, native red foxes are much more significant predators from both an economic and environmental perspective than any introduced mammalian carnivore.

7.5.7 Environmental and economic benefits of introduced vertebrates

It should be borne in mind that introduced species can bring benefits as well as costs. The relative importance of the benefits and costs varies according to the time span over which they are being assessed, and this can lead to varying conclusions regarding the desirability of different introduced species.

For example, coypu, mink, and muskrats brought economic benefits for the fur industry when they were first brought into the country, prior to escape and establishment in the wild. Sika deer, pheasants, and red-legged partridges generate significant revenues from game shooting, and many alien fish, in particular species such as rainbow trout, carp, and zander, provide recreational and local economic benefits through angling. Recreational fishing is a very important activity in Britain, involving up to 2.2 million people spending a total of £3.15 billion each year,[160] and these alien species will therefore bring significant indirect benefits to local economies.

Besides being a major agricultural pest, rabbits have a significant positive effect in increasing floral diversity on chalk downland and breckland. The decline in rabbit grazing caused by myxomatosis has been suggested as one of the reasons for the extinction of the British population of the large blue butterfly (*Maculina arion*).[161] Rabbits also play an extremely important role in the food web, providing more than 75% of the total food energy available to mammalian predators.[162] The mandarin duck is a subject of conservation measures in its native range, and therefore alien populations in Britain may represent important refuges for this species.[1] One alien vertebrate species, the brown hare, is the subject of a U.K. Biodiversity Action Plan to expand its numbers and range.[163] Harris et al. even suggest that the British populations of feral goat, feral sheep, and Chinese water deer are of greater international conservation importance that any of Britain's native mammals.[162]

7.5.8 Future management of alien vertebrates in Britain

Alien species make up almost half the biomass of the current British mammal and bird faunas, although this proportion is considerably less for the other vertebrate groups. Eradicating all the alien species would not be practical and would have massive implications for populations of remaining native species. Some authors argue that long-established alien species in Britain that cause no significant environmental or economic costs should not represent a major cause for concern,[1] although others disagree.[131] For some alien species, the ecological status of the species as a whole or public perception may promote a desire to conserve them. Moreover, some alien species can actually make positive contributions to the environment or the economy.

Nevertheless, there are a number of alien species that do impose significant costs. The annual costs of economic damage due to alien vertebrates in Britain are in excess of £190 million, and the costs of control and mitigation are in excess of £49 million (Table 7.3). Almost all the damage costs are due to rabbits. Although rabbits are by far the most economically damaging alien vertebrate species in Britain, their relative importance is exacerbated by the lack of data on the costs imposed by other alien vertebrates. The majority of mitigation costs is also due to rabbits, but commensal rodents contribute almost a third of all these costs.

In addition to the introduced vertebrate species that are currently acting as significant pests, there are also a number of species that could potentially cause environmental or economic damage in the future, even though they do no damage in their current state of abundance or distribution. Given that the majority of established alien vertebrates in Britain are currently increasing in range and/or abundance, known populations of these species should be subject to stringent monitoring, and a precautionary approach taken to control them if there are signs that these populations are increasing.

Acknowledgments

We are very grateful to Dave Cowan, Roger Quy, Graham Smith, and Richard Watkins (Central Science Laboratory), Dave Goffin and Dick Youngson (Deer Commission for Scotland), Tom Langton (Froglife), and Colin Booty (Royal Society for the Protection of Animals) for providing unpublished data and reports. We are also grateful to Charles Critchley (Forest Enterprise) for informative discussions. We thank Nick Arnold (Natural History Museum), Jeremy Greenwood (British Trust for Ornithology), Tom Langton (Froglife), Gordon McKillop (Sports Turf Research Institute), Mark Williamson and Dave Raffaelli (University of York), and Alwyne Wheeler for useful comments on the manuscript.

References

1. Manchester, S.J. and Bullock, J.M., The impacts of non-native species on UK biodiversity and the effectiveness of control, *J. Appl. Ecol.*, 37, 845, 2000.
2. Pimentel, D., et al., Environmental and economic costs associated with non-indigenous species in the United States, http://www.news.cornell.edu/releases/Jan99/species_costs.htm, 1999.

3. Lever, C., *They Dined on Eland: The Story of the Acclimatisation Societies*, Quiller Press, London, 1992.
4. Williamson, M., *Biological Invasions*, Chapman & Hall, London, 1996.
5. Brown, J.H., Patterns, modes and extents of invasions by vertebrates, in *Biological Invasions: A Global Perspective*, Drake, J., et al., Eds., John Wiley & Sons, Chichester, 1989, 85.
6. Lever, C., *The Naturalized Animals of the British Isles*, Hutchinson, London, 1977.
7. Harradine, J., Sporting shooting in the United Kingdom — some facts and figures, Proceedings of the Second Meeting of the IUGB Working Group on Game Statistics, Netherlands, 1982.
8. Beebee, T.J.C. and Griffiths, R.A., *Amphibians and Reptiles: A Natural History of the British Herpetofauna*, Harper Collins, London, 2000.
9. Harris, S., et al., *A Review of British Mammals: Population Estimates and Conservation Status of British Mammals Other than Cetaceans*, Joint Nature Conservation Committee, Peterborough, 1995.
10. British Ornithologists' Union Records Committee, *The British List*, BOU, Tring, Hertfordshire, 2000.
11. Gibbons, D.W., Reid, J.B., and Chapman, R.A., *The New Atlas of Breeding Birds in Britain and Ireland: 1988-1991,* Poyser, London, 1993.
12. Holmes, J.S., et al., Ducks breeding in the United Kingdom in 1994, *Brit. Birds*, 91, 336, 1998.
13. Langton, T.E.S. and Beckett, C.L., Reptiles and amphibians in the United Kingdom: a review of their status and international significance, unpublished report to Joint Nature Conservation Committee, 1995.
14. Maitland, P.S. and Lyle, A.A., Conservation of freshwater fish in the British Isles: the current status and biology of threatened species, *Aquat. Conserv.*, 1, 25, 1991.
15. Farr-Cox, F., Leonard, S., and Wheeler, A., The status of the recently introduced fish *Leucaspius delineatus* (Cyprinidae) in Great Britain, *Fisheries Manag. Ecol.*, 3, 193, 1996.
16. Maitland, P.S. and Campbell, R.N., *Freshwater Fishes of the British Isles*, Harper Collins, London, 1992.
17. Putman, R.J. and Moore, N.P., Impact of deer in lowland Britain on agriculture, forestry and conservation habitats, *Mammal Rev.*, 28, 141, 1998.
18. Doney, J. and Packer, J., An assessment of the impact of deer on agriculture, in *Population Ecology, Management and Welfare of Deer,* Goldspink, C.R., King, S., and Putman, R.J., Eds., Manchester Metropolitan University, Manchester, 1998, 38.
19. Langbein, J., Quantock landholders deer management questionnaire — overview of main results, collated by Dr. J. Langbein on behalf of the Quantock Deer Management and Conservation Group, unpublished document, 1998.
20. Gill, R.M.A., A review of damage by mammals in north temperate forests. 1. Deer, *Forestry,* 65, 145, 1992.
21. Welch, D., et al., Leader browsing by red and roe deer on young sitka spruce trees in western Scotland. I. Damage rates and the incidence of habitat factors, *Forestry*, 64, 61, 1991.
22. Gill, R.M.A., A review of damage by mammals in north temperate forests. 3. Impact on trees and forests, *Forestry*, 65, 363, 1992.
23. Gill, R.M.A., Webber, J., and Peace, A., *The Economic Implications of Deer Damage: A Review of Current Evidence,* Forest Research Agency, Wrecclesham, Surrey, 2000.
24. Ward, A.I., The ecology and sustainable management of roe deer in multiple use forestry, Ph.D. thesis, University of York, 2001.
25. Fenner, F. and Ross, J., Myxomatosis, in *The European Rabbit: The History and Biology of a Successful Colonizer*, Thompson, H.V and King, C.M., Eds., Oxford University Press, Oxford, 1994, 205.
26. Ross, J. and Tittensor, A.M., Influence of myxomatosis in regulating rabbit numbers, *Mammal Rev.*, 16, 163, 1986.
27. Ross, J., et al., Myxomatosis in farmland rabbit populations in England and Wales, *Epidemiol. Infect.*, 103, 333, 1989.
28. Kolb, H.H., Rabbit *Oryctolagus cuniculus* populations in Scotland since the introduction of myxomatosis, *Mammal Rev.*, 24, 41, 1994.

29. Rees, W.A., et al., Humane control of rabbits, in *Humane Control of Land Mammals and Birds*, Britt, D.P., Ed., Universities Federation for Animal Welfare, Potters Bars, London, 1985, 96.
30. McKillop, I.G., et al., Developing a model to predict yield loss of winter wheat due to grazing by European wild rabbits, *Brighton Crop Protection Conference 1996: Pests and Diseases*, 1, 145, 1996.
31. Bell, A.C., Byrne, P.M., and Watson, S., The effect of rabbit (*Oryctolagus cuniculus*) grazing damage on the growth and yield of winter cereals, *Ann. Appl. Biol.*, 133, 431, 1998.
32. Kenward, R.E., Bark-stripping by grey squirrels in Britain and North America: why does the damage differ?, in *Mammals as Pests*, Putman, R. J., Ed., Chapman & Hall, London, 1989, 144.
33. Gill, R.M.A., A review of damage by mammals in north temperate forests. 2. Small mammals, *Forestry*, 65, 281, 1992.
34. Rowe, J., Grey squirrel bark-stripping damage to broadleaved trees in southern Britain up to 1983, *Q. J. Forest.*, 78, 231, 1984.
35. Rowe, J. and Gill, R., The susceptibility of tree species to damage by grey squirrels in England and Wales, *Q. J. Forest.*, 79, 183, 1985.
36. Pepper, H.W., Grey squirrel damage control with warfarin, Forestry Commission Research Information Note 180, HMSO, London, 1990.
37. Kenward, R.E., et al., Comparative demography of red squirrels (*Sciurus vulgaris*) and grey squirrels (*Sciurus carolinensis*) in deciduous and coniferous woodland, *J. Zool.*, 244, 7, 1998.
38. Prickett, A.J., English Farm Grain Stores 1987 Part 1. Storage practice and pest incidence, ADAS Central Science Laboratory Research Report Number 23, 1988.
39. Prickett, A.J. and Muggleton, J., Commercial grain stores 1988/89 England and Wales Part 2. Pest incidence and storage practice, HGCA Research Report Number 29, 1991.
40. Barnett, S.A., Damage to wheat by enclosed populations of *Rattus norvegicus*, *J. Hyg. Cambridge*, 49, 22, 1951.
41. Goldenberg, N. and Rand, C., Rodents and the food industry: an in-depth analysis for a large British food handler, *Pest Control*, 39, 24, 1971.
42. Cotton, K.E., The coypu, *River Board Association Yearbook*, 11, 31, 1963.
43. Jones-Walters, L.M. and Corbet, G.B., Fat dormouse *Glis glis*, in *The Handbook of British Mammals*, Corbet, G.B. and Harris, S., Eds., Blackwell Scientific Publications, Oxford, 1991, 264.
44. Clarke, S.P., Feral mink in southwest England, *Mammal Rev.*, 1, 92, 1970.
45. Chanin, P.R.F., The ecology of feral mink (*Mustela vison* Schreber) in Devon, Ph.D. thesis, University of Exeter, 1976.
46. Chanin, P.R.F., The diet of the otter *Lutra lutra* and its relations with the feral mink *Mustela vison* in two areas of southwest England UK, *Acta Theriol.*, 26, 83, 1981.
47. Dunstone, N. and Birks, J.D.S., The feeding ecology of mink (*Mustela vison*) in coastal habitat, *J. Zool.*, 212, 69, 1987.
48. Tapper, S., *Game Heritage — An Ecological Review from Shooting and Gamekeeping Records*, Game Conservancy Limited, Fordingbridge, Hampshire, 1992.
49. Harrison, M.D.K. and Symes, R.G., Economic damage by feral American mink (*Mustela vison*) in England and Wales, in *Mammal as Pests*, Putman, R.J., Ed., Chapman & Hall, London, 1989, 242.
50. Cuthbert, J., Food studies of feral mink *Mustela vison* in Scotland, *Fish Manage.*, 10, 17, 1979.
51. Chanin, P.R.F. and Linn, I., The diet of feral mink (*Mustela vison*) in southwest Britain, *J. Zool.*, 192, 205, 1980.
52. Birks, J.D.S., *Mink*, The Mammal Society, London, 1986.
53. Birks, J.D.S. and Dunstone, N., Mink *Mustela vison*, in *The Handbook of British Mammals*, Corbet, G.B. and Harris, S., Eds., Blackwell Scientific Publications, Oxford, 1991, 406.
54. Simpson, W., Agricultural damage and its prevention, in *Canada Geese Problems and Management Needs*, Harradine, J., Ed., Rossett, 1991, 21.
55. Kirby, J.S., et al., Index numbers for waterbird populations. III. Long-term trends in the abundance of wintering wildfowl in Great Britain, 1966/67–1991/92, *J. Appl. Ecol.*, 32, 536, 1995.

56. Allan, J.R., Kirby, J.S., and Feare, C.J., The biology of Canada geese *Branta canadensis* in relation to the management of feral populations, *Wildlife Biol.*, 1, 129, 1995.
57. Greenwood, J., personal communication, 2001.
58. Maitland, P.S. and Lyle, A.A., Conservation of freshwater fish in the British Isles: proposals for management, *Aquat. Conserv.*, 2, 165, 1992.
59. Gratz, N.G., Rodents as carriers of disease, in *Rodent Pests and Their Control*, Buckle, A.P. and Smith, R.H., Eds., CAB International, Oxford, 1994, 85.
60. Hart, C.A., Trees, A.J., and Duerden, B.I., Zoonoses, *J. Med. Microbiol.*, 46, 4, 1997.
61. Davies, R.H. and Wray, C., Mice as carriers of *Salmonella enteritis* on persistently infected poultry units, *Vet. Rec.*, 137, 337, 1995.
62. Pocock, M.J.O., et al., Patterns of infection by *Salmonella* and *Yersinia* spp. in commensal house mouse (*Mus musculus domesticus*) populations, *J. Appl. Microbiol.*, 90, 755, 2001.
63. Healing, T.D. and Greenwood, M.H., Frequency of isolation of *Campylobacter* spp., *Yersinia* spp. and *Salmonella* spp. from small mammals from two sites in southern Britain, *Int. J. Environ. Heal. R.*, 1, 54, 1991.
64. Henzler, D.J. and Opitz, H.M., The role of mice in the epizootiology of *Salmonella enteritis* infection on chicken layer farms, *Avian Dis.*, 36, 625, 1992.
65. Quy, R.J., et al., The Norway rat as a reservoir host of *Cryptosporidium parvum*, *J. Wildlife Dis.*, 35, 660, 1999.
66. Krebs, J., *Bovine Tuberculosis in Cattle and Badgers. Report by the Independent Scientific Review Group*, MAFF Publications, London, 1997.
67. Donaldson, A., et al., What is the risk of feral wild boar becoming infected with FMD and subsequently causing new incidents of FMD in livestock?, Foot and Mouth Disease Veterinary Risk Assessment no. 7, Department for Environment, Food and Rural Affairs, http://www.defra.gov.uk/animalh/diseases/fmd/disease/risks/risk7a.htm, 2001.
68. Gurnell, J., Grey squirrel *Sciurus carolinensis*, in *The Handbook of British Mammals*, Corbet, G.B. and Harris, S., Eds., Blackwell Scientific Publications, Oxford, 1991, 186.
69. Ward, L.K., Clarke, R.T. and Cooke, A.S., Long term scrub succession deflected by fallow deer at Castor Hanglands National Nature Reserve, National Environment Research Council, Huntingdon, Cambridgeshire, 1994.
70. Mitchell, F.J.G. and Kirby, K.J., The impact of large herbivores on the conservation of semi-natural woodland in the British uplands, *Forestry*, 63, 333, 1990.
71. Watson, A., Eighteenth century deer numbers and pine regeneration near Braemar, Scotland, *Biol. Conserv.*, 25, 289, 1983.
72. Scottish Natural Heritage, Red deer and the natural heritage, Scottish Natural Heritage Policy Paper, Scottish Natural Heritage Publications & Graphics, Perth, 1994.
73. Ramsay, P., *Revival of the Land — Creag Meagaigh National Nature Reserve*, Scottish Natural Heritage, Perth, 1997.
74. Kay, S., Factors affecting severity of deer browsing damage within coppiced woodland in the south of England, *Biol. Conserv.*, 63, 217, 1993.
75. Putman, R., Deer management on national nature reserves: problems and practices, *English Nature Research Reports*, 173, 1-50, 1996.
76. Langbein, J., The ranging behaviour, habitat use and impact of deer in oak woods and heather moors of Exmoor and the Quantock Hills, Report, British Deer Society, Fordingbridge, Hampshire, 1997.
77. Key, G., Moore, N.P., and Hart, J., Impact and management of deer in farm woodlands, in *Population Ecology, Management and Welfare of Deer*, Goldspink, C.R., King, S. and Putman, R.J., Eds., Manchester Metropolitan University, Manchester, 1997, 44.
78. Cooke, A.S., Is the muntjac a pest in Monks Wood National Nature Reserve?, *Deer*, 9, 243, 1994.
79. Cooke, A.S., Colonisation by muntjac deer *Muntiacus reevesi* and their impact on vegetation, in *Monks Wood National Nature Reserve, the Experience of 40 Years 1953-1993*, Massey, M.S. and Welch, R.C., Eds., English Nature, Peterborough, 1994, 45.
80. Watt, A.S., On the causes of failure of natural regeneration in British oakwoods, *J. Ecol.*, 7, 173, 1919.

81. Thomas, A.S., Changes in vegetation since the advent of myxomatosis, *J. Ecol.*, 48, 287, 1960.
82. Thomas, A.S., Further changes in vegetation since the advent of myxomatosis, *J. Ecol.*, 51, 151, 1963.
83. Watt, A.S., A comparison of grazed and ungrazed grassland A in East Anglian Breckland, *J. Ecol.*, 69, 499, 1981.
84. Cullen, W.R., Wheater, C.P., and Dunleavy, P.J., Establishment of species-rich vegetation on reclaimed limestone quarry faces in Derbyshire, UK, *Biol. Conserv.*, 84, 25, 1998.
85. Boorman, L.A. and Fuller, R.M., The changing status of reedswamp in the Norfolk Broads, *J. Appl. Ecol.*, 18, 241, 1981.
86. Ellis, E.A., Some effects of selective feeding by the coypu (*Myocastor coypus*) on the vegetation of Broadland, *Transactions of the Norfolk and Norwich Naturalists' Society*, 20, 32, 1963.
87. Moss, B., The Norfolk broadland: experiments in the restoration of a complex wetland, B*iol. Rev.*, 58, 521, 1983.
88. Tapper, S.C., Brown hare *Lepus europaeus, The Handbook of British Mammals*, Corbet, G.B. and Harris, S., Eds., Blackwell Scientific Publications, Oxford, 1991, 154.
89. Bullock, D.J., Feral goat *Capra hircus*, in *The Handbook of British Mammals*, Corbet, G.B. and Harris, S., Eds., Blackwell Scientific Publications, Oxford, 1991, 541.
90. Jefferies, M.J., Morris, P.A. and Mulleneux, J.E., An enquiry into the changing status of the water vole *Arvicola terrestris* in Britain, *Mammal Rev.*, 19, 111, 1989.
91. Woodroffe, G.L., Lawton, J.H. and Davidson, W.L., The impact of mink on water vole populations in the North York Moors National Park, *Biol. Conserv.*, 51, 49, 1990.
92. Strachan, C. and Jefferies, D.J., The Water Vole *Arvicola terrestris* in Britain 1989-1990: Its Distribution and Changing Status, The Vincent Wildlife Trust, London, 1993.
93. Halliwell, E.C. and Macdonald, D.W., American mink *Mustela vison* in the Upper Thames catchment: relationship with selected prey species and den availability, *Biol. Conserv.*, 76, 51, 1996.
94. Barreto, G.R., et al., The role of habitat and mink predation in determining the status and distribution of water voles in England, *Anim. Conserv.*, 1, 129, 1998.
95. Birks, J.D.S., Feral mink and nature conservation, *British Wildlife*, 1, 313, 1990.
96. Dunstone, N., *The Mink*, Poyser, London, 1993.
97. Ferreras, P. and Macdonald, D.W., The impact of American mink *Mustela vison* on water birds in the Upper Thames, *J. Appl. Ecol.*, 36, 701, 1999.
98. Craik, J.C.A., Effects of North American mink on the breeding success of terns and smaller gulls in west Scotland, *Seabird*, 17, 3, 1995.
99. Craik, J.C.A., Long-term effects of North American mink on seabirds in western Scotland, *Bird Study*, 44, 303, 1997.
100. Rae, S., The effects of predation on ground-nesting birds in the Outer Hebrides, Report to the Mink Eradication Scheme for the Hebrides, 1999.
101. Burton, N.H.K. and Fuller, R.J., A review of the status and population trends of ground-nesting birds vulnerable to mink predation on Harris and Lewis. British Trust for Ornithology Report No. 230, British Trust for Ornithology, Thetford, Norfolk, 1999.
102. Churcher, C.S. and Lawton, J.H., Predation by domestic cats in an English village, *J. Zool.*, 212, 439, 1987.
103. Wattaola, G., Allan, J.R. and Feare, C.J., Problems and management of naturalised introduced Canada geese *Branta canadensis* in Britain, in *The Introduction and Naturalisation of Birds*, Holmes, J.S. and Simons, J.R., Eds., HMSO, London, 1996, 71.
104. Beebee, T.J.C., Habitats of the British amphibians (3): river valley marshes, *Biol. Conserv.*, 18, 281, 1981.
105. Langton, T., personal communication, 2001.
106. Langton, T. and Burton, J.A., *Amphibians and Reptiles. Conservation Management of Species and Habitats,* Council of Europe Publishing, Strasbourg, 1997.
107. Wauters, L.A. and Gurnell, J., The mechanism of replacement of red squirrels by grey squirrels: a test of the interference competition hypothesis, *Ethology*, 105, 1053, 1999.
108. Kenward, R.E. and Holm, J.L., On the replacement of the red squirrel in Britain: a phytotoxic explanation, *P. Roy. Soc. Lond. B Bio.*, 251, 187, 1993.

109. Okubo, A., et al., On the spatial spread of the grey squirrel in Britain, *P. Roy. Soc. Lond. B Bio.*, 238, 113, 1989.
110. Wauters, L.A., Lurz, P.W.W. and Gurnell, J., Interspecific effects of grey squirrels (*Sciurus carolinensis*) on the space use and population demography of red squirrels (*Sciurus vulgaris*) in conifer plantations, *Ecol. Res.*, 15, 271, 2000.
111. Shuttleworh, C., No more grey days for Anglesey red squirrels, *Mammal News*, 128, 11, 2001.
112. Bibby, C., More than enough exotics?, *British Birds*, 95, 2, 2000.
113. Davison, A., et al., Hybridization and the phylogenetic relationship between polecats and domestic ferrets in Britain, *Biol. Conserv.*, 87, 155, 1998.
114. Blandford, P.R.S., Biology of the polecat *Mustela putorius*: a literature review, *Mammal Rev.*, 17, 155, 1987.
115. French, D.D., Corbett, L.K., and Easterbee, N., Morphological discriminants of Scottish wildcats (*Felis silvestris*), domestic cats (*F. catus*) and their hybrids, *J. Zool.*, 214, 235, 1988.
116. Daniels, M.J. et al., Morphological and pelage characteristics of wild living cats in Scotland: implications for defining the 'wildcat', *J. Zool.*, 244, 231, 1998.
117. Daniels, M.J. et al., Ecology and genetics of wild-living cats in the north-east of Scotland and the implications for the conservation of the wildcat, *J. Appl. Ecol.*, 38, 146, 2001.
118. Harrington, R., The hybridization of red deer (*Cervus elaphus* L. 1758) and Japanese sika deer (*Cervus nippon* Temminck 1838), *Transactions of the International Congress of Game Biologists*, 14, 559, 1982.
119. Putman, R.J. and Hunt, E.J., Hybridisation between red and sika deer in Britain, *Deer*, 9, 104, 1993.
120. Balharry, E., et al., Hybridisation in British mammals, JNCC report no. 154, Joint Nature Conservation Committee, Peterborough, 1994.
121. Robertson, P.A. and Rosenberg, A.A., Harvesting gamebirds, in *Ecology and Management of Gamebirds*, Hudson, P.J. and Rands, M.R.W., Eds., BSP Professional Books, Oxford, 1988, 177.
122. Lever, C., *Naturalized Animals: the Ecology of Successfully Introduced Species*, Poyser, London, 1994.
123. Hughes, B., The ruddy duck *Oxyura jamaicensis* in the Western Palearctic and the threat to the white-headed duck *Oxyura leucocephala*, in *The Introduction and Naturalisation of Birds*, Holmes, J.S. and Simons, J.R., Eds., HMSO, London, 1996, 79.
124. Snell, C., The pool frog: a neglected native?, *British Wildlife*, 6, 1, 1994.
125. Wheeler, A., Status of the Crucian carp, *Carassius carassius* (L.), in the UK, *Fisheries Manag. Ecol.*, 7, 315, 2000.
126. Beveridge, M.C.M., Ross, L.G., and Kelly, L.A., Aquaculture and biodiversity. *Ambio*, 23, 497, 1994.
127. Bratton, S.P., The effect of the European wild boar (*Sus scrofa*) on Gray Beech Forest in the Great Smoky Mountains. *Ecology*, 56, 1356, 1975.
128. Welander, J., Are wild boar a future threat to the Swedish flora? *Ibex*, 3, 165, 1995.
129. Ministry of Agriculture, Fisheries and Food, Current status and potential impact of wild boar (*Sus scrofa*) in the English countryside: a risk assessment, Central Science Laboratory, York, 1998.
130. Gosling, L.M. and Baker, S.J., The eradication of muskrats and coypus from Britain, *Biol. J. Linn. Soc.*, 38, 39, 1989.
131. Yalden, D., *The History of British Mammals*, Poyser, London, 1999.
132. Atkinson, T.C., Briffa, K.R., and Coope, G.R., Seasonal temperatures in Britain during the past 22,000 years, reconstructed using beetle remains. *Nature*, 325, 587, 1987.
133. Bilton, D.T., et al., The Mediterranean region of Europe as an area of endemism for small mammals rather than the major source for the postglacial colonisation of northern Europe, *P. Roy. Soc. Lond. B Bio.*, 265, 1219, 1998.
134. Corbet, G.B. and Harris, S., Eds., *The Handbook of British Mammals*, 3rd edition, Blackwell Scientific Publications, Oxford, 1991.
135. Mitchell-Jones, A.J., et al., *The Atlas of European Mammals*, Poyser, London, 1999.
136. Greenwood, J.J.D., et al., Relations between abundance, body size and species number in British birds and mammals, *Philos. T. Roy. Soc. B*, 351, 265, 1996.

137. Maroo, S. and Yalden, D.W., The Mesolithic mammal fauna of Great Britain, *Mammal Rev.*, 30, 243, 2000.
138. Goffin, D., personal communication, 2001.
139. Toleman, R.D.L., Deer cull returns 1994-95, Unpublished report, Forestry Commission, 1995.
140. Youngson, R.W., Red deer control in Fiunnary Forest, Argyll: control effort, Unpublished summary of a joint project report between Forestry Commission and Red Deer Commission, 2000.
141. Youngson, R.W., Costs of sika control in native woodland, Unpublished summary of a joint report between Forestry Commission and Red Deer Commission, 2000.
142. Mayle, B., Managing deer in the countryside, Forestry Commission Practice Note no. 6, Forestry Commission, Edinburgh, 1999.
143. Staines, B.W., Humane control of deer in rural areas, in *Humane Control of Land Mammals and Birds*, Britt, D.P., Ed., Universities Federation for Animal Welfare, Potters Bar, Hertfordshire, 1985, 105.
144. Ratcliffe, P.R., The control of red and sika deer populations in commercial forests, in *Mammals as Pests*, Putman, R.J., Ed., Chapman & Hall, London, 1989, 98.
145. Baines, D. and Summers, R.W., Assessments of bird collisions with deer fences in Scottish forests. *J. Appl. Ecol.*, 34, 941, 1997.
146. Trout, R., personal communication, 2001.
147. Smith, G.C., Designing an expert system for rabbit management in the UK – where are we?, *Proceedings of the Second European Vertebrate Pest Management Conference*, in press, 2002.
148. De'Ath, A., Garthwaite, D.G., and Thomas, M.R., Rodenticide usage on farms in Great Britain growing arable crops 1996, PUSG Report 144, MAFF Publications, London, 1999.
149. Bankes, J. and Garthwaite, D.G., Rodenticide usage by local authorities in Great Britain 1997, PUSG Report 155, MAFF Publications, London, 2000.
150. Quy, R., personal communication, 2001.
151. Remfry, J., Feral cats in the United Kingdom, *J. Am. Vet. Med. Assoc.*, 208, 520, 1996.
152. Booty, C., personal communication, 2001.
153. Cats Protection, http://www.cats.org.uk, May 2001.
154. Watkins, R., personal communication, 2001.
155. Chapman, N., Harris, S., and Stanford, A., Reeves' muntjac *Muntiacus reevesi* in Britain: their history, spread, habitat selection, and the role of human intervention in accelerating their dispersal, *Mammal Rev.*, 24, 113, 1994.
156. Holdgate, M.W., Summary and conclusions: characteristics and consequences of biological invasions, *Philos. T. Roy. Soc.* B, 314, 733, 1986.
157. Williamson, M. and Brown, K.C., The analysis and modelling of British invasions. *Philos. T. Roy. Soc.* B, 314, 505, 1986.
158. Williamson, M., Environmental risks from the release of genetically modified organisms (GMOs) – the need for a molecular ecology, *Mol. Ecol.*, 1, 3, 1992.
159. Lennon, J.J., Greenwood, J.J.D. and Turner, J.R.G., Bird diversity and environmental gradients in Britain: a test of the species-energy hypothesis, *J. Anim. Ecol.*, 69, 581, 2000.
160. Hughes, S. and Morley, S., Aspects of fisheries and water resources management in England and Wales, *Fisheries Manag. Ecol.*, 7, 75, 2000.
161. Sumption, K.J. and Flowerdew, J.R., The ecological effects of the decline of rabbits (*Oryctolagus cuniculus* L.) due to myxomatosis. *Mammal Rev.*, 15, 151, 1985.
162. Harris, S., et al., Abundance/mass relationships as a quantified basis for establishing mammal conservation priorities, in *Priorities for the Conservation of Mammalian Biodiversity: has the Panda had its Day*?, Entwistle, A. and Dunstone, N., Eds., Cambridge University Press, Cambridge, 2001, 101.
163. Anon., *Biodiversity: the UK Steering Group Report, Volume 2: Action Plans*. HMSO, London, 1995.
164. Baker, S.J. and Hills, D., Escapes and introductions, in *The New Handbook of British Mammals*, Harris, S., Ed., in press, 2002.
165. Hutchings, M.R. and Harris, S., *The Current Status of the Brown Hare* (Lepus europaeus) *in Britain*, Joint Nature Conservation Committee, Peterborough, 1996.

166. Ogilvie, M. and the Rare Breeding Birds Panel, Non-native birds breeding in the United Kingdom in 1996, *Brit. Birds*, 2, 176, 1999.
167. Delany, S., Introduced and escaped geese in Britain in summer 1991, *Brit. Birds*, 86, 591, 1993.
168. Tapper, S., Ed., *A Question of Balance – Game Animals and their Role in the British Countryside*, The Game Conservancy Trust, Fordingbridge, Hampshire, 1999.
169. Pithon, J.A. and Dytham, C., Census of the British ring-necked parakeet *Psittacula krameri* population by simultaneous counts of roosts, *Bird Study*, 46, 112, 1999.
170. Anon., SNH earmarks £443,000 for Western Isles mink eradication. *Scotland's Natural Heritage*, 19, 21, 2001.
171. Wheeler, A., A list of the common and scientific names of fishes of the British Isles, *J. Fish Biol.*, 41A, 1, 1992.
172. Rushton, S.P., et al., Modelling the spatial dynamics of parapoxvirus disease in red and grey squirrels: a possible cause of the decline in the red squirrel in the UK? *J. Appl. Ecol.*, 37, 997, 2000.

chapter eight

Non-native invasive species of arthropods and plant pathogens in the British Isles

David Pimentel

Contents

8.1 Introduction

Since humans first invaded the British Isles about 10,000 years ago,[1] approximately 30,000 non-indigenous species are estimated to have been introduced into the British Isles. Introduced species, such as wheat, corn, potatoes, and other food crops, and sheep, poultry, and other livestock now provide more than 99% of Britain's food supply, representing a value of more than $20 billion dollars per year.[2] Other exotic species have been used effectively for biological pest control, sport, pets, and food processing. Some alien species, however, have caused major economic losses in agriculture, forestry, and several other segments of the British economy and have harmed the environment. One recent report indicates that 26,000 plant species have been introduced into Britain, which has a native flora of only 1600.[3]

Most of the plant and vertebrate animal introductions into Britain have been intentional, whereas most arthropod animal and microbe introductions have been accidental. However, an estimated 5000 to 10,000 invertebrate specimens, representing 150 different species, are imported each year from the Philippines, Costa Rica, Madagascar, and El Salvador into the British Isles.[4] During the past 40 years, the rate of invasions of biotic invaders has increased enormously because of human population growth, rapid modes of travel, and alteration of the environment. In addition, more goods and materials are being traded among nations.[5,6]

Developing an estimate of the full extent of the environmental and economic damages caused by exotic arthropod and plant pathogen species and the number of species extinc-

0-8493-0836-4/02/$0.00+$1.50

tions they have caused is difficult, because nearly half of the species in the British Isles have not been described. Nonetheless, assuming conditions similar to those in the United States, more than 40% of the species that are listed as threatened or endangered are probably at risk primarily because of competition and predation by non-native species.[7] Many other species not listed are also negatively affected by alien species and/or ecosystem changes caused by alien species. Estimating the economic impacts of non-native arthropod and plant pathogen species in the British Isles is also difficult; nevertheless, enough data are available to quantify some of the impacts on agriculture and forestry. This chapter assesses the magnitude of the environmental impacts and economic costs associated with the diverse non-native arthropod and plant pathogen species that have invaded the British Isles.

8.2 Insects and mites

Approximately 1500 insect and mite species are pests and cause economic damage in the British Isles; approximately 30% of these pests are alien species.[8–10] Each year, insects and mites damage or destroy approximately $3.2 billion in crops in the British Isles, based on 10% crop losses attributed to insect and mite pests.[11] Alien arthropod species cause 30% of these crop losses, a share worth $960 million per year.

Several species of non-native insects have received attention recently. One of these species is the Mediterranean climbing cutworm (*Spodoptera littoralis*) that infests greenhouses in the United Kingdom.[12] The pest was probably introduced on chrysanthemum cuttings. The insect is widely distributed throughout the subtropical and tropical areas of Africa, Europe, and the Near East.[12] The animal can grow to be up to 5 cm long and thus is capable of consuming large amounts of vegetation.

Another major non-indigenous pest insect is the western flower thrips (*Frankliniella occidentalis*), which is polyphagous and feeds on more than 244 species of plants.[13] The pest is an exceptionally damaging insect for greenhouse crops and has proven to be very difficult to control, because it has developed resistance to a great many insecticides. The insect is expanding its geographical distribution rapidly.[13]

The tobacco whitefly (*Bemisia tabaci*) is endemic to South and Central America and has gained entrance to the British Isles.[13] This pest is another polyphagous insect that parasitizes about 500 species of crop plants, with poinsettia being the most commonly infested economic species. There are some Hymenoptera species that are parasitic on this whitefly pest that can provide control under some circumstances.

Pyrethroid insecticides are being applied against two alien leaf miners (*Liriomyza trifolii* and *L. huidobrensis*) that have become well established in the British Isles.[14] However, both species are demonstrating some resistance to the pyrethroid insecticides, causing major concern.

In addition to increasing insecticide resistance in insect pests, global warming is expected to have an impact on crops produced in the British Isles and Europe.[15,16] A study of the risks posed to potatoes in Britain by the introduced Colorado beetle (*Leptinotarsa decemlineata*) predicted that the beetle will increase its range by 102% and that damage can be expected to increase 76%.[16] Other introduced insect pests will probably increase their range, and the damage they do to crops will probably grow significantly.

Four alien species of cynipid gall-forming wasps (*Andricus corruptrix, A. lignicola, A. Kollari,* and *A. quercuscalicis*) have invaded the British Isles and are attacking both native and introduced oaks.[17] These insects do not appear to be causing major damage to the oaks in Britain. All four invading cynipid wasp species are being attacked by native parasitoid species.[17] The parasitoid species are providing some control of the pest cynipids.

Of the 29 major insect pest species attacking forest trees in Great Britain, 18 species, or 62%, are non-native insect species.[18] Efforts are under way to diversify the control techniques for all pests of forests in Great Britain, and some progress has been made in reducing the damage and forest losses to insect pests; nevertheless, Britain still imports more than 80% of its forest products.[18] Assuming a 10% loss in forest products in Britain, and that 62% of the loss is due to non-native species, the estimated losses of forest products due to invaders is $2 million per year.

8.3 Plant pathogens

In the British Isles, an estimated 74% of the plant pathogens on crops are introduced species.[19] Most of these alien plant pathogens were brought to the British Isles with seeds and other crop parts needed in agriculture. The economic loss due to plant pathogens amounts to 8.3% of potential production, or about $2.7 billion per year.[11] If 74% of the losses are due to alien plant pathogens, then about $2 billion per year in damages are therefore associated with alien plant pathogens attacking crops.[20]

One of the many plant diseases introduced into the British Isles is fireblight disease (*Erwinia amylovora*).[13] This infectious agent is a bacterium that is native to North America. Fireblight is restricted to the plants of the following genera: *Chaenorneles, Cotoneaster, Cretaegus, Cydonia, Malus, Pyracantha, Pyrus, Sorbus,* and *Stranvaesia.*[13]

Other invading plant pathogens include the following: *Leveillula taurica, cubensis, Colletotrichum acutatum*, and *Cladosporiu allii-cepae*.[13]

One of the most notable diseases that was introduced into the British Isles was late blight of potato (*Phytophtora infestans*). During 1845, late blight moved across Europe into Ireland, causing a 25% yield loss in potatoes.[21] The blight was more severe the next year (1846), when 80% of the potato crop was destroyed. During this period about 25% of the Irish population of 8 million died of starvation, and about 1 million emigrated to the United States.[21]

Several reports suggest that climate change, including global warming, will probably increase both insect pest and plant disease attacks on crops in the British Isles.[22] These attacks will involve both non-indigenous and native pests. The same trend is predicted for most crops worldwide.[23,24]

8.4 Conclusion

Although an estimated 26,000 species of plants have been introduced into the British Isles, plus large numbers of animals and microbes, relatively little is known about the environmental and economic impacts of all these non-indigenous species. The non-beneficial plants, animals, and microbes are having negative impacts on the native biota of the British Isles, but little information is available concerning this. Several serious insect pests present major problems to crops and forest production, but only rough guesses can be made as to the economic seriousness of the problems. Nevertheless, I would estimate that invading insect pests and plant pathogen pests cause $5 billion per year worth of damage to crops and forests in the British Isles.

References

1. Bennett, K.D., Boreham, S., Sharp, M.J. and Switsur, V.R. Holocene history of environment, vegetation and human settlement on Catta Ness, Lunnasting, Shetland. *J. Ecol.*, 80, 1992, 241.
2. Agra Europe. *UK Farm Income Highest for 20 Years*, Agra Europe Ed., No. 1681, N/1-N/2, 1996.
3. Crawley, M.J., Harvey, P.H. and Purvis, A. Comparative ecology of the native and alien floras of the British Isles, *Philosophical Transactions of the Royal Society of London B Biological Sciences*, 351, 1996, 1251.
4. Williams, D.L. Imported invertebrate: considering the veterinary aspects, *Veterinary Practice*, 22, 1990, 1.
5. Bryan, R.T. Alien species and emerging infectious diseases: past lessons and future applications, in *Proceedings of the Norway/UN Conference on Alien Species, July1–5*, Sundland, G.T., Schel, P.J. and Viken, A. Eds., Norwegian Institute for Nature Research, Trondheim, Norway, 1996.
6. USBC. *Statistical Abstract of the United States*, U.S. Bureau of the Census, U.S. Government Printing Office, Washington, D.C., 1999.
7. Wilcove, D.S. and Bean, M.J. *The Big Kill: Declining Biodiversity in America's Lakes and Rivers,* Environmental Defense Fund, Washington, D.C., 1994.
8. Gratwick, M. (Ed.). *Potato Pests, Reference Book 187*, Ministry of Agriculture, Fisheries and Food, Her Majesty's Stationery Office, London, 1989.
9. Seymour, P.R. *Invertebrates of Economic Importance in Britain: Common and Scientific Names*, Her Majesty's Stationery Office, London, 1989.
10. van Lenteren, J.C. Frequency and consequences of insect invasions, in *Plant and Microbial Biotechnology Research Series,* Hokkanen, H.M.T. and Lynch, M. Eds., Cambridge University Press, Cambridge, 1995, 30.
11. Oerke, E.C., Dehne, H.W., Schonbeck, F. and Weber, A. *Crop Production and Crop Protection: Estimated Losses in Major Food and Cash Crops*, Elsevier, Amsterdam, 1994.
12. Bartlett, P.W. and MacDonald, O. *Spodoptera littoralis*: Experiences with an alien polyphagous glasshouse pest in the United Kingdom, in *British Crop Protection Monograph: Plant Health and the European Single Market*, Edbbels D. Ed., British Crop Protection Council, London, 371.
13. Hume, H.D. and Conway, T. Recent introduction of alien pests and disease into Ireland. In *Plant Health 1992*, Kavanagh, J.A. and Brennan, P. Eds., Royal Irish Academy, 1993, 67.
14. MacDonald, O.C., Responses of the alien leaf miners *Liriomyza trifolii* and *Liriomiza huidobrensis* (Diptera: Agromyzidae) to some pesticides scheduled for the control in the U.K., *Crop Protection*, 10, 1991, 509.
15. Froud-Williams, R.J., Harrington, R., Hocking, T.J., Smith, H.G., and Thomas, T.H. Eds. *Implications of "Global Environmental Change" for Crops in Europe*, Aspects of Applied Biology, No. 45, Aspects of Applied Biology, Cambridge, 1996.
16. Baker, R.H.A., Cannon, R.J.C., Walters, K.F.A. An assessment of the risks posed by selected non-indigenous pests to UK crops under climate change, in *Implications of Global Environmental Change for Crops in Europe,* Froud-Williams, R.J., Harrington, R., Hocking, T.J. Smith, H.G. and Thomas, T.H. Eds., Aspects of Applied Biology, No. 45, Cambridge, 1996.
17. Schonrogge, K. and Crawley, M.J. Quantitative webs as a means of assessing the impact of alien insects, *J Animal Ecol.*, 69, 2000, 841.
18. Evans, H.F. the present position of forest entomology in Great Britain, *Forestry,* 70, 1997, 327.
19. Carlile, W.R. *Control of Crop Diseases*, Edward Arnold, London, 1988.
20. Pimentel, D. McNair, S., Janecka, J, Wightman, J., Simmons, C., O'Connell, C., Wong, E., Russel, L., Zern, J., Aquino, T., and Tsomondo, T. Economic and environmental threat of alien plant, animal and microbe invasions, *Agriculture, Ecosystems and the Environment,* 84, 2001, 1.
21. Parry, D.W. *Plant Pathology in Agriculture.* Cambridge University Press, Cambridge, 1990.

22. Hardwick, N.V., Armstrong, A.C., Ellis, S.A., Gladders, P., Yarham, Parker, W.E., Cannon, R.J.C., Baker, R.H.A., Polleyi, R.W. and Turner, J.A. The impact of climate change on crop pests and diseases, in *Implications of Global Environmental Change for Crops in Europe*, Froud-Williams, R.J., Harrington, R., Hocking, T.J., Smith, H.G. and Thomas.T.H. Eds., Aspects of Applied Biology, No. 45, American Association of Applied Biologists, Cambridge, 1996, 261.
23. Pimentel, D., Brown, N., Vecchio, F., La Capra, V., Hausman, S., Lee, O., Diaz, A., Williams, J., Cooper, S., and Newburger, E. Ethical issues concerning potential global climate change on food production, *J Agri Environ Ethics,* 5, 1992, 113.
24. Rosezwig, C. and Hillel, D. *Climate Change and the Global Harvest: Potential Impacts of the Greenhouse Effect on Agriculture*, Oxford University Press, New York, 1998.

section five

India

chapter nine

Alien plant pathogens in India

Rama S. Singh and Jaspal Kaur

Contents

9.1 Introduction

India is primarily an agricultural country. More than 70% of the population live in villages and earn a living from agriculture or an agriculture-based profession. The history of plant diseases is as old as agriculture itself. Plant diseases have great impact on the lives of human beings and animals. They modify the habits of human beings, affect the standard of living for individuals and their communities, and sometimes affect

0-8493-0836-4/02/$0.00+$1.50

the economy of an entire country. Certain famines in India were the result of severe outbreaks of plant diseases in crops due to the introduction of pathogens from outside of the country. The famous example, the Bengal Famine of 1943, was the result of an epiphytotic outbreak of *Helminthosporium* blight in paddy; about 2 million people died of starvation. According to Padmanabhan,[1] the Bengal Famine was the most devastating outbreak of paddy brown spot that has been recorded in plant pathology literature. Most of the time, the plant pathogens are introduced in India through seeds and plant propagation materials.

Keeping in view the importance of the spread of dangerous plant diseases, some regulatory measures were taken up in order to keep out alien plant pathogens. Such procedures are observed in many countries, through plant quarantine laws that provide a legal restriction on the entry of imported plant products, soil, cultures of living organisms, packing materials, and related commodities. In India, the Destructive Insect and Pest (DIP) Act was passed in 1914 and subsequently supplemented by other provisions. The Act was aimed at prevention and rapid eradication of infestations of plant diseases in India from foreign sources. The Act was revised eight times between 1930 to 1956, and further corrected several times later on. However, it still does not meet current needs under the prevailing circumstances. Table 9.1 shows examples of plant diseases that were introduced in India both before and after the enforcement of quarantine laws.[2]

The Directorate of Plant Protection Quarantine and Storage was established in 1946 as a central organization under the Union Ministry of Food and Agriculture and opened its headquarters in New Delhi. On July 9, 1951, India signed the International Plant Protection Convention sponsored by the FAO, recognizing the usefulness of international cooperation in preventing the introduction and spread of pests and diseases across national boundaries. Also signed in July 1956 was the Plant Protection Agreement for South East Asia and Pacific Region, also sponsored by the FAO. The first plant quarantine and fumigation station was established at Bombay in 1949. Today there are 27 quarantine and fumigation stations throughout India: 11 at airports, 9 at seaports, and 7 at land frontiers. Several plant pathologists are appointed for inspection, disinfection, and the issuance of phytosanitary certificates for agricultural commodities. At this time the import of living plants, seeds, and other plant materials is governed by special licensing procedures and by the policies for registered importers and exporters. All imports of seeds, plant materials, fruits, etc., are regulated under the Regulation of Import into India Order of 1984.

The government of India has also appointed three national agencies for the introduction of planting materials into India:

1. National Bureau of Plant Genetic Resources (NBPGR), which has two stations: the Division of Plant Quarantine, NBPGR, in New Delhi, and the Regional Quarantine Station, in Hyderabad
2. The Forest Research Institute, Dehradun
3. The Botanical Survey of India, Calcutta, for Ornamental and Other Plants

In this chapter, we would like to discuss some of the important fungal, bacterial, viral, and nematode plant pathogens introduced in India. Some of the alien plant pathogens causing losses in agriculture production in India are depicted in **Figure 9.1**.

Table 9.1 Plant Pathogens Introduced in India from Outside and Presently Causing Significant Economic Losses in Crop Production

Pathogens/Disease	Host	Origin	Year
A. Fungal Pathogens			
Hemileia vastatrix (coffee rust)	Coffee	Sri Lanka	1870
Phytophthora infestans (late blight of potato)	Potato	Europe	1883
Urocystis tritici (flag smut of wheat)	Wheat	Australia	1906
Plasmopara viticola (downy mildew of grapes)	Grapes	Europe	1912
Scleropspora phillipinensis (downy mildew of maize)	Maize	S.E Asia	1918
Urocystis cepulae (onion smut)	Onion	Europe	1958
Pyricularia oryzae (blast of rice)	Rice	S.E.Asia	1918
Phytophthora parasitica var. *nicotiana*e (black shank of tobacco)	Tobacco	Dutch East	1938
Synchytrium endobiotichum (wart of potato)	Potato	Netherlands	1953
Helminthosporium oryzae (leaf spot of rice)	Rice	Japan	1919
Erysiphe cichoracearum (powdery mildew of cucurbits)	Cucurbits	Sri Lanka	1910
Oidium hevae (powdery mildew of rubber)	Rubber	Malaya	1938
B. Bacterial Pathogens			
Xanthomonas campestris pv. *oryzae* (bacterial blight of rice)	Rice	Japan	1951
C. Viral Pathogens			
Bunchy top of banana	Banana	Sri Lanka	1940
Peanut stripe virus	Groundnut	China	1984
Cotton leaf curl	Cotton	Pakistan	1996
D. Nematode Pathogens			
Heterodera rostochinensis (golden nematode of potato)	Potato	Europe	1961

9.2 Fungal pathogens

9.2.1 Phytophathora infestans (late blight of potato)

One of the worst diseases of potato, *P. infestans*, takes a heavy toll year after year in many countries. It made history in Europe by causing the widespread famine during 1845, which resulted in a mass migration of people from Europe, especially from Ireland.

Butler[3] and Lal[4] reported the earliest outbreaks of late blight of potato in India. The disease was observed in the Nilgiris between 1870–80 and then in Darjeeling in 1883,

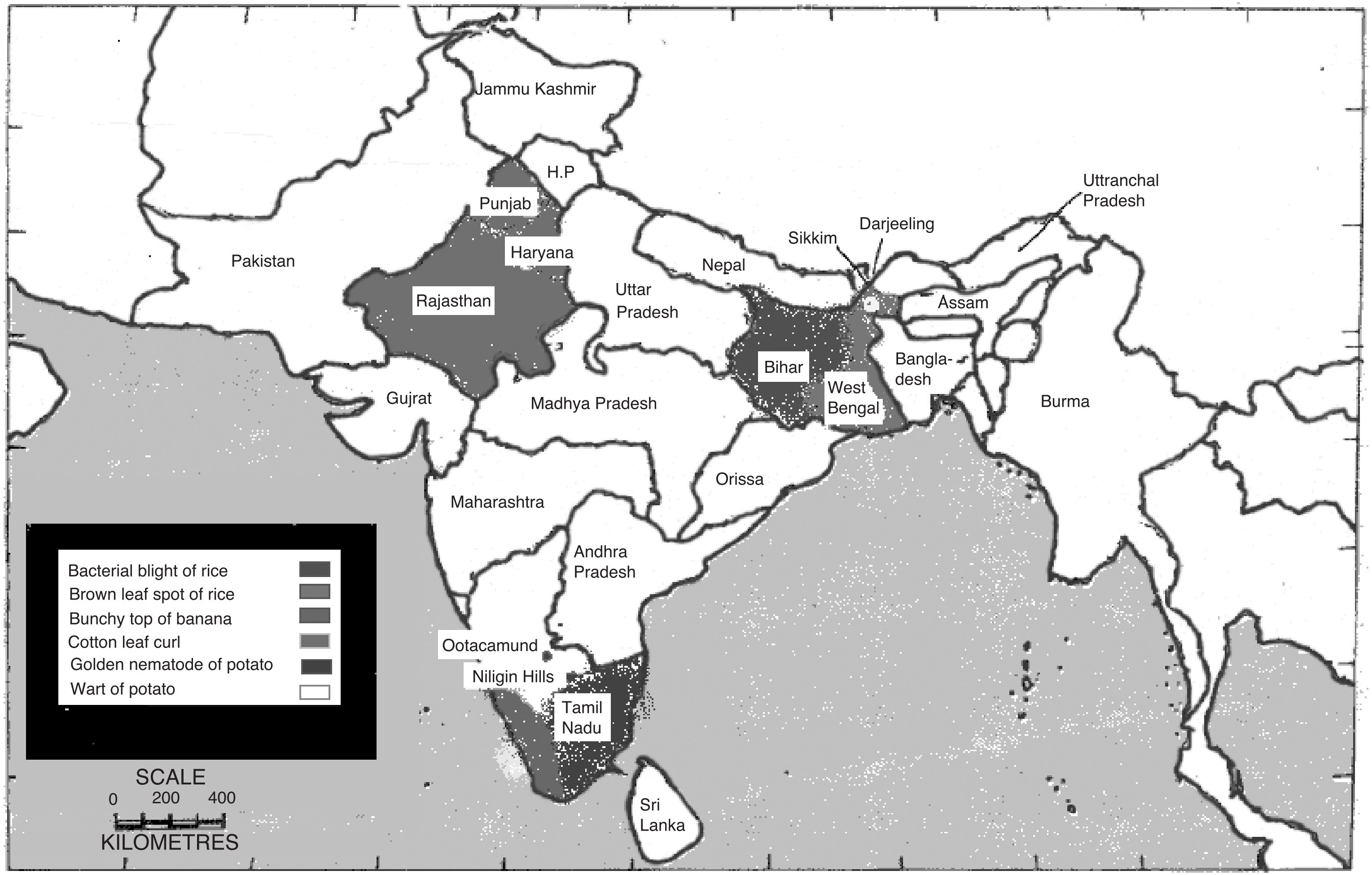

Figure 9.1 Alien plant pathogens established in certain parts of India and causing significant loss due to certain epidemics.

Figure 9.2 Infection of *Phytophathora infestans* on foliage and tubers of potato.

evidently having been introduced with the potatoes initially imported by the Europeans, and the disease spread to other places in the hills and plains with popularization of the crop. Now the disease commonly occurs in potato-growing areas in India and can cause a nearly 70% reduction in yields in susceptible varieties in epidemic years. The disease appears on foliage as well as on tubers (Figure 9.2). The early infection may cause 100% loss in a particular field.

Epiphytotics of late blight are quite common in Punjab. The state witnessed a serious epiphytotic of potato late blight due to continuous and long spells of intermittent rains and foggy weather conditions during the 1997–98 crop season. The disease was first noticed in certain sections of the district of Hoshiarpur in November 1997, and it soon assumed epidemic proportions. Severe attacks, with disease severity up to 100%, were observed in the districts of Hoshiarpur, Jalandhar, Amritsar, Kapurthala, Fategarh Sahib, and Patiala. Losses in tuber yield varied from 15 to 100%, depending on the variety and the stage at which the crop was attacked, and the plant protection measures used by farmers.[5] During the 1987 crop season, another bad year, the losses ranged from 20 to 25% in Punjab, 40 to 45% in Haryana, 15 to 50% in Uttar Pradesh, and 5 to 10% in Bihar and West Bengal (Figure 9.3). According to India's Central Potato Research Institute, 1 to 38% of the freshly harvested tubers are infected, and 0.1 to 0.3% of the tubers carry infection to cold storage.[6] Today this disease occurs regularly in moderate to severe forms in the Himalayan Hills, the Indo-Gangetic plains, and the Nilgiri Hills of Tamil Nadu.[7]

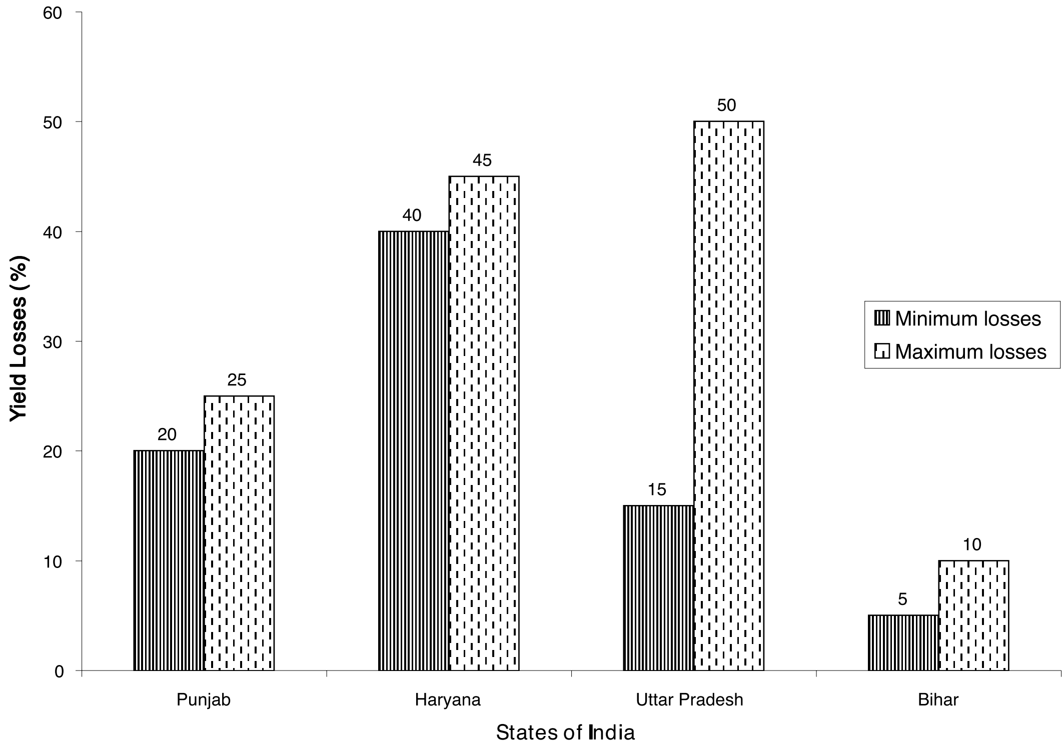

Figure 9.3 Losses due to late blight of potato (*Phytopthora infestans*) in various parts of India during epidemic year 1987.

9.2.2 *Synchytrium endobioticum (potato wart)*

Potato wart is a very destructive disease. It is believed to be indigenous to South America, from where it spread to Europe; it was known in England in 1876.[8] The disease is also known as black wart, black scab, potato canker, and cauliflower disease in various places. In India, the disease was first reported on an imported variety, Furose, in a small field at Rangbul Farm in Darjeeling in 1952.[9] Although immediate steps were taken to eradicate it by various means,[10] which included the burning of infected crops and chemical disinfection of the soil, yet the disease was observed on potatoes in markets of that district in subsequent years. In Darjeeling, almost 50% of the potato-growing area had been invaded by 1958.[4,11] Indeed, potato wart is the second most dangerous disease of the potato. Accordingly, a ban was imposed on the movement of potatoes from West Bengal in 1959, and now the disease is confined to districts in Darjeeling.[12.]

9.2.3 *Urocystis tritici (flag smut of wheat)*

U. tritici causes flag smut of wheat, a disease that is prevalent in almost all the important wheat-growing areas of the world. It was first identified in Australia in 1868.[13] In India, Butler in 1918 was the first person to report this disease, finding it in Layallpur, now in West Punjab (Pakistan). It was believed that the disease was introduced from Australia.[14] In India, the disease has been observed in Punjab, Madhya Pradesh,[15] Delhi, and Rajasthan.[16] Although its occurrence in the country is not widespread, its importance cannot be underestimated. It is long-persisting, unlike most smuts, and if sowing methods and environments are favourable and susceptible varieties are grown, the disease may assume a serious form. Reduced yield due to complete loss of productivity of infected plants is the most significant effect of the disease. Bedi[14] estimated that the incidence of disease in some parts of Punjab, Himachal Pradesh, and Haryana is as high as 75%.

9.2.4 Helminthosporium oryzae (brown leaf spot of rice)

The *H. oryzae* pathogen is causing disease in almost all rice-growing areas of the world. In India, the disease is prevalent in all rice-growing areas, especially in heavy monsoon areas in the West Bengal, eastern parts of Uttar Pradesh, Assam, Tamil Nadu, and parts of Kerala. The disease has been known to cause enormous losses in the leaf-spotting phase, when it can assume epiphytotic proportions. The first report of the disease in India was from Madras in 1919, and now it is reported from all the rice-growing areas. Two major epidemics of the disease have been recorded in India. The most recent of these outbreaks was in Bengal in 1943, when losses in yield as high as 90% were recorded.[17] "Nothing as devastating as the Bengal Epiphytotic of 1943 has been recorded in plant pathological literature," Padmanabhan[1] writes of the great Bengal Famine. Bedi and Gill[18] reported losses in the weight of grains ranging from 4.6 to 29%. According to Kawada,[19] India's annual losses in yield are 22,000 to 28,000 tons.

9.2.5 Pyricularia oryzae (rice blast)

The occurrence of blast has been suspected for as long as rice has been cultivated, the disease being known by different names in various countries. Early records for blast are mainly from China, Japan, and Italy.[20] On a global basis, it is still the number one disease of rice because of its destructiveness; it can cause huge losses in yield. In India the disease was first reported in 1913, and a devastating epidemic occurred in 1919 in the Tanjore Delta in Tamil Nadu.[21] The disease is known to occur in coastal areas, and in hilly tracts of the sub-Himalayan range from Kashmir to the northeastern states of India. Usually, the maximum damage is recorded in upland rice. With the introduction of semi-dwarf, high-yielding varieties in the 1960s, the incidence of rice blast became almost insignificant, especially in the plains of North India during the Kharif season.[22]

9.2.6 Hemileia vastatrix (coffee rust)

Among the diseases of coffee, the rust caused by *H. vastatrix* is the most common and destructive.[23] The disease was reported in Ceylon (Sri Lanka) in 1868. In 1870 the rust was introduced in India and was reported for the first time from Karnataka. Thereafter, rust occurrence was recorded every year, threatening the coffee industry. The disease occurs in Karnataka, Kerala, and even in Madhya Pradesh, where coffee is grown in small tracts. Within a decade of its appearance in Ceylon, the rust paralyzed coffee cultivation to such an extent that many plantations were abandoned. A similar situation occurred in some parts of southern India, but the disease's spread was less devastating there, and coffee, particularly of the rust-tolerant cultivar *C. robusta*, continues to be grown in the area.

9.2.7 *Plasmopara halstedii (downy mildew of sunflower)*

In India, the important constraints in sunflower cultivation, particularly in the Marathwada region of Maharashtra state, is the downy mildew disease caused by *Plasmopara*.[24] The disease has been introduced to most of the sunflower-growing countries of the world, including India,[26] mainly through the seed trade.[25] The disease was first noticed in 1984 on cv. Modren in the Manjra Command area in Latur and Maharashtra. Since the disease is seed-, soil-, and airborne, it is difficult to eradicate from the area of its establishment.

9.2.8 *Sclerophthora rayssiae var. zeae (downy mildew of maize)*

Payak and Renfro[27] first reported the disease from Pantnagar, Uttar Pradesh, in 1967. There are nine different fungus species that are reported to cause downy mildew of corn.[28] Five of these, *Sclerospora phillipinensis, S. sacchari, S. maydis, S. sorghi*, and *Sclerophthora rayssiae* var. *zeae* caused economic losses of corn in Southeast Asia. The discovery of *Sclerophthora rayssiae* var. *zeae* has stimulated interest in the significance of this disease. The disease is restricted to India, and severe outbreaks have been reported from several states. Several other fungal plant pathogens introduced in India from time to time are causing significant losses and are considered to be of economic importance (Table 9.1).

9.2.9 *Phytophthora parasitica var. nicotianae (black shank of tobacco)*

This disease has been known in the Dutch East Indies for 60 years. It was introduced in India in 1938 and today occurs sporadically in every type of tobacco grown under high-rainfall or irrigated conditions in the light soils of Karnataka, Andhra Pradesh, and Gujrat. Nursery growers have sustained severe losses due to blight disease of tobacco in seedlings. During the rainy season the disease appears first on lower leaves; then, following a rain shower, the spread of disease is fast, and eventually wet rot results. In 1953, during the week preceding the outbreak of epiphytotic, the humidity was greater than 90% on all days and the temperature varied from 74°F to 89°F.[29]

9.3 *Bacterial pathogens*

9.3.1 *Xanthomonas campestris pv. oryzae (bacterial blight of rice)*

Bacterial blight was reported from the Philippines almost 60 years ago, and it remained a minor disease until 1950, when it attracted attention in Japan. In India, bacterial blight was first reported in 1951 in the Khopoli area near Bombay, but at that time the pathogen responsible had not yet been identified.[30] Srinivasan et al.[31] reported that bacterial blight is caused by a strain of *Xanthomonas oryzae*. The blight phase is most common on foliage (Figure 9.4). A detailed survey showed that the disease was present in most of the rice-growing states in India.[32,33] The disease broke out in epidemic form in the Shahabad district of Bihar in 1963. With the introduction of variety Taichung Native 1, the disease appeared in a severe form in 1966 throughout India.[34] The disease is of common occurrence throughout the Punjab, with severe losses being sustained in certain districts. There was a maximum of 47.2% incidence with 54% disease severity in Amritsar during 1994. Similarly, during the 1996 crop season a maximum of 76.4% incidence and 41% disease severity were recorded in Kapurthala districts. Bacterial blight is currently a major hurdle in stepping up rice cultivation. Losses due to bacterial blight can reach 50%.

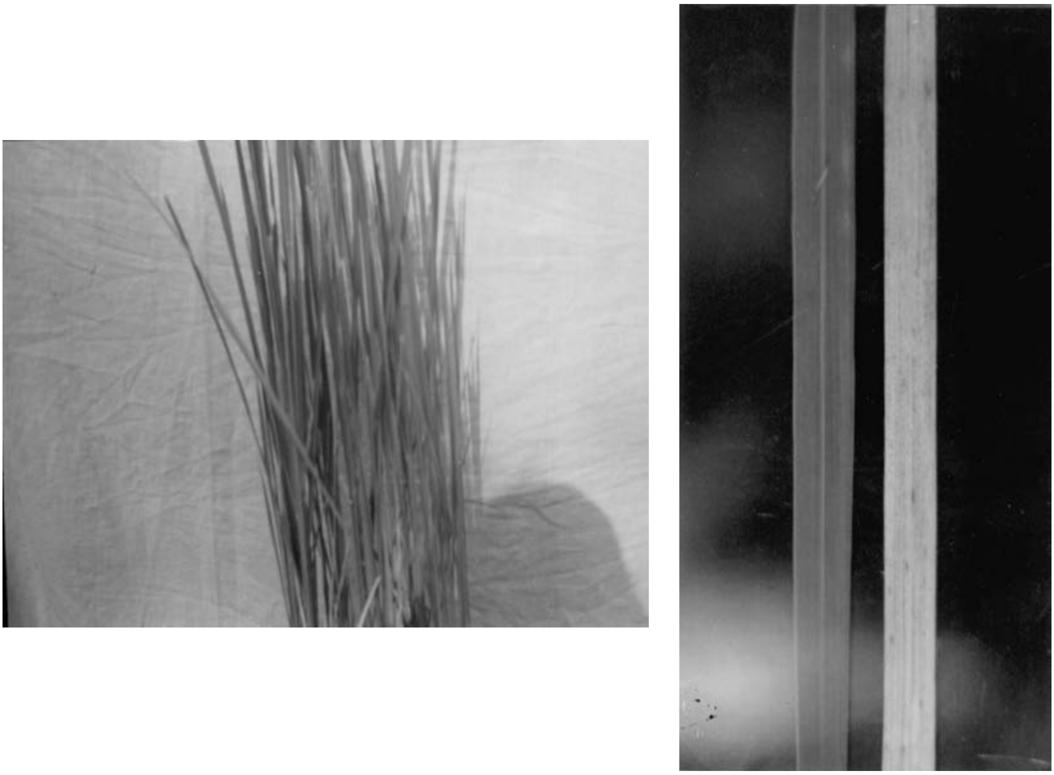

Figure 9.4 Infection of *Xanthomonas campestris* pv. *oryzae* on rice plants.

9.4 Viral pathogens

9.4.1 Banana bunchy top virus

Bunchy top virus is believed to have been introduced to India through infected suckers brought from Sri Lanka to Kerala. It was first observed in India in about 1940.[35] By 1943 it had spread to a few areas in Kottayam, and it has now affected an area of about 3000 mi^2 in Kerala. It spread to Tamil Nadu, Orissa, West Bengal, and Assam.[11] Domestic quarantine measures were followed to keep the disease from spreading to other areas. No detailed survey of the existence of the disease in different parts of India has been made.

9.4.2 Cotton leaf curl

In 1994 the virus that causes leaf curl of cotton was reported from border areas of Rajasthan and Punjab.[36] It is believed to have been introduced from Pakistan. The most conspicuous symptoms are a thickening of vein and veinlets, along with subsequent curling of the leaves (Figure 9.5); the leaflets emerged from infected foliage. There are several weed hosts of the virus; the important ones are *Peeli Booti* and *Kangi Booti*, on which it perpetuates in the absence of a cotton crop. By 1998 the disease had spread to most of the cotton-growing areas of Haryana, Punjab, and Rajasthan and was causing a lot of damage. In Punjab, during the 1995 crop season the disease was noticed in severe form in Shri Ganganagar,

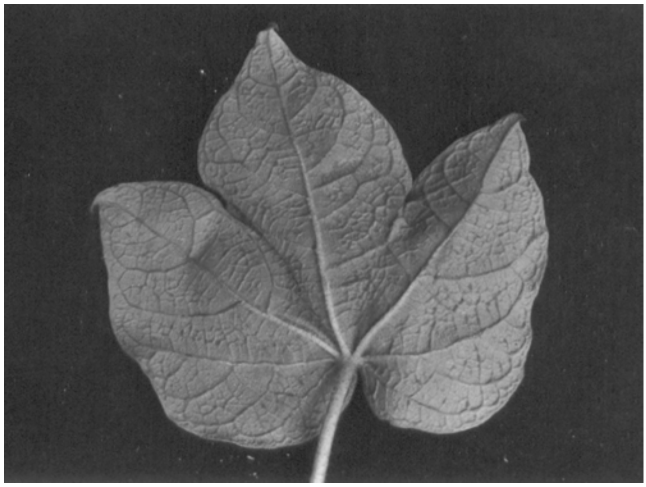

Figure 9.5 Infection of leaf curl virus depicting thickening of veins, veinlets, and leathery leaf of cotton.

Abohar, Muktsar, Malout, and Fazelika. However, in the 1997 crop season, the disease was severe in the Ferozepur, Kotakpura, Bathinda, Talwandi, Moga, Barnala, and Zeera districts of Punjab state. The best way to manage this disease is to grow tolerant cotton varieties, such as LH144 and Desi.

9.4.3 Peanut stripe virus

Peanut stripe virus (Pstv) is a seed-borne Poty virus imported from China that infects the groundnut (peanut), and it is of quarantine importance in India.[37] Pstv was first noted on peanuts in the United States, in Georgia, in 1982.[38] It has since been reported from major groundnut growing areas in Southeast Asia, including India, and it has caused economically significant crop losses. Besides groundnut, the virus infects other hosts, such as soybean, cowpea, etc. The virus is seed-borne and is transmissible mechanically by sap, as well as by *Aphis craccivora*, the insect vector.[39]

9.5 Nematode pathogens

9.5.1 Heterodera rostochiensis (potato cyst nematode)

Cyst nematode is considered one of the major diseases of potatoes throughout the world, and is popularly known as golden nematode.[40–42] Kuhan in Germany[42] first identified it in 1881. In India, the nematode was detected in 1961 from a field at Vijayanagaram state farm, Ootacamund, situated at an elevation of about 2125m.[43] The nematode is thought to have been introduced from Britain along with tubers and weeds, such as *Stellaraia media* and *Spergula arvensis*.

Detailed surveys all over the potato-growing areas of India have revealed that nematodes are confined and prevalent in the Nilgiri Hills. Seshadri and Sivakumar[41] reported that the cyst nematode was distributed in many fields around Ootacamund. The nematode was reported from the hills of Kodiakanal in Madurai district during 1971. By 1974 the nematode infestation was recorded from as much as 1939 ha, including 65 ha in government farms in the Ootacamund region. Prashad[44] reported cyst nematode infestation in

Table 9.2 Plant Pathogens of Quarantine Significance Intercepted at Various Ports of Entry and Recommended after Suitable Action

Year	Host	Specimen	Port of entry	Origin	Remarks
1977	Chickpea	*Fusarium* spp.	Bombay	Australia	Treated with Thiram 0.31+ 0.2% and released
1990	Sorghum seed	*Curvularia* spp. *Alternaria alternata*	Bombay	U.S	Recommended fungicide treatment before release
1990	Cabbage seeds	*Alternaria alternata*	Bombay	U.S	Recommended fungicide treatment before release
1990	Sunflower seeds	*Rhizopus* spp.	Bombay	Turkey	-do-
1991	Sudangrass	*Rhizopus* spp.	Bombay	U.S.	-do-
1991	Gladiolus bulbs	*F. oxysporum*	Bombay	Holland	42,000 bulbs released; 8000 were destroyed
1992	Groundnut seeds	Peanut stripe virus	Bombay	U.S.	Destroyed the consignment
1992	Maize seeds	*Fusarium graminae*	Bombay	U.S.	Treated and recommended for release
1993	Aster	*Alternaria tenuis*	Bombay	Holland	Released after treatment

3050 ha in Niligiris and about 200 ha in the hills of Kodiakanal. This nematode has posed a major quarantine problem, requiring strict vigilance in enforcing domestic quarantine.

Large numbers of seed samples and planting materials of good cultivars of agricultural and horticultural crops are being introduced in India from several other countries. Along with these materials, often a serious pathogen that is associated with the seeds or planting materials is introduced and can become a limiting factor in the cultivation of specific crops. A single serious pathogen introduced along with imported plant material can entirely offset the benefits of introducing high-yielding varieties. The alertness of plant quarantine officials has averted the introduction of several plant pathogens in imported seeds and planting materials. Ram Nath and Lambat,[45] as well as Raychaudhary,[46] listed a number of fungal pathogens from imported plant materials (Table 9. 2). At the present time, attempts are being made to accelerate food production for feeding the increasing population of India, and farmers are being trained to adopt new technologies. It is also important for Indian farmers to obtain better-quality seed, in terms of high-yielding varieties, from developed and developing countries. Continuous vigilance will be required to minimize the danger of introducing new alien plant pathogens.

9.6 Loss estimates

Estimates of losses among India's important agricultural crops have been made from time to time. In general, about 5 to 10% of the total crop losses every year can be attributed to diseases that have been introduced to India from other countries. However, this estimate may be too high in certain localities or in certain epidemic years of a disease. The Council of Scientific and Industrial Research estimated that 20 to 30% of crop production is lost due to pests and diseases of cereals, pulses, and other crops.[47] Although the disease proportion for loss estimation may vary from crop to crop, in general as much as 10% of the losses may be exclusively from plant diseases. In rice, the total loss is estimated to be nearly Rs. 9468 crores per year, and the major rice diseases responsible are brown leaf spot, blast, and bacterial leaf blight, all of them alien pathogens. Similarly, in wheat an estimated loss of nearly 1213 crores rupees is incurred per year, with the major responsible factors being the rusts and smut diseases. For cotton, sugarcane, groundnut, and maize, losses from alien plant pathogens are approximately Rs. 3105, 1500, 813, and 650 crores per year, respectively.

Several additional alien fungal, bacterial, and viral plant pathogens are responsible for damage to other, less economically important crops but are not discussed in this chapter.

9.7 Conclusion

India is primarily an agriculture-based country, and a majority of its population depends on the agricultural professions. A number of plant pathogens have been introduced in the country through seeds or other propagating plant materials from neighboring or other foreign countries. The DIP Act was passed in 1914 and subsequently modified several times. Similarly, the Directorate of Plant Protection Quarantine and Storage was established in 1946. However, it could not completely prevent the introduction of alien plant pathogens. *Phytophthora infestans* and *Synchytium endobioticum* were introduced from European countries and are responsible for significant losses in potato cultivation. The Bengal Famine of 1943 was caused by *Helminthosporium oryzae*, introduced from Japan. Bacterial leaf blight is a limiting factor for rice cultivation in India; it, too, was basically introduced from Japan. Among the viral diseases, bunchy top of banana, which was introduced from Sri Lanka, has been established throughout the coastal region of India and is causing significant losses. Most recently, cotton leaf curl virus, which was introduced from Pakistan in 1995, is becoming a limiting factor in all cotton-growing areas of Punjab, Haryana, and Rajasthan.

The Ministry of Agriculture and Co-operation of India has formulated new policies for the import of seeds or planting material, and these efforst may check the introduction of additional serious plant pathogens. There is a need to be ever vigilant in this regard so as to minimize potential losses.

References

1. Padmanabhan, S.Y. The great Bengal famine, *Ann. Rev. Phytopath.*, 11, 11, 1973.
2. Mehrotra, R.S. *Plant Pathology,* Tata McGraw-Hill Publishing Co. Ltd., New Delhi, 1990, 771.
3. Butler, E.J. *Fungi and Diseases in Plants,* Thacker Spink and Co, Calcutta, 1918.
4. Lal, T.B. *Indian Phytopath.*, 1, 164, 1948.
5. Thind, T.S. and Mohan, C. Severity of late blight and assessment of yield losses in potato during 1997–98 epiphytotic in Punjab, *Pl. Dis. Res.,* 13, 204, 1998.
6. Anonymous, *Annual Scientific Report*, CPRI, 91, 1987.
7. Bhattacharya, S.K., Sehawat, G.S.S., and Singh, P.P. *Potato Leaf Blight Bulletin*, CPRI, 27, 46, 1990.
8. Anonymous, Potato wart disease, International Potato Center, *CIP Circ.*, 7, 3, 1979.
9. Ganguly, A. and Paul, D.N. Wart disease of potato, *Science and Culture*, 18, 605, 1953.
10. Mehta, P.R. *Indian Potato J.,* 5, 72, 1963.
11. Mehta, P.R., Sundaram, N.V., and Renjen, P.L. *Indian Potato J.,* 4, 107, 1962.
12. Dutt, B.L. *Bacterial and Fungal Diseases of Potato,* Indian Council of Agricultural Research, New Delhi, 1979, 199.
13. McAlpine, D. The smuts of Australia, their structure, life history, treatment, and cla*ssification.,* Dept. Agri. Cell Div., 224, 1, 1911.
14. Bedi, K.S., Further studies on the control of loose smut of wheat in Punjab, *Indian Phytopath.,* 10, 133, 1957.
15. Mundkar, B.B. Some races and new smuts from India. *Ind. J. Agr. Sci.*, 7: 469:478, 1944.
16. Rao, V.P. Progress in plant protection in India, *FAO Plant Protection Bulletin*, 1: 11-16, 1952.

17. Ghosh, R.L.M., Chatge, M.B., and Subramanyam, V. *Rice in India*, Council of Agr. Res., New Delhi, 1960.
18. Bedi, K.S. and Gill, H.S. Losses caused by the brown leaf spot in Punjab, *Indian Phytopath*., 13, 161, 1960.
19. Kawada, C.S. *Insects and Diseases of Rice in Japan*, Tokyo, National Institute of Agricultural Science, 1954, 33.
20. Ou, S.H. *Rice Diseases*, 2nd ed., Common Wealth Agric. Bureau., 1985, 380.
21. Padmanabhan, S.Y. Breeding for blast resistance in India, in *Rice Blast Disease*, Proc. Sym. at IRRI, Los Banos, Phillipines, Johns Hopkins Press, Baltimore, 343, 1965.
22. Singh, R.A. Current status of rice blast in India and challenges ahead. *Indian Phytopath*., 50, 186, 1997.
23. Kushalappa, A. and Eskes, A.B. Advances in coffee rust research, *Ann. Rev. Phytopathol*., 28, 139, 1989.
24. Patil, M.a., Mayee,C.D., and Phad, C.A. *Sunflower Downy Mildew Information Bulletin*, Oilseed Res. Stn., Marathwada Agri. Univ. Latur, 1982, 86.
25. Leppik, E.E. Origin and specialization of *Plasmopara halstedii* complex on the *compositae*, *FAO Plant Protection Bulletin*, 14, 72, 1966.
26. Mayee, C.D. and Patil, M.A. Downy mildew of sunflower, *Indian Phytopath*., 39, 314, 1986.
27. Payak, M.M. and Renfro, B.L. A new downy mildew disease of maize, *Phytopath*., 57, 394, 1967.
28. Waterhouse, G.M. The genus *Sclerospora*, diagnosis (or description) from the original papers and a key, *Misc. Publ*., C.M.I.,1964, 30.
29. Govindarao, P. and Koteswararao,D. Seedling blight of tobacco caused by Phytophathora *parasitica* var. *nicotianae* (Van breda de Haan) Tucker, Indian Phytopath., 9, 145, 1956.
30. Bhapkar, D.G.; Kulkarni, N.B. and Chavan, V.M. Bacterial blight of paddy. *Poona Agr. Coll. Mag.* 36, 1960.
31. Srinivasan, M.C., Thrimulachar, M.J., and Patel, M.K. Bacterial blight of rice, *Curr. Sci*., 28, 469, 1959.
32. Srivastava, D.N. and Rao, Y.P. Seed transmission and epidemiology of bacterial blight disease of rice in North India, *Indian Phytopathol*., 17: 77-78, 1964.
33. Srivastava, D.N. Epidemiology of bacterial blight of rice and its control in India, *Tropical Agr. Res. Series*, Agriculture, Forestry and Fisheries Research Council, Ministry of Agriculture and Forestry, Japan, 1, 11, 1967.
34. Srivastava, D.N., Rao, Y.P., and Durgapal, J.C. Can Taichung Native-1 stand up to bacterial blight, *Ind. Fmg*., 16, 15, 1966.
35. Verghese, M.K. Bunchy top disease of plantations. *Indian Fmg*., 6, 239, 1945.
36. Kapur, S.P., Singh, J., Chopra, B.L., Sohi, A.S., Rewal, H.S., and Narang, D.D. Cotton leaf curl in Punjab. *Pl. Dis. Res*., 9, 86, 1994.
37. Demskai, J.W., Reddy, D.V.R., Sowell, G.J.R., and Bays, D. Peanut stripe virus-new seed borne potyvirus from China infecting groundnut (*A. hypogaeae*). *Ann. Appl. Biol*., 105, 495, 1984.
38. Singh, V., Ghewande, M.P., and Reddy, P.S. Peanut stripe virus disease-percent status and *its management, International J. Pest Mgt*., 39, 422, 1993.
39. Mishra, A., Gohel, V.R., Valand, G.B., Patel, J.G., and Shukla, D.D. Peanut stripe virus disease of groundnut- review, *International J. Pest Mgt*., 39, 210, 1993.
40. Chitwood, B.G. The golden nematodes of potatoes. *USDA Circular*, 875, 1951.
41. Seshadri, A.R. and Sivakumar, C.V. The golden nematode of potatoes (*Heterodera rostochiensis* Woll., 1923)- a threat to potato cultivation in the Nilgiris (Madras State), *The Madras Agril. J.,* 1963.
42. Evans, K. and Brodie, B.B. The origin and distribution of the golden nematode and potential in the USA, *American Potato J*., 57, 79, 1980.
43. Jones, F.G.W. The potato eel worm *Heterodera rostochiensis* Woll. in India, Curr. Sci., 30, 187, 1961.
44. Prasad, K.K.S. and Sharma, R.K. *Survey for Plant Parasitic Nematodes*, Annual Scientific Report of CPRI, Shimla for the Year 1983, CPRI, 1985.

45. Ram, N. and Lambat, A.K. Fungi recorded on the imported seed and other plant material in India, *Indian Phytopath.*, 24, 189, 1971.
46. Raychaudhari, S.P. Seed pathology in relation to plant quarantine and seed industry in India, *Indian Phytopath*., 50, 185, 1973.
47. Saksena, S. Crop protection special : facts sheet, *Agriculture Today*, August, 2000, 21.

section six

New Zealand

chapter ten

Economic impacts of weeds in New Zealand

Peter A. Williams and Susan Timmins

Contents

10.1 Introduction

After some 200 years of European colonization, and the attendant processes of animal and plant introductions, New Zealand today ranks among the most highly invaded areas on earth. We have the largest number of introduced mammals of any country in the world, and the second highest number of birds.[1] At least 1800 exotic plant species have become naturalized, resulting in an equal proportion of exotic to native species in the wild flora.[2] The primary industries of agriculture, horticulture, and forestry are based on a total of only 140 species,[3] nearly all of them introduced. In contrast, there are about 500 introduced plant species that threaten these industries or the native biodiversity, or both. There appears to be no slackening of the rate at which these introduced plants are naturalizing either, for this has been a steady 12 per year since records first began about 150 years ago.[4] Statistics such as this, and the huge annual costs for the control of animal pests — such as rabbits (*Oryctolagus cuniculus*) and the Australian brush-tailed possum (*Trichosurus vulpecula*), which destroys forests and is the major vector of tuberculosis in cattle (bovine TB) — recently prompted an attempt at quantifying the costs of pests to New Zealand.[5]

0-8493-0836-4/02/$0.00+$1.50

The study showed that the New Zealand economy loses about $400 million* a year due to existing exotic pests, both plant and animal species. A further $440 million is spent each year on preventing any increase in these losses, mainly by border surveillance and control of existing populations. These combined costs of about $840 million, plus the damage caused to the conservation estate, are close to 1% of New Zealand's gross domestic product (GDP). Extrapolations from these figures indicate that in present-value terms (at discount rates of between 5 and 10%), the ongoing annual economic cost of 1% of GDP is equivalent to a capitalized burden of $9 billion to $19 billion.[5] This estimate represents the value to New Zealand of eradicating all existing pests across all sectors and ensuring no need for further costs for control or prevention of pests, apart from border control.

Neither these figures, nor much of the reported analyses,[5] distinguished the individual costs attributable to a wide range of phyla, including viruses, fungi, insects, plants, and animals. Here we aim to produce, for the first time, the economic costs of weeds, or "plant pests" in the parlance of New Zealand's biosecurity legislation. First, we describe how we estimated the total costs for weeds across all sectors of the economy. Second, we describe the cost of weed management on the public conservation estate in New Zealand. Third, we provide examples of the economic costs of individual weed species to other sectors. We acknowledge that there are economic benefits of weeds as well. These range from *Pinus* species, which are the mainstay of the forestry industry, to the pollen provided by scrub weeds to the honey industry, but we do not consider these benefits further.

10.2 The national total

In analyzing the impact of pests on New Zealand's economy, Bertram[5] distinguished between two major components. The first is *defensive expenditure*, the financial cost of resources devoted to preventing pest plants from entering the country and to controlling the populations of those already here. The second is the *loss of economic output* that is foregone each year as a result of the existing level of infestation. The Bertram study acknowledged, but did not attempt to quantify, the "welfare loss" of existing pest populations on such values as damage to indigenous biodiversity. They assembled information from central government, regional councils, and the private sector.

The total pest-related defensive expenditure by central government for 1999 was $151.5 million. Expenditure on vectors of bovine TB (mainly possums) amounted to $19.3 million (13%), leaving $132.2 million (87%) for all other pests, including plant pests. Included here are costs for border quarantine, pest surveillance and response, and pest control on the conservation estate (public lands) for the protection of indigenous habitats and species. A further $40 million of defensive expenditure was devoted by central government to scientific research aimed at animal and plant pests, of which possibly 13% was devoted exclusively to bovine TB. The total for central government expenditures on all pests, excluding those related to bovine TB, therefore comes to $167 million.

The legislation covering pests underwent major changes in the late 1980s, culminating with the Biosecurity Act of 1993, which redefined the role of central government, local government, and property owners in the control of plant pests, largely by removing subsidies to property owners and making the owners primarily responsible for weeds on their land. Some of the funds spent by local government are derived from central government subsidies to the regions; the net figure for local government expenditures was estimated at about $22.5 million for all pests in the 1998 financial year, with an

* All reported figures are in $NZ($US0.5) and have been inflation adjusted to year 2000 except for Bertram (1999)[5] data, or when otherwise indicated.

additional $20 million funded by the central government.[5] We determined the proportion of this $42.5 million devoted exclusively to plant pests, as explained below.

Agriculture is of great importance to New Zealand's economy, and the industry spends large sums on pest control, mostly on the part of the private sector. During the mid-1990s, the agriculture sector purchased goods and services (non-factor costs) related to weed and pest control to the tune of about $180 million per year.[5] These costs would be comparable today. A further $40 million is probably spent on household weed and pest control, plus all the costs of clearing weeds by local governments from city parks and reserves, waterways, and so on. We assume that farmers and the government, in equal proportion, spend about 10% of that $220 million total on bovine TB — a conservative estimate, since not all areas have bovine TB. This brings the private sector's total defensive expenditure on animal and plant pests, excluding bovine TB, to $198 million.

The total figure for central government, regional councils, and the private sector on defensive expenditures on all pests exclusive of bovine TB therefore comes to $167 million + $42.5 million + $198 million, a total of $407.5 million. How much of this was for weeds ?

We assumed that the proportion of money spent on average by regional councils on plant pests, as opposed to other pests, is a reasonable indicator of the proportion spent by central government and the private sector, exclusive of expenditures on bovine TB. These proportions are likely to differ widely among the regional councils because of the different mixes of urban and rural land, and the uses of that land, so we used an average figure. We queried the 15 regional and local government agencies in New Zealand and received replies from 12. All of the respondents provided figures (budgeted or actual expenditure) for the 1999 financial year, and five provided data going back to 1991. Expenditure on weeds as a proportion of expenditures on all plant and animal pests averaged 14.7% (range 7 to 30%) and was lowest in areas with major animal problems, particularly possums and rabbits. This average figure was used to derive the amount spent on weeds for the three non-respondents from data on all regional council pests.[5] The average sum for 1999 was $4.08 million, and it has historically been at about this level in inflation-adjusted terms all the way back to 1991. Most of the funds are spent implementing the Regional Pest Management Strategies (RPMS), wherein the councils state their objectives with respect to a defined list of weeds. These range from those they are attempting to exterminate to any emerging weeds they feel they need to monitor. The $4.08 million mostly covers monitoring, education, enforcement of regulations covering boundary clearance, and review of the strategies themselves. Very little is spent by regional councils on actually killing plants, which is the responsibility of the landowner, and is a cost borne by the private sector.

Returning to the national figure for all pests of $407.5 million, and applying the regional councils ratio of 14.7%, the total national figure for defensive expenditure on plant pests, or weeds, is $59.9 million. The annual *defensive expenditure*, the amount spent for protecting the country against weeds, therefore comes to roughly $60 million.

To determine the economic damage directly attributable to the pests, Bertram[5] gave detailed accounts of individual species of plants and animals, to produce a total figure of $400 million. He suggested this might be an underestimate, a view that is supported by examples of individual pests we discuss. For example, no account was taken of the direct losses caused by weeds to the forestry industry, as will be described below. However, even if we take a parsimonious view in calculating loss of economic output attributable to weeds, say, 10%, the figure comes to $40 million. When added to the $60 million for defensive expenditure, the total cost of weeds to New Zealand is $100 million per year.

10.3 The conservation estate

The Department of Conservation (DOC) manages 8 million ha of protected land, wetland, freshwater, and coastal marine and island sites, amounting to 30% of New Zealand's land area. Several Acts of Parliament set forth the objectives, approaches, and criteria for protecting these areas, and the control of invasive weeds is one objective. The DOC is not required to manage weeds other than in these places, but it contributes on a wider scale through involvement in strategies drawn up by central and local government under the Biosecurity Act.

Of the 2000 protected natural areas classified as having high conservation value, more than half require weed control to protect their values. A further 300 sites have recently been identified as potentially threatened by weeds over the next 5 to 15 years if weed invasions are not controlled. The high-priority areas under threat sum to 575,600 ha, or about 7% of the total land under DOC management.[6] Most of these are in tussock lands and other kinds of grassland, and in montane to alpine communities (324,000 ha). Forest and scrub is the next largest category with 152,800 ha, and the remainder is scattered over a range of coastal and duneland communities, freshwater and saline wetlands, geothermal sites, and arid sites. Many components of the ecosystem are impacted by these weeds, especially threatened plant species. Weeds are the main risk to 61 of the 125 indigenous species with high priority for management.[7]

Total expenditure by DOC on weed control has risen from $1.56 million in 1994–95 to $3.5 million in 1999–2000. This is all defensive expenditure[5] and excludes the economic value of biodiversity loss. The proportion of all pest expenditure spent on weeds comes to 25%. This percentage is toward the top of the range spent by regional governments and reflects a greater emphasis by DOC on protecting biodiversity, in contrast to the focus on animal pests of agricultural land. The area directly treated by DOC in 1999–2000 amounted to 163,000 ha, which represents about 2% of all land under DOC control. Nevertheless, this expenditure is only half the $6 million required to implement the departmental strategic plan for managing invasives.[6] For example, in the Nelson–Marlborough region at the northern end of the South Island, there are about 60 important conservation sites requiring weed control; a third have adequate control, a third have inadequate control, and a third have no control. This disparity between the size of the problem and the available money is not surprising, given the expense of weed control.

An analysis of 10 weed control projects throughout New Zealand conducted by DOC and involving a range of ecosystems and weed densities defined the costs. Light infestations of weeds, where monitoring with or without some control can be conducted over large areas, can be treated for as little as $10/ha. Once the infestations become well established, the costs increase tenfold, to $100–$200/ha, and also require $50 to $100/ha for follow-up and maintenance. Very dense infestations in difficult habitats necessitating helicopters, intensive labor, expensive chemicals, or a combination of all these, require a further tenfold increase in expenditure, from $1000 to $2500/ha (DOC, unpublished data). These logarithmic increases in costs can occur over only 5 to 8 years in the case of woody species, such as broom (*Cytisus scoparius*) and lodgepole pine (*Pinus contorta*), with either readily dispersed seeds or a persistent seed bank.

If the assumption is made that $6 million represents a realistic figure to reduce the impacts of weeds or even maintain the status quo, then the present expenditure of only $3.5 million means that the costs of treating many infestations will be increasing logarithmically. For example, the untreated lodgepole pine invasions of the montane basins in the South Island, which presently require hundreds of thousands of dollars each year to prevent canopy closure over huge areas, will require tens of millions to treat in a few decades.[8]

The above figures are just the cost of trying to restrict the distribution of weed populations (defensive expenditure); they do not accommodate the cost of biodiversity loss (loss of economic output). The total (direct, indirect, and passive) value of New Zealand's land-based biodiversity was estimated to be $43 billion in 1994, of which $26 billion is attributable to the conservation value.[9] Using the above figure, i.e., that high-conservation-value areas directly threatened by weeds represent 7% of the land managed by DOC, then, conservatively, weeds cause a loss of native biodiversity worth $1.8 billion.

10.4 Gorse

A spiny European shrub, gorse (*Ulex europaeus*) was introduced into New Zealand with the first colonists of the early 19th century. Gorse spines deter grazing; adult bushes recover from damage; and a persistent seed bank gives rise to a mat of seedlings following fire. These attributes have enabled gorse to cover large areas of farmland, particularly in the hill country. On conservation land it has often replaced the native *Myrtaceous* species as the first woody species to emerge in vegetation successions back to forest.

Gorse was widely planted for hedges, as noted by Charles Darwin on his first visit here in 1835, and no doubt it escaped soon after. Gorse has been the major weed of concern to farmers for almost 100 years, for it was ranked as an "important weed" in the first national surveys of weeds in 1917.[10] By 1980 it contributed to scrub that covered 941,300 ha, some 3.5% of the total land area of the country.[11] It was reported via a mail survey to be a serious problem for 34% of all South Island farmers, and a minor problem to a further 40%.[12] Gorse and broom are among the five most serious weeds impacting on the survival of threatened plant species in New Zealand.[7]

The costs of gorse control are available from a period in New Zealand history prior to the mid-1980s, when weed control on farmland was subsidized by the state and such data were summarized.[13] During the early 1980s, the annual subsidy for spraying gorse averaged $8.4 million. Combined with on-farm costs, and expenditures by other parties not eligible for the subsidy, such as territorial authorities, the costs of gorse control for 1984–85 was $29.1 million.

We have not collated the costs of gorse control to the Department of Conservation, but these are likely to run into many thousands of dollars. One reason there is less gorse control than might be expected, considering the extent of aerial efforts on conservation land, is that once gorse occupies a site that formerly supported native forest, the vegetation usually reverts back to indigenous vegetation in about three decades if it is not burned again. The economic impacts of gorse have been underestimated,[5] because there are undesirable effects of gorse on non-forest ecosystems that are not naturally involved in a succession, such as open land with endangered species.[7]

Some of the best data on the direct and indirect costs of weeds in New Zealand have been produced by the plantation forestry industry, which relies heavily on radiata pine (*Pinus radiata*). Gorse is a major weed of these plantations, particularly between rotations, when the seedlings are planted. Furthermore, the spread of gorse and other scrub weeds was, until recently, a major factor in large areas of high country making the change from pastoral farming to forestry. The costs of clearing gorse are incurred during the initial site preparation phase, and over the first few years immediately following planting. These costs, and the additional costs incurred during subsequent thinning regimes, totaled $13.1 million in the 1980s.[14] Since the 1980s,the annual area of cut-over in New Zealand has approximately doubled,[14] with the result that the total gorse control costs to forestry in New Zealand will now be greater.

Oversowing of cut-over land with introduced legumes and grasses to suppress weed growth reduces forest establishment costs during the first two years from $275/ha to

$355/ha. This was practiced over 12,850 ha of cut-over land (48% of the national total) in New Zealand in 1993,[14] which equates to $5.3 million over a 2-year period. No data are provided on the treatment of the remainder. However, while oversowing is not as widely practiced today, virtually all forestry land is now given some form of weed control.[15] The cost to forestry for weed control in post-harvest site preparation, and over the following two years, would be on the order of $6.0 million annually. This does not include the initial costs of clearing the land of weeds, the expense of maintaining access tracks, and regulatory compliance costs. These costs are substantial and resulted in the largest forestry company in New Zealand spending $8 million on all aspects of scrub weed control in 1999.[15a] Scrub weeds other than gorse are included here, too, but gorse is a major contributor. These data are mostly defensive expenditure, sensu Bertram.[5]

Estimates of loss of economic output to forestry attributable to weeds are derived from the work of Richardson and West.[15] They calculated the economic benefit of weed control in the early stages of tree growth, as reflected in gains in timber volume. Data are available only up to the mid-rotation (15 to 18 years) stands, when the gains for controlling weeds are likely to be greatest when applied to tall weeds, such as shrubs (*Buddleja davidii*, *Cytisus scoparius*, *Cortederia* spp.). Depending on the site, these may be less competitive with trees in the very early stages of growth, but they influence tree growth for a longer period of time than short-lived herbaceous weeds. Gains from weed control amounted to approximately 23 m^3/ha/yr of wood, and the relative value of gains was greatest on low-productivity sites because of the effect on the proportion of high-value logs. Overall, weed control may be equivalent to 1 to 4 years worth of extra growth. Based on the economics of log production in 1993, this equates to between $334/ha and $1790/ha at net present value (NPV), and depending on a range of other factors, the cost-benefit ratio of these gains from weed control ranges from marginal to extremely economic.[15] If we take the figures provided by these authors,[15] and assume 50% of the remaining cut-over was also treated, then the NPV gain from weed control in that year for increased wood production alone was between $5.8 million and $30.1 million, with a mean of $20.6 million for New Zealand as a whole. Conversely, had the control work not been undertaken, this figure would represent the loss of wood value due to weeds. Interestingly, this figure for lost production is close to the figure of defensive expenditure ($13 million) spent on gorse for the initial site preparations and release-cutting costs calculated a decade earlier.[13]

10.5 Broom

Scotch broom (*Cytisus scoparius*), the second most important woody weed after gorse, infests a wide range of land-use classes and presents a problem on approximately 1% of New Zealand's land area.[16] Jarvis et al.[16] conducted an economic study of broom as a prerequisite to allowing the entry of a beetle (*Gonioctena olivacea*) as a biocontrol agent. A measure of the importance of this weed is that regional council expenditures, and the application costs, in 1999 came to $500,000, predominantly in the South Island.[16] Jarvis et al. derived an estimate of the total costs and benefits likely to accrue to a biological control program, using data for central government subsidies on broom control similar to those used in the gorse study. These subsidies came to about $600,000 per year for the period 1981–85, and, together with all associated non-subsidy on-farm costs, the total expenditure was $1.6 million.[16] Most on-farm costs are incurred during land development, which increases farm productivity. Estimates of net benefits per year that could be derived if all the grazing land in New Zealand were developed for agriculture were calculated to be the savings on the costs of control ($1.8 million), plus the share of the increased productivity attributed to this control ($6.5 million), a total of $8.3 million. As an alternative means of assessing the costs of broom, changes in the market value of all land in New

Zealand infested with broom, before and after development, were calculated. Combined with the savings in ongoing broom maintenance, this gave a figure of $4.3 million, which was an estimate of the lower boundary of the benefits of broom control on farmland.[16]

The economic importance of broom to forestry is indicated by the $1.3 million spent in 1999 on broom control by the 17 major forestry companies, which managed a total of 1.6 million ha in that year.[16] Broom has a particularly severe impact on production volumes in the 8000 ha of dry production forests in the eastern part the South Island. If broom causes a productivity loss similar to that of gorse, i.e., a loss of 2 years of production, then the annual production loss, net of harvesting and freight costs, is $600,000.[16] These estimates show that even when weed species such as gorse and broom have only slightly different ecologies, and thus control regimes, their costs to the forestry industry are additive.

Broom is widespread on conservation land. Like gorse, broom is among the five most serious weeds jeopardizing the survival of threatened plant species in New Zealand, since it changes naturally open communities into woody cover. The cost to control all the broom would be enormous, so the DOC controls it only on high-value sites. Further, the broom must be compromising the conservation values at these sites, and the achievable level of broom control must result in an increase in biodiversity at the site. In 1999 the DOC spent $500,000 on broom control. This expenditure is an underestimate of the true conservation cost of broom; the true figure would be orders of magnitude higher. An additional $700,000 was spent on controlling broom in 1999 by other Crown agencies with land under their control and by utilities to control broom around power lines and highways.[16]

Summing across all sectors, a total of $4.7 million is spent each year on controlling broom.

These analyses of both gorse and broom rely heavily on data from the early 1980s, when the government subsidised the costs of clearing weeds from land and the chemicals required for ongoing control. Much land that was so cleared was certainly uneconomic for pastoral farming, and it has since reverted back to weeds or been subsequently cleared again without direct subsidies and planted as commercial forests. Because most of the data on weed expenditure in forests were derived after the 1980s, there is in effect an overestimate of the amount spent on gorse and broom when projected to the present land-use pattern in New Zealand. While we are uncertain as to the extent of this bias, the total cost of shrub weeds would be very much higher if we were able to place a dollar value on its impact upon conservation land.

10.6 Old man's beard

Old man's beard (*Clematis vitalba*) is a vine that has proved to be particularly destructive of native forest in many parts of New Zealand. It damages revegetation plantings and has the potential to be a weed of pine plantations. An investigation into the economics of a biological control program involved a compilation of the existing expenditure on the weed and a study of the benefits of such research to New Zealand society.[17] This latter part is the only such study we know of in New Zealand.

There is no market value for native forest in New Zealand, as it is for the most part not harvested. The authors used a non-market valuation technique known as *contingency valuation* to elicit the community's willingness to pay for the research in expectation of the benefits they might receive. A postal survey of randomly selected adult New Zealanders was conducted, and the results suggested that the adult population was prepared to pay between $44 million and $110 million. In contrast, the total identified costs of *Clematis vitalba* control by the territorial authorities and the Department of Conservation for the 5 years immediately prior to the study (1985–90) had a net present value (in 1990 NZ dollars, at a 10% discount rate) of $3.8 million. The cost of a biocontrol research program for old man's beard was estimated at $1 million to $2 million over 6 years. The survey figures also exceed the amount being spent by the DOC and local authorities on all environmental weeds. These very large sums of money suggested for a single weed contrast to the amount being spent on controlling it, and to the even smaller amount that would be required for research into biological control.

While the figures from the survey may be an overestimate of what the public would really be prepared to pay, they show that New Zealanders place a high non-market value on their native forests and would seek to avoid any biodiversity loss. Understandably, willingness to pay was highest among those who visited native bush and were aware of the problem prior to the questionnaire (68%), and lowest amongst those who had not visited bush and were unaware of the problem (5%). It was interesting, though, that willingness to pay was not confined to the regions currently affected by old man's beard; the public considered it to be a national problem.[17]

10.7 Thistles

Thistles are a major problem for agriculture in many areas of New Zealand. Californian thistle (*Cirsium vulgare*) ranks as one of the top six weeds on the South Island, infesting a third of the farms of the southern regions.[18] The cost of control and loss of production together were estimated at $13 million for the region.[18] These two components were not separated, and in the absence of data on their relative weightings, Bertram[5] ascribed a factor of 50% to the loss of production to give a figure of $10 million. There are no data for the country as a whole, but Bertram felt that it must lie "above $10.0m and possibly as high as $20.0m."

10.8 Alternative funding

The huge sums required to control weeds, especially on conservation land, where there is no measurable economic benefit, require alternatives to general taxation. One hope of meeting more conservation objectives for weed control in New Zealand is through volunteer labor, where the DOC subsidises travel expenses. One such scheme has been in place for the volcanic mountains of the central North Island for decades. Lodgepole pine is hand-pulled over about 500 ha on a 3-year rotation. Here the direct costs, apart from those borne by the volunteers, amount to only about $1/ha, for very light infestations, and up to $13/ha for heavy infestations (50 trees/ha), which is less than 10% of the full commercial costs.

Many conservation weeds originated from horticulture,[19] including those that have only recently become invasive, for example, kiwi fruit (*Actinidia deliciosa*). If the sale and distribution of highly valuable crops were controlled through centralized commercial agencies, it would be a simple matter to levy the growers. A mere one quarter of 1 cent levy on every tray of kiwi fruit produced in New Zealand per year would yield $150,000, which would be more than sufficient to control all known wild populations of the species.[19a] If such responsibilities were accepted before new species were allowed to be released in New Zealand, there might be a greater opportunity to maintain the diversity of agricultural plant species that the country requires,[3] while at the same time protecting the indigenous biodiversity.

10.9 Conclusions

The cost to New Zealand of defending its borders against new weeds, and managing or controlling the alien species already here, amounts to about \$60 million per year. Those species that are not successfully controlled and that directly effect the nation's productive output cost a further \$40 million per year. This represents about 12% of all the economic costs of all classes of pests as estimated by Bertram.[5] Many individual weed species have direct control costs of many millions of dollars per year. Assuming 200 weed species are intensively controlled in New Zealand, the average cost per weed species in measurable lost production amounts to \$200,000 a year, which adds up to \$1 million in 5 years for each species on average.

The DOC spends \$3.5 million per year defending the 30% of the country's land area on which it manages weeds. This is a little more than about half of what is needed. Costs of weed control can increase very steeply as populations spread, and on conservation land, and on much private land, weeds are undoubtedly increasing beyond the capacity of the currently allocated resources. Some relief could come if part or all of the cost of weed control were borne by those who gain from the cultivation of a plant species that has subsequently become weedy on conservation land. In theory, the public is prepared to pay considerably more than is being spent at present to protect conservation land from weeds. The public also seems prepared to offer volunteer labor to help control weeds that are of concern to conservation.

The dollar costs of controlling weeds on conservation land are but a fraction of the true costs of alien weeds to New Zealand. Not accounted for are the weed invasions that are not controlled for lack of resources, nor do the costs take into account the loss of biodiversity values wrought by weeds. The true economic cost of weeds to this weedy island country is enormous.

References

1. Anon.The New Zealand biodiversity strategy, Department of Conservation and Ministry for the Environment, Wellington, 2000.
2. Webb, C.J., Sykes, W.R., and Garnock-Jones, P.J., *Flora of New Zealand, Volume IV.* Botany Division, DSIR, Christchurch, 1988.
3. Halloy, S.R.P. The dynamic contribution of new crops to the agricultural economy: is it predictable?, in Janick, J., Ed., *Perspectives on New Crops and New Uses*. ASHS Press, Alexandria, USA, 1999, 53.
4. Lee, W.G., Williams, P.A., and Cameron E., Plant invasions in urban environments: key to limiting new weeds in New Zealand, in *Managing Urban Weeds and Pests,* Suckling, D.M. and Stevens, P.S., Eds., New Zealand Plant Protection Society, Christchurch, 2000, 43.
5. Bertram, G., The impact of introduced pests on the New Zealand economy, in *Pests and Weeds,* Hackwell, K. and Bertram, G. Eds., New Zealand Conservation Authority, Wellington, 1999, 45.
6. Owen, S.J., Department of Conservation strategic plan for managing invasive weeds, Department of Conservation, Wellington 1998.
7. Reid, V.A., The impact of weeds on threatened plants, Department of Conservation, Wellington, 1998.
8. Stevens, T., Brown, D., and Thornley, N., Conservation Achievement: the Twizel area, Unpublished report, Department of Conservation, Wellington, 2000.
9. Patterson, M. and Cole, A., Assessing the value of New Zealand's biodiversity. Occasional Paper No.1., School of Resource and Environmental Planning, Massey University, Palmerston North, 1999.
10. Cockayne, A.H., Noxious weeds in New Zealand: notes on a recent enquiry, *J. Agric.(VIC)*,14, 339, 1917.

11. Blaschke, P.M., Hunter, G.G., Eyles, G.O., and Van Berkel, P.R., Analysis of New Zealands vegetation cover using land resource inventory data, *N. Z. J. Ecol.,* 4, 1, 1981.
12. Bascand, L.D. and Jowett, G.H., Scrubweed cover of South Island agricultural and pastoral land, *N. Z. J. Exp. Agric.*, 9, 307,1981.
13. Sandrey, R.A., Biological control of gorse: an ex-ante evaluation. Research report No. 172. Agribusiness and Economic Research Unit, Lincoln University, Lincoln, 1985.
14. West, G.G., and Van Rossen, R., Cutover oversowing for weed control and improved tree growth, *Weedworks 93,* New Zealand Forestry Research Institute, Rotorua, 1993.
15. Richardson, B. and West, G., Radiata pine growth trends following weed control, *Weedworks '93*, Forestry Research Institute, Rotorua, 1993.

15a. Stevens, P. personal communication.

16. Jarvis, P.J., Fowler, S.V. and Syrette, P. The economic benefits and costs of introducing a biological control agent, *Gonioctena olivacea*, for broom. *Landcare Research*, Lincoln, 2000.
17. Greer, G. and Sheppard, R.L., An economic evaluation of the benefits of the research into biological control of *Clematis vitalba*. Research report No. 203. Agribusiness and Economic Research Unit, Lincoln University, Lincoln, 1990.
18. Mitchell, R.B. and Abernethy, R.J., Integrated management of California thistle in pasture. *Proceedings of the* 46th *New Zealand Plant Protection Conference*, 1993, 278.
19. Timmins, S.M., and Williams, P.A., Characteristics of problem weeds in New Zealand's protected natural areas, in Saunder, D.A., Arnold, G.W., Burbidge, A.A., and Hopkins, A.J.M. Eds., *Nature Conservation: The Role of Remnants of Native Vegetation*. Surrey Beatty in association with CSIRO and CALM, Chipping Norton, 1987, 241.

19a. Stahl, W. personal communication.

chapter eleven

Ecological and economic costs of alien vertebrates in New Zealand

M.N. Clout

Contents

11.1 Introduction

New Zealand is one of the most isolated ancient land masses on earth, having been separated from other land masses for more than 65 million years. This isolation resulted in the evolution of high levels of endemism in both the flora and the fauna. The terrestrial vertebrate fauna was particularly unusual; dominated by birds and reptiles, it contained no indigenous land mammals apart from some small bats. New Zealand was among the last habitable land masses to be settled by humans; the first settlers were Maori people from Polynesia, who arrived perhaps as recently as only 700 years ago.[1]

An ecological catastrophe followed the arrival of people and the alien mammals they introduced. The Maori settlers brought dogs (*Canis familiaris*) and Polynesian rats (*Rattus exulans*) with them. At least 35 endemic bird species were lost in this initial settlement phase, including several large flightless species, such as moa (*Dinornithidae*), which were probably hunted to extinction within less than 100 years.[2] The Polynesian rat seems to have eliminated several species of small birds, flightless insects, and reptiles.[3]

European settlement of New Zealand started about 200 years ago, and what had been a trickle of alien species rapidly became a flood. In the past two centuries, Europeans have successfully introduced more than 90 species of alien vertebrates, including 32 mammals, 36 birds, and 19 fish (Table 11.1). Among the mammals are three further species of rodents, three mustelids, six marsupials, and seven deer species.

0-8493-0836-4/02/$0.00+$1.50

Table 11.1 Numbers of Vertebrate Species Introduced to New Zealand and Their Present Status[28,30–33]

Group	Number introduced	Number established	Number of pests
Terrestrial mammals	55	32	28
Freshwater fish	40	19	9
Birds	137	36	6
Frogs and reptiles	6	4	—

Number introduced refers to species released to the wild in New Zealand. *Number established* refers to species with self-sustaining wild populations. *Number of pests* refers to species listed in current legislation or policy documents as pests, and/or under some form of control to reduce their abundance.

Some alien vertebrates, such as sheep (*Ovis aries*) and cattle (*Bos taurus*), have proved to be economically beneficial, forming the basis of profitable farming and export industries. New Zealand's economy still depends heavily on the export of primary produce, including meat, wool, and dairy produce. Several other introduced vertebrates have, however, proved to be both ecologically and economically damaging.

11.2 Ecological costs

There have been major ecological costs flowing from many of the vertebrate introductions to New Zealand. In addition to the early losses following Polynesian settlement, at least nine endemic bird species have become extinct in the past 150 years, primarily because of predation by European mammals, such as ship rats (*R. rattus*), cats (*Felis catus*), and stoats (*Mustela erminea*). Stoats, along with two other mustelids, were introduced in the 1880s in a failed attempt to control rabbits (*Oryctolagus cuniculus*), which had become established two decades before and were proving to be severe pasture pests. Predation and competition by introduced mammals continues to threaten the extinction of many endemic species of reptiles, frogs, and birds, several of which are now restricted to mammal-free areas.[4,5]

In fresh waters, the alien fishes introduced and established in the 19th century include brown trout (*Salmo trutta*) and rainbow trout (*S. gairdnerii*), which today form the basis of important recreational fisheries. Introduced trout are widely recognized as being invasive predators in many parts of the world. In New Zealand, there is growing evidence of their negative effects on native fishes and invertebrates, including declines and local extinctions of several endemic galaxiid fishes.[6] Despite this, the negative effects of trout on New Zealand fresh-water ecosystems have received little publicity and have caused virtually no official concern, largely because of the popularity of trout fishing.

Other invasive fish, namely rudd (*Scardinius erythrophthalmus*) and koi carp (*Cyprinus carpio*), were illegally introduced to New Zealand in the 1960s.[7] Like trout, they can become the dominant fish species in waters where they thrive, and can have negative effects on native freshwater ecosystems. Unlike trout, they are not widely valued as sports fish, are officially classed as noxious, and are the subject of pest management strategies.

Brushtail possums (*Trichosurus vulpecula*), deer (*Cervus elaphus* and others), and goats (*Capra hircus*) are all significant agents of floristic change in native forests, through their selective browsing and inhibition of regeneration of many native plants.[8,9] Possums are also significant nest predators of threatened birds,[10] so these Australian marsupials have multiple impacts on native ecosystems.[5] Rats, cats, and stoats are important agents of faunal change through their predation on behaviorally vulnerable and slow-breeding native animals, such as large-bodied invertebrates and ground-feeding or hollow-nesting birds. Larger flightless birds have proved especially susceptible to mammal predation, with surviving species such as takahe (*Porphyrio mantelli*) and kakapo (*Strigops habrobtilus*) now close to extinction as a result.[11] Even New Zealand's national bird, the brown kiwi (*Apteryx australis*), is declining at a rapid rate[12] due to predation by alien mammals.

New Zealand's Department of Conservation classes 403 New Zealand taxa (species, subspecies, and forms) as threatened,[13] including 159 plants, 98 invertebrates, and 146 vertebrates. Analysis of the recent IUCN (International Union for the Conservation of Nature) Red List[14] reveals that 45 of 287 surviving New Zealand bird species are internationally listed as threatened — a higher proportion than in any other country. Forty-one of these threatened bird species are endemic, and many of them now occur only on mammal-free islands.

Native vertebrates (especially birds, but also fish and reptiles) have suffered disproportionate rates of extinction and endangerment. The primary cause in most cases seems to be predation or competition from alien vertebrates.

Alien vertebrates have clearly caused major ecological costs, through extinctions and declines of many endemic species and changes in the composition and structure of native ecosystems.

Putting a value on this damage is difficult. How does one place a monetary value on the extinction of an endemic bird such as the huia (*Heteralocha acutirostris*) a century ago, for example? It is, however, possible to estimate the ongoing costs of managing vertebrate pests for conservation purposes.

11.3 Economic costs

Bertram[15] divided the measurable economic costs of pests in New Zealand into two major components: defensive expenditures (the costs of controlling pests) and production losses (foregone economic output). A third category of costs is the "welfare loss" caused by the existence of pests, in addition to their impact on commercial activities. As an example of the latter, Bertram[15] cites the continuing damage to native forests caused by possums, deer, and goats arising from the decision that control, rather than eradication, is the purpose of defensive expenditures. Since conservation lands are largely held out of market production, the foregone values attributable to continuing pest presence can only be measured indirectly (e.g., by contingent valuation), but research on such values is currently inadequate.[15]

11.3.1 Defensive expenditures

Defensive expenditures for vertebrate pests include quarantine and border-control costs, surveillance, research, pest control, and eradication attempts. For expenditures on quarantine and border control, and on pest surveillance, it is difficult to separate out the expenditures devoted to vertebrate pests (Table 11. 2).

Despite the great concern about animals such as snakes (individuals of three different species of snakes were detected in 2000), incursions of new alien vertebrates to New Zealand are relatively uncommon. It is therefore reasonable to suppose that most of the $30 million (sums in this chapter are expressed in New Zealand dollars) spent on quarantine and border control and the $25 million spent on pest surveillance and response in 1998–99[15] was expended on the prevention or detection of plant diseases, invertebrate pests, etc., rather than vertebrates.

In contrast to quarantine and pest-surveillance costs, a large fraction of central government expenditure on the control of established pests is directed against alien vertebrates. Especially large amounts are expended on control of, and research into, brushtail possums,

Table 11.2 Defensive Expenditure ($NZ millions) on Pests in New Zealand in 1998/99

	Central govt.	Regional govt.	Agricultural sector	Other sectors
Quarantine/border control	30	—	—	—
Surveillance and response	—	25	—	—
Pest control	56	25	180	40
Species protection/islands	32	—	—	—
Research	40	—	—	—
Other (inc. policy advice etc.)	12	—	—	—

Figures shown are estimated totals for all alien species, not only vertebrates [15]

Data from Bertram, G. The impact of introduced pests on the New Zealand economy, in *Pests and Weeds: A Blueprint for Action*, Hackwell, K. and Bertram, G., Eds., New Zealand Conservation Authority, Wellington, New Zealand, 1999, 45.

which are primary vectors of bovine tuberculosis (bovine TB). In 1997–98 the Animal Health Board spent more than $43 million on controlling bovine TB.[16,17] More than $24 million of this was spent on management of TB vectors, mainly possums, representing a threefold increase since 1990–91.[18] Expenditure on research into possum TB control has risen sharply in recent years, from about $3 million in 1990–91[19] to $15.5 million in 1998–99 (NSSC Annual Report 1998), with most of this coming from central government sources (Table 11.3).

The New Zealand Department of Conservation (DOC) administers nearly one third of the land area of New Zealand and spends a large proportion of its budget on pest and weed control to protect biodiversity values. In 1999–2000 the DOC spent $23.9 million on animal pest control, compared with $8.6 million on weed control. The majority of the animal pest budget was spent on controlling alien mammals, dominated by possum control at $12.5 million and goat control at $5.9 million.[21] Much of the regional government and agricultural sector expenditures on pest control (Table 11.2) are also directed against possums and other vertebrate pests.

Of the $32 million spent annually by DOC on protecting species and managing island reserves, a large share is expended on managing alien vertebrates, especially predators, that are major threats to native biodiversity. The new DOC "mainland islands" program, costing $2.8 million in 1999–2000, concentrates especially on the reduction of alien vertebrates to minimal density at key conservation sites on the mainland of New Zealand. Most of the DOC expenditure on the management of alien vertebrates is for ongoing control purposes, but some of the funds spent on managing island reserves ($2.7 million per year) are aimed at the complete eradication of mammalian pests.

The eradication of alien mammals from islands has been a major advance in New Zealand conservation practice in recent years.[22] Successes on large islands include the eradication of cattle and sheep from Campbell Island (11,216 ha), goats from Raoul Island (2938 ha), brushtail possums, Norway rats, and Polynesian rats from Kapiti Island (1970 ha), Polynesian rats from Codfish Island (1350 ha), and rabbits and mice from Enderby Island (710 ha). On islands, where eradication is possible and the risks of reinvasion are low, this type of one-time expenditure to permanently remove a threat is more efficient than paying for perpetual control, with its attendant uncertainties.

The DOC also spends substantial funds on research into vertebrate pest control, including a special budget of $6.6 million over 5 years (commencing in 2000) for specific research into the control of stoats.[21]

11.3.2 Production losses

Production losses due to the damage caused by alien vertebrate pests are difficult to calculate, so estimates of these losses have not been attempted for most species in New

Table 11.3 Expenditure ($NZ millions) on Research into the Control of Brushtail Possums and Bovine Tuberculosis in 1998/99[20]

Public Good Science Fund	7.3
Ministry of Agriculture and Forestry	3.2
Animal Health Board	2.8
Department of Conservation	1.5
Other	0.7
Total	**15.5**

Zealand. Exceptions are brushtail possums and rabbits, but even here the estimates are often vague.[15,23,24]

11.3.2.1 Possums

Cowan[25] estimated that possums were responsible for losses of agricultural and forestry production, and damage to erosion-control plantings, ranging from $30 million to $60 million a year. Possum impacts on pasture production are a matter of contention, with Bertram[15] suggesting annual losses of $12 million, while others suggest that the impacts of possums on pastures are negligible.[23] There is also uncertainty about the economic significance of damage caused by possums in forestry plantations, although Butcher[23] points out that even a 5% loss at planting in a *Pinus radiata* plantation represents a loss at harvesting of more than $282/ha. Since there are in excess of 1 million ha of pine plantations in New Zealand, even very small losses at planting may translate to millions of dollars of lost production per year.

Although there is doubt about the precise level of production losses caused by possums, they probably barely match the defensive expenditures for this pest (Tables 11.2 and 11.3). This apparent paradox is partially explained by the immense *potential* damage to the economy that could be caused by possums as vectors of bovine TB. Exports of dairy produce, beef, and venison may be threatened if the persistence of bovine TB in New Zealand causes the country's trading partners to place import restrictions on New Zealand produce. Exports of dairy produce alone earn New Zealand more than $3 billion a year, so the potential damage is huge. In addition to reducing the potential for this kind of economic damage, much possum control, especially on the part of the DOC, is also justified by the protection of natural assets such as indigenous forests and wildlife.

11.3.2.2 Rabbits and hares

Losses due to rabbit damage are also a matter of contention, with available estimates varying widely. Parkes[25] estimated the total cost of rabbit damage to pastoral production, horticulture, and forestry at $6.8 million per year, but Bertram[15] argued that some 2 million sheep are displaced by rabbits, and that the total cost of rabbit damage to pastoral production was more likely to be in the neighborhood of $50 million a year. Calculation of overall costs of rabbit damage depends partly on the level of infestation, which has been affected in recent years by the illegal introduction of rabbit calicivirus disease (RCD) in 1996. This disease has at least temporarily reduced rabbit densities across large tracts of sheep grazing country, especially in parts of the South Island. In several areas the abundance of hares (*Lepus europaeus*) has evidently risen as rabbit numbers have fallen, partially replacing one alien lagomorph with another. Hares also damage young forestry plantations by browsing on seedlings, but the economic significance of this is unclear.

11.3.2.3 Other vertebrates

Among the other alien vertebrates that may cause some locally significant production losses in New Zealand are pigs (*Sus scrofa*), ferrets (*Mustela furo*), rodents, and some bird species. Wild pigs can cause local losses on some sheep farms by preying on young lambs. Along with ferrets, they are also minor vectors of bovine TB. Rats and mice are pests of stored products and have at least a nuisance value to foodstuffs industries and domestic households. In rural areas near to native beech (*Nothofagus*) forests, the abundance of mice and ship rats can reach plague proportions in years following the mast seeding of beech trees,[5] causing major inconvenience and minor economic losses. Some alien bird species, especially sparrows (*Passer domesticus*), starlings (*Sturnus vulgaris*), blackbirds (*Turdus merula*), rooks (*Corvus frugilegus*), and various finches, can cause localized damage to commercially grown fruits and other horticultural crops.

In none of the above cases are there any reliable estimates of the current overall costs of the damage in lost production. Bertram[15] crudely estimated the total production losses caused by minor animal pests (including insects) at around $36 million a year. Based on the fraction of his estimates of production losses caused by major vertebrate pests (possums and rabbits), the combined losses to minor vertebrate pests might come to about $10 million a year.

11.3.3 Other losses and side effects

In addition to the costs of defensive expenditures and production losses, the presence of alien vertebrates in New Zealand continues to degrade the natural environment and threaten native species. As mentioned previously, these ongoing ecological costs are extremely difficult to measure.

One side effect of the presence of alien mammals such as possums, rodents, and rabbits is that their control often involves the use of toxins, some of which may accumulate in food chains and affect other species, including native wildlife.[26] Without the alien mammals, there would be no need to continue to distribute these chemicals and to accept the associated risks.

There are also social costs to the presence of some alien vertebrates, stemming from the conflict that can result from disagreements over the need to control pests, or over the best way of conducting control. In New Zealand, toxic control of deer, chamois, and Himalayan tahr is not done, despite the damage these animals cause to native ecosystems, because of opposition from hunters. It would be feasible to eradicate species such as Himalayan tahr, and possibly chamois as well, from New Zealand. This has not happened because of intense lobbying pressure from hunting interests, so ongoing ecological costs are incurred for the natural environment. Similarly, but for different reasons, proposals to cull a herd of wild horses to protect fragile native plant communities in the Kaimanawa area of the central North Island met with opposition from some sectors of the public. Strong political pressure from horse lovers eventually caused the cull to be abandoned, against the recommendations of DOC officials, in favor of a more complex and costly "muster and sale" option.[27]

Public resistance to the repeated application of toxins such as 1080 (sodium monofluoracetate) and anticoagulants already precludes the use of these chemicals in some areas. A recent example is the opposition by local Maori people to the use of toxins for vertebrate pest control in the Te Urewera, "mainland island," conservation area in the eastern North Island. Because of concerns that poisoning of possums and rats might result in toxin accumulation in hunted deer and pigs, and thereby pose a risk to hunters and their dogs, the DOC has now ceased using poisons in this conservation area. All possum and rat control in Te Urewera is now conducted by trapping.[27a]

Another side effect of the presence of an alien vertebrate is that it can sometimes encourage other deliberate introductions in an attempt to control it. The classic example of this is the checkered history of attempts at biological control of rabbits. This commenced with the importation and release of mustelids in the 1880s, which was a disaster for vulnerable native wildlife and did not result in effective rabbit control. It continued with the failed introduction of myxomatosis in the 1950s and with the illegal importation and release of RCD by farmers in 1996. This latter introduction flouted recent laws, including the Biosecurity Act of 1993, that were established primarily to protect the interests of the agricultural sector of the economy.[5] Without the initial mistake of introducing rabbits, the biocontrol mistakes using mustelids and myxomatosis, and the risky introduction of RCD, would not have happened.

Finally, another set of side effects flowing from the presence of alien vertebrates, especially mammals, is their role as vectors of human diseases. Possums, rodents, and hedgehogs (*Erinaceus europaeus*), among others, are know to carry diseases such as leptospirosis and *Salmonella*,[28] which can infect water supplies and cause illness among people. The costs of these and other diseases that can be carried by alien vertebrates are difficult to calculate, but they may be significant in terms of defensive expenditures (sanitation, medicines) and production losses (lost working time). The presence of potential vectors of diseases that have yet to establish in New Zealand constitutes an ongoing health risk. For example marsupials (e.g., possums) are the main vertebrate hosts of the Ross River virus, which is carried by the mosquito *Aedes vigilax* and can affect humans. With climate warming, it is possible that *A. vigilax* could establish in New Zealand, where potential vectors already exist in the form of alien marsupials.[29]

11.4 Conclusions

In the only published estimate of the overall costs of alien species to the New Zealand economy, Bertram[15] concluded that identifiable costs of all pests and weeds amount to \$840 million per year, or 0.9% of the gross national product. Of this total cost, \$400 million was considered production losses and \$440 million was estimated to be defensive expenditures. Approximately 25%, or about \$100 million, of the production losses could be assigned to alien vertebrate pests, mostly possums and rabbits. The proportion of defensive expenditures directed against alien vertebrates is probably higher, but is difficult to estimate precisely. Much of the central government expenditure by the DOC on pest and weed control and research is directed against possums, goats, and carnivores, comprising well over 70% of total DOC expenditures in this area.[21] Expenditures on pest and weed control by the agricultural sector have a higher component of weed and insect control, but given the concern about possums and bovine TB, expenditures on control of vertebrates are likely to at least match the production losses caused by these species.

Based on the overall figures (Table 11.2) provided by Bertram,[15] and making some brave assumptions, it is possible to derive crude estimates of total defensive expenditures against alien vertebrates. Assuming that vertebrate pest control and research consumes 70% of the \$153 million of central and regional government expenditures in this area, the annual cost to these sectors could total \$107 million. Assuming that 25% of the estimated \$220 million spent on pest control by the agricultural community and other sectors[15] is expended on vertebrates, another \$60 million of costs may be incurred here. Assuming that only 5% of border control and surveillance expenditures are directed against vertebrates, a further \$2 million in annual costs may be incurred. Finally, if it is assumed that expenditures on policy advice and other biosecurity activities relating to vertebrate pests are in proportion to defensive expenditures against vertebrates in all other sectors combined (about 40% of all costs), a further \$5 million may be spent here. The total, very crude estimate, which relies heavily on Bertram,[15] is $174 million of defensive expenditures against alien vertebrates per year. Added to the approximately $100 million of estimated production losses, the total cost of alien vertebrates in New Zealand may therefore exceed $270 million per year.

Regardless of the exact figures, there is no doubt that the presence of alien vertebrates represents a major drain to the New Zealand economy. Added to these direct economic costs are the ecological costs of species extinctions and the continuing degradation of the unique biodiversity of the New Zealand archipelago.

The lesson to be learned from the New Zealand experience with alien vertebrates is simple: The breaching of natural biogeographic boundaries by introducing alien species is something that should never be undertaken lightly. The precautionary principle demands that all alien introductions, both deliberate and accidental, be considered potentially harmful unless proven otherwise, and every effort should be made to prevent introductions when there is any doubt about their impacts.

References

1. Ogden, J., Basher, L., and McGlone, M. Fire, forest regeneration and links with early human habitation: evidence from New Zealand, *Annals of Botany*, 81, 687, 1998.
2. Anderson, A. *Prodigious Birds*, Cambridge University Press, Cambridge, UK, 1989.
3. Atkinson, I.A. E. and Moller, H. Kiore in, *The Handbook of New Zealand Mammals,* King, C.M. Ed., Oxford University Press, Oxford, UK, 1990, 175.
4. Clout, M.N. and Saunders, A.J. Conservation and ecological restoration in New Zealand, *Pacific Conservation Biology*, 2, 111, 1999.
5. Clout, M.N. Biodiversity conservation and the management of invasive animals, in I*nvasive Species and Biodiversity Management*, Sandlund, O.T., Schei, P.J. and Vikens, A., Eds., Kluwer, London, UK, 1999, 349.
6. McDowall, R.M. *New Zealand Freshwater Fishes: A Natural History and Guide*, Heinemann Reed, Auckland, 1990, 553.
7. McDowall, R.M. *Gamekeepers for the Nation: The Story of New Zealand's Acclimatisation Societies*, 1861–1990, Canterbury University Press, Christchurch, 1994, 508.
8. Paton, I. Damage to native forests, in *The Brushtail Possum: Biology, Impact and Management of an Introduced Marsupial, Montague*, T.L. Ed., Manaaki Whenua Press, Lincoln, New Zealand, 2000, 111.
9. Challies, C.N. Red deer, in *The Handbook of New Zealand Mammals*, King, C.M. Ed., Oxford University Press, Oxford, UK, 1990, 436.
10. Sadleir, Evidence of possums as predators of native animals in, *The Brushtail Possum: Biology, Impact and Management of an Introduced Marsupial,* Montague, T.L. Ed., Manaaki Whenua Press, Lincoln, New Zealand, 2000.
11. Clout, M.N. and Craig, J. L. The conservation of critically endangered flightless birds in New Zealand, *Ibis*, 137, S181, 1995.
12. Heather, B.D. and Robertson, H.A. *The Field Guide to the Birds of New Zealand*, Viking, Auckland, New Zealand, 1996, 432.
13. Molloy, J. and Davis, A. Setting priorities for the conservation of New Zealand's threatened plants and animals, Department of Conservation, Wellington, New Zealand, 1994.
14. IUCN. *2000 IUCN Red List of Threatened Species*, The IUCN Species Survival Commission, Gland, Switzerland, 2000, 61.
15. Bertram, G. The impact of introduced pests on the New Zealand economy, in *Pests and Weeds: A Blueprint for Action*, Hackwell, K. and Bertram, G., Eds., New Zealand Conservation Authority, Wellington, New Zealand, 1999, 45.
16. Animal Health Board. *Annual Report for the Year Ending 30 June 1998*, Animal Health Board, Wellington, New Zealand, 43, 1998.

17. Coleman, J. and Livingstone, P. Fewer possums less bovine TB, in *The Brushtail Possum: Biology, Impact and Management of an Introduced Marsupial,* Montague, T.L. Ed., Manaaki Whenua Press, Lincoln, New Zealand, 2000, 220.
18. Coleman, J.D. The integration of management of vertebrate pests in New Zealand, *New Zealand Journal of Zoology,* 20, 1993, 341.
19. Allison, A.J. Towards a national strategy for science: the problem of possums, and bovine and cervine tuberculosis, *Ministry of Research Science and Technology Report No. 7,* Ministry of Research Science and Technology, Wellington, New Zealand, 88, 1992.
20. National Science Strategy Committee (NSSC) for Possum and Bovine Tuberculosis Control, *Annual Report,* NSSC, Wellington, New Zealand, 1998, 79.
21. Department of Conservation. *Annual Report for the Year Ended June 2000*, Department of Conservation, Wellington New Zealand, 2000.
22. Veitch, C.R. Habitat repair: a necessary prerequisite to translocation of threatened birds, in *Reintroduction Biology of Australian and New Zealand Fauna,* Serena, M. Ed., Surrey Beatty & Sons, Chipping Norton, New South Wales, Australia, 97, 1994.
23. Butcher, S. Impact of possums on primary production, in *The Brushtail Possum: Biology, Impact and Management of an Introduced Marsupial,* Montague, T.L. Ed., Manaaki Whenua Press, Lincoln, New Zealand, 2000.
24. Parkes, J. *Rabbits as Pests in New Zealand: A Summary of the Issues and Critical Information*, Landcare Research NZ Ltd, Lincoln, New Zealand, 1995.
25. Cowan, P.E. Possum control: prospects for fertility regulation, *Reproduction, Fertility and Development 8,* 655, 1996.
26. Innes, J. and Barker, G. Ecological consequences of toxin use for mammalian pest control in New Zealand: an overview, *New Zealand Journal of Ecology,* 23, 1999, 111.
27. Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M.N., and Bazzazz, F. Biotic invasions: causes, epidemiology, global consequences and control, *Ecological Applications*, 10, 689, 2000.

27a. Wright, L. personal communication.

28. King, C.M. *The Handbook of New Zealand Mammals*, Oxford University Press, Oxford, UK., 1990, 600.
29. Clout, M.N., Lowe, S.J. Invasive species and environmental changes in New Zealand, *Invasive Species in a Changing World,* Monney, H.A. and Hobbs, R.J., Eds., Island Press, Covelo, California, USA, 2000, 369.
30. Buddenhagen, C.E., Timmins, S.M., Owen, S.J., Champion, P.D., Nelson, W., and Reid, V.A. An overview of weed impacts and trends, in *Department of Conservation Strategic Plan for Managing Invasive Weeds,* Owen, S.J. Ed., Department of Conservation, Wellington, New Zealand, 1998, 11.
31. Williams, P.A. Ecology and management of invasive weeds, *Conservation Sciences publication No. 7*, Department of Conservation, Wellington, New Zealand. 67, 1997.
32. Owen, S.J. 1997. *Ecological Weeds on Conservation Land in New Zealand: A Database*, Department of Conservation, Wellington, New Zealand. 72 p.
33. Hackwell, K. Restoring an indigenous dawn chorus, in *Pests and Weeds: A Blueprint for Action*, Hackwell, K., Bertram, G., Eds., New Zealand Conservation Authority, Wellington, New Zealand, 1999, 17.

chapter twelve

Alien invertebrates in New Zealand

Nigel D. Barlow and S.L. Goldson

Contents

12.1 Introduction

An estimated 2200 species of alien invertebrates have established in New Zealand,[1] and around 90% of the country's invertebrate pests are alien species. They therefore pose a genuinely significant problem for New Zealand, both economically and ecologically. This significance is manifested by the existence of the Minister for Biosecurity in government, where "biosecurity" is the term used in New Zealand for "protection from the risks posed by organisms to the economy, environment and peoples' health, through exclusion, eradication and control."[2]

In this chapter we aim to do three things. First, we provide a brief overview of the ecological and economic impacts of naturalized invertebrates. Their ecological impact is very poorly understood, and any attempts to estimate its magnitude are based more on deduction than measurement. For this and other reasons, we consider ecological costs in

0-8493-0836-4/02/$0.00+$1.50

non-economic terms rather than in dollar values. The economic impact on primary production and health is more easily assessed, at least in theory, though it has never been attempted in any detail. Our approach is also a straightforward, high-level one, building on and refining the only previous analysis carried out.

Second, we consider some general features of invertebrate incursions in New Zealand, and in this context we discuss specific ecologically and economically significant case studies from the past 10 to 12 years.

Finally, we focus on three incursions that illustrate key biological aspects of invasions by invertebrates in New Zealand, and the ecological and economic impacts. These examples involve the lucerne pests *Sitona discoideus* and blue-green lucerne aphid *Acyrthosiphon kondoi*, the common wasp *Vespula vulgaris*, and the clover root weevil *Sitona lepidus*. The lucerne pests exemplify the typical patterns of invertebrate invasions in New Zealand and responses to them. The common wasp demonstrates ecological impacts, displacement of an existing species by an invader, and attempts at managing a well-established incursion. The clover root weevil case study documents the early response to a recent and particularly serious economic threat. The wasp and the clover root weevil arguably represent the two most significant invertebrate invaders of the past 30 years in New Zealand in terms of their ecological and economic impacts, respectively.

12.2 Overview of ecological and economic impacts

12.2.1 Ecological impacts

In general, few alien invertebrates naturally established in New Zealand appear to have had a significant impact on native biota, although there is a considerable dearth of knowledge. Certain species, like introduced social wasps, have probably had a major effect (see below), while other species, like the Argentine ant *Linepithema humile*, have the potential to do considerable damage. Among the few other recorded impacts are loss of the southern blue butterfly *Zizina oxyleyi* through hybridization with the invasive Australian common blue *Zizina labradus*.[3] Another invader from Australia, the large ichneumonid parasitoid *Ecthromorpha intricatoria*, parasitises the endemic red admiral butterfly *Bassaris gonerilla*,[4] but nothing is known about its impact on abundance. This is also the case for deliberately introduced biological control agents; all reports of their effects on non-target population densities are anecdotal. In an attempt to remedy this rather unsatisfactory situation, we have recently initiated a combined modeling and field study to determine the population dynamics and compensatory mechanisms in the red admiral butterfly, and thence to estimate the general impact of parasitism, from both deliberately introduced and naturalized parasitoids.

12.2.2 Economic impacts

As a starting point, we take the only economic analysis attempted of the costs of pests to New Zealand, including invertebrates, vertebrates, pathogens, and weeds.[5] We then correct for the likely proportion of these costs attributable to the invertebrate portion of Bertram's analysis, and amend his figures for invertebrates in the light of new data and the probable significance of pests in sectors other than pastoral agriculture, which was the main emphasis of his analysis. Finally, we consider the contribution of animal health threats, such as intestinal nematodes and ectoparasitic flies, both of which are alien invertebrates with a rather atypical invasion history. In effect, they invariably accompanied the (productive) host, so in that sense they could be regarded as simply a fixed reduction in the productive value of the stock rather than as a discretionary cost. It remains the case, however, that New Zealand possesses only a sample of the species of internal and external parasites that accompany farmed animals worldwide. A key issue now is the invasion of one country by drug-resistant nematodes from another, and the potentially devastating economic impact that can ensue. Up until now, New Zealand has avoided being a recipient, but it may have been a donor.

Excluding parasites, Bertram[5] cited the direct economic losses, due to reduced output, from some key invertebrate aliens in New Zealand as $165 million (all sums in this chapter expressed as New Zealand dollars) for Argentine stem weevil (*Listronotus bonariensis*), $20 million for clover root weevil, $9 million for rose grain aphid (*Rhopalosiphum padi*), and $1 million for wasps (*Vespula* spp.). This is a total of $195 million for the listed key invertebrates out of $358 million for the total direct losses from invertebrates, vertebrates, and weeds listed in the same table,[5] that is, about 55% of the total. Bertram then suggested that the total direct losses were likely to be nearer $400 million, to which could be added another $440 million for defensive expenditures on research and control to maintain pest impacts at their current levels. If the proportion of total defensive expenditures attributed to invertebrates is the same as the proportion of direct losses attributable to these species, relative to the corresponding values for all pests combined, then the defensive cost associated with invertebrates is 0.55×440 = $242 million.

We now update the figure of $195 million for direct costs of invertebrate aliens. First, Bertram's figures for losses from Argentine stem weevil[5] are based on the analysis by Prestidge et al.,[6] and since that time, successful biological control of the weevil from the introduced parasitoid *Microctonus hyperodae*[7] has significantly reduced the direct losses and defensive expenditure associated with this weevil, and the direct cost today is likely to be at most around $80 million. On the other hand, the figure of $20 million for clover root weevil is almost certainly an underestimate, with indirect costs higher than assumed because of the increasingly obvious significance of the pest, and direct costs probably in the range of $75 million to $450 million. This is based on the loss of nitrogen fixation from clover alone, assuming clover losses from the weevil of 5 to 30% nationwide (see below). Taking a reasonably conservative figure within this range of 20% gives an estimate of $300 million for the direct cost of this pest. Overall, therefore, Bertram's figure of $195 million for the direct costs of pasture pests can be amended upward with some confidence, possibly to around $380 million. It remains to add an estimated contribution from alien invertebrate pests of other primary production sectors, including forestry, horticulture, and cropping. In terms of 1999 export earnings, these sectors together represent about 42% of the value of pasture-based exports ($2.4 billion for forestry, $1.1 billion for horticulture, and $400 million for crops, compared with $9.3 billion for pasture.[8] So if we make the simplest assumption that direct and indirect losses from pests are approximately similar in relative terms between sectors, then the total losses will be in proportion to the value of the sectors. In other words, the total direct costs of pests are 1.42 × $380 million = $540 million, and the indirect costs are 1.42 × $242 million = $344 million. The total cost of alien invertebrate plant pests is therefore around $880 million per year.

In terms of parasites, in the mid-1990s around $90 million was spent annually on treatment for intestinal nematodes in sheep and cattle, and against flystrike in sheep.[9–11] Brunsdon[12] estimated a further $270 million, at 1988 prices, for the value of production losses from parasites in sheep, and it is reasonable to assume at least an equal amount due to production losses in cattle, goats, horses, and deer. Adjusted to present-day (2001) values using a 7% discount rate gives $145 million for treatment and $1.215 billion for production losses, or a total of $1.36 billion for the economic cost of invertebrate parasites of livestock.

The overall cost of around $2 billion compares with a central government expenditure on border control, quarantine, pest surveillance, and response (covering vertebrates as well as invertebrates) of about $50 million in 1999,[5] of which the invertebrate portion is included in the above estimate of total cost for plant pests.

12.3 Features of invertebrate incursions

12.3.1 General characteristics

The great majority (around 90%) of invertebrate pests in New Zealand are aliens. For example, in the pastoral sector the most damaging invertebrate pests (other than plant-parasitic nematodes) are clover root weevil, Argentine stem weevil (*Listronotus bonariensis*), New Zealand grass grub (*Costelytra zealandica*), porina (*Wiseana* spp.), the weevil *Sitona discoideus* and the blue-green lucerne aphid *Acyrthosiphon kondoi* in lucerne, black beetle (*Heteronychus arator*), Tasmanian grass grub (*Acrossidius tasmaniae*), and lucerne flea (*Sminthurus viridis*). All are aliens, with the exception of grass grub, which is the larva of a native scarab beetle, and porina, the generic name for larvae of several related species of hepialid moth. Both pests originally lived at low densities in native tussock grasslands.

With the exception of the clover root weevil, which is considered in more detail below, these pasture pests first established many years ago. The same is true for major horticultural pests such as codling moth *Cydia pomonella*, European red mite *Panonychus ulmi*, and light-brown apple moth *Epiphyas postvittana*. In general, though, established alien insects continue to be discovered at a rate exceeding two per year. On average, 2.2 species new to New Zealand were detected each year in forestry plantations between 1950 and 1987,[13] while the Ministry of Agriculture and Foresty (MAF) has recorded two introductions of herbivorous insects and 0.67 predators or parasitoids per year over the 3 years from June 1996 to June 1999.[13a] Alien terrestrial molluscs have appeared at a somewhat lower rate of about one every 5 years over the past 150 years.[14]

In addition to species establishing, many potentially serious invertebrate pests — for example, the Asian gypsy moth *Lymantria dispar* — have been successfully intercepted at the border. Most of the species that have eluded border controls have tended to be "hitchhikers" — not associated with commodities that are subject to international phytosanitary standards for export and inspection — and species that are not recognized as a sufficient threat for a routine surveillance system to be put in place, as is the case for fruit flies and the Asian gypsy moth, for example.

Of the incursions that occur, many have the potential to infest large parts of New Zealand, but the ecological and economic costs of others are limited by climate, and the species concerned are at the edges of their climatic ranges. For example, black beetle, a native of southern Africa, first arrived in the Auckland area in the 1930s. Its subsequent spread in New Zealand has been characterized by periodic outbreaks, on average more than 2 to 3 years every decade, during abnormally warm seasons.[14a] These outbreaks were associated with severe but localized pasture damage, usually in the most recently infested areas as its range increased. This currently spans the north of the North Island and some of the more southerly coastal fringes, closely coinciding with areas of about 13°C mean annual temperature. Further spread is unlikely under existing climatic conditions, but climate warming of 1°C could extend the beetle's potential distribution to include northern areas of the South Island.[14a]

The tropical grass webworm (*Herpetogramma licarsisalis*) has an even more bounded range, covering about 250 km^2 at the time of discovery in 1999.[15] Moreover, this range is more likely to decrease than increase, since the webworm appears to be at the extreme edge of its climatic range and may have survived over winter only because of exceptionally favorable weather. A mild winter and a moist summer encouraged good pasture growth,[15] and in spite of this the webworm survived winter only on north-facing slopes in the extreme north of the North Island[16] — effectively the warmest sites in New Zealand.

12.3.2 Specific examples

In terms of their potential for ecological or economic damage, the most significant invertebrate incursions in the past decade or so have involved the Argentine ant, the Mediterranean fruit fly *Ceratitis capitata*, the white-spotted tussock moth *Orgyia thyellina*, the willow sawfly *Nematus oligospilus*, the painted apple moth *Teia anartoides*, the clover root weevil, the southern salt-marsh mosquito *Aedes camptorhynchus*, and the varroa mite *Varroa destructor*. Of these eight species, two — the white-spotted tussock moth and the Mediterranean fruit fly — were successfully eradicated, while the painted apple moth and the southern salt-marsh mosquito are currently the subjects of eradication programs. Largely because the populations were too widespread at the time of detection, the other pests successfully established and are now the subject of sustained control and research. These case studies are discussed briefly in turn.

12.3.2.1 Argentine ant

The damage potential of this ant is well known through its documented environmental, economic, and social impacts in the United States, Australia, and South Africa. In South Africa the ant invades houses, steals honey from beehives, and disrupts the symbiotic relationship between native ants and *Protea* species that rely on the ants for seed dispersal and burial. In California, coastal horned lizard populations have declined because the Argentine ant has displaced native ants, which form a large part of the lizards' diet. The Argentine ant is an agricultural pest in California as well, since it protects aphids from attack by beneficial predators.[17] New Zealand's leaf litter and soil fauna may be at particular risk from ants because of the lack of native ant species[18] and the concomitant diversity of other groups.[19]

The Argentine ant was first discovered in New Zealand in 1990 at the site of the Commonwealth Games, and no eradication attempt was made because of the lack of effective monitoring tools and a lack of knowledge of the ant's existing distribution. It was also considered not to be a threat to agriculture. Nevertheless, concerns have been raised about possible international trade bans because of the risk of exporting infested goods to countries in Asia that are currently free of the ant.[17] Partly as a result of human activity, the Argentine ant is now established throughout Auckland and has been discovered in locations throughout the North Island and in Canterbury on the South Island.[17] Under the recently formed Biosecurity Authority within the Ministry of Agriculture and Forestry, government funding has recently been secured for a national survey of the ant's distribution and an examination of control options, including a promising new bait.

12.3.2.2 Mediterranean fruit fly

This pest had been identified in advance as a key threat to New Zealand, and as a result, a monitoring system using pheromone traps and an emergency response procedure was in place at the time the pest was intercepted. Two male flies were found in the traps in May 1996, and the pest was successfully eradicated at a cost of $5.3 million, using insecticide in targeted protein baits applied over an area of about 7 km^2.[20]

12.3.2.3 White-spotted tussock moth

In contrast to the fruit fly, the incursion by the white-spotted tussock moth was unexpected, and little was known about the species; it was not a pest in its native range, and the extent of its host plant range was unclear. Because of an anticipated impact on forestry, however, an eradication program was undertaken after the moth was reported in Auckland in April 1996. This used the biopesticide *Bacillus thuringiensis kurstaki* (Btk), aerially applied over 40 km^2 of Auckland's suburbs. Not surprisingly, there was considerable public interest and some concern. The cost of the eradication program, initially estimated at $5.14 million, eventually reached $12 million, with many more sprays, both aerial and ground-applied, than had been anticipated. Some properties received as many as 45 sprays. The level of public and political interest, the cost, and uncertainty regarding the benefits led to a ministerially commissioned review.[20]

12.3.2.4 Willow sawfly

In this case the pest was discovered too late for eradication to be possible. It was reported in Auckland in February 1997, and a delimiting survey carried out by the then Ministry of Agriculture and Fisheries found that it was present throughout the Auckland isthmus on several willow species. Research was begun on its management and control, but was compromised soon after by funding limitations.[20]

12.3.2.5 Painted apple moth

Two incursions of the painted apple moth were detected in Auckland, in May and September 1999, by members of the public. This lymantrid moth, related to the tussock moth and the Asian gypsy moth, is a native of Australia and has a wide host range, with a correspondingly serious potential to impact on the natural environment, forests, and horticulture. The two incursions were thought to be unrelated, and the presence of all stages of the moth suggested it had been there for at least a year before detection.[17] For both incursions, the Ministry of Agriculture and Forestry undertook delimiting surveys within 1 km of the infestation site, and localized ground spraying of insecticide (chlorpyriphos and deltamethrin) was carried out on infested properties. Any egg masses and flightless females found were destroyed, transport of vegetation from infested sites was eliminated, and repeated surveys were carried out, while research was initiated to identify a pheromone attractant. The cost of the response, including an amount now allocated for its continuance to July 2002, has been around $2.5 million to date.[21] A simplified cost-benefit analysis carried out by the MAF has conservatively estimated the cost of the moth's economic impact to be $47 million over 20 years, or $3 million per year with a discount rate of 7%, based only on private and public amenity and plantation forestry.[17] This analysis ignored impacts on horticulture and the natural environment, though it suggested a precautionary approach in the latter case. Given the moth's known preference for the native kowhai tree, the Parliamentary Commissioner for the Environment's report[17] interpreted this to imply that costs should be considered high until proven otherwise. The same report also suggested some biosecurity lessons from the painted apple moth incursion, notably a lack of contingency funding for a response, the need for a risk analysis for indigenous fauna and flora, the need for good communication among affected agencies and experts, and the importance of the public's role in discovering incursions.

12.3.2.6 Southern salt-marsh mosquito

This mosquito was first found in 1999 in a small area near Napier, on the east coast of the North Island, and the MAF is now initiating an eradication attempt. The mosquito is a vector for Ross River virus, an epidemic of which could cost the region $230,000 to $2.3

million a year.[17] Three other exotic mosquito species have already established in New Zealand (*Aedes notoscriptus, Aedes australis,* and *Culex quinquefaciatus*),[22] and all three have the potential to transmit Ross River virus. Potentially more serious is the tiger mosquito, *Aedes albopictus*, which can also transmit dengue fever. This species has been intercepted and successfully eradicated, both at the border in 1993, and inland in used car tires, while *Aedes japonicus, Culex annulirostris,* and *Aedes aegypti* have also been intercepted at the border.[22]

12.3.2.7 Varroa mite

Varroa destructor (formerly misidentified as *V. jacobsoni*) is a brood parasite of the Asian hive bee, *Aphis cerana*, sucking the blood of developing pupae and adults. Some time in the last century it expanded its host range to the western honeybee, *Apis mellifera*, and by 1999 had spread worldwide, with only Africa, Australia, and New Zealand remaining free.[23] The mite was found in South Auckland in April 2000, possibly introduced on a smuggled queen bee,[17] and a delimiting survey at the end of May showed it to be present throughout Greater Auckland and in four other centers of infection in the northern half of the North Island. An operational plan was developed for its eradication, but this was rejected in favor of a two-year joint government and industry management program. It was estimated that this would cost as much as $40 million,[24] of which $7.5 million has currently been allocated. The program involves treatment of infected hives, movement controls, and surveillance, with a particular effort being made to prevent the mite from reaching the South Island.[25] The estimated future cost of *Varroa* in the absence of any intervention was estimated at $400 million to $900 million,[24] or $26 million to $59 million per year, assuming a 7% discount rate. Eradication was likely to cost more than the management program, and it was rejected on the advice of a technical committee for four reasons, three of which were technical: the mite might be more widespread than previously thought; it may be impossible to treat new infections before they have spread still further; it may not be possible to eradicate all feral hives; and there may be public concerns over the environmental and health impacts of a poisoning program.[24] Widespread media coverage of the threat and the scale of the response left the New Zealand public in no doubt as to the potential seriousness of invasive species.

12.4 Case studies of invasion processes and impacts

Among them, the following three case studies demonstrate in more detail some of the biological processes underlying invasions, their ecological and economic impacts, and management responses to both new and established incursions.

12.4.1 Lucerne pests

The lucerne pests *Sitona discoideus* and the blue-green lucerne aphid exemplify the typical patterns of invertebrate invasions in New Zealand and responses to them. Analyses of invasions in the literature typically focus on spatial spread rather than on the pattern of local density changes over time. Where such data do exist, however, they sometimes exhibit two interesting features. One is a "latent period" before an obvious irruption occurs, and this seems to be more characteristic of plant invasions than of animal ones. The other appears to pertain more to animals than to plants, and was described as early as 1911 thus: "A newly introduced pest usually increases out of all proportion during the first few years of its existence in a new country, after which it gradually reverts to more normal conditions."[26] Caughley and Lawton[27] and Mead[28] describe similar patterns of an irruption followed by a decline to a post-invasion endemic state.

A typical new pest species invasion of New Zealand shows no obvious latent period, possibly because the true arrival time is often unknown, but invasions do tend to exhibit a boom-and-bust pattern of abundance. In this case, much of the post-irruptive decline is man-induced rather than natural, and hence is easier to explain; it is caused chiefly by an intensive period of research and the development of sustained control methods, such as resistant plants and biological control measures. Yet the decline may also contain a natural element if, for example, the invasion causes a reduction in the area of susceptible crop planted. In this event, there is likely to be a nonlinear impact on pest abundance, and local density within the remaining crop area would also decline as the total crop area declines.[29]

Considering some of these patterns in relation to lucerne pests, the weevil *Sitona discoideus* was first observed in New Zealand in 1974, where it is presumed to have arrived from the Mediterranean via Australia.[30] It causes yield reductions in lucerne, both through defoliation by adults and destruction of root hairs and root nodules by larvae.[30] The blue-green lucerne aphid became established in New Zealand at around the same time, in 1975, and rapidly became widespread.[31]

Figure 12.1a shows the densities of *Sitona discoideus* in Darfield, Canterbury, South Island, over time following its invasion in 1974. The initial populations are unknown (indicated by a dotted line) and may well have exceeded the observed peak of $74/m^2$. In this case the subsequent decline was initially precipitated by drought, but was later sustained by classical biological control from the parasitoid *Microctonus aethiopoides*, introduced in 1982.[29,32] Kean and Barlow[29] showed theoretically that the local density of *S. discoideus* is likely to be affected by the area of lucerne planted, so it is possible that the decline in lucerne growing may also have contributed to the decline in weevil abundance seen in Figure 12.1a. Note that the figure represents a time profile of local pest density, rather than global density summed over all available habitat.

The blue-green lucerne aphid shows a broadly similar pattern (Figure 12.1b), but in this case the figure represents suction-trap catches of flying aphids at Lincoln, South Island — distant from major lucerne-growing areas, and representative of global rather than local aphid densities (Figure 12.1a). In this case, the cause of the decline is less clear than for *S. discoideus*. Parasitoids of the genus *Aphidius* were introduced in 1977[31] but appear to have had minimal effect. Although it is suggested that generalist predators may have increased in abundance,[32a] their impact seems to be generally limited by asynchronous phenologies.[33] An alternative explanation is the decline in resource (lucerne) abundance, due to the combined effects of a reduction in total area grown and the introduction of aphid-resistant varieties, which comprised more than half the total plantings from the mid-'80s onward, based on Ministry of Agriculture and Fisheries Seed Certification Statistics. The total area of lucerne declined by about half between 1975 and 1983 (New Zealand Agricultural Statistics, 1976–84), partly because of disease and insect pests.[34] It is also possible that the high aphid densities in the first season of 1976–77 (Figure 12.1b) coincided with particularly favorable weather. Finally, part of the reason for the spectacular early declines is likely to be density-dependence in the aphids themselves, acting from one year to the next and possibly mediated through a decline in plant condition during spring, a season of high initial aphid densities.

12.4.2 Common wasp

The common wasp demonstrates the displacement of an existing species by an invading one, ecological impacts, and attempts at managing a well-established incursion.

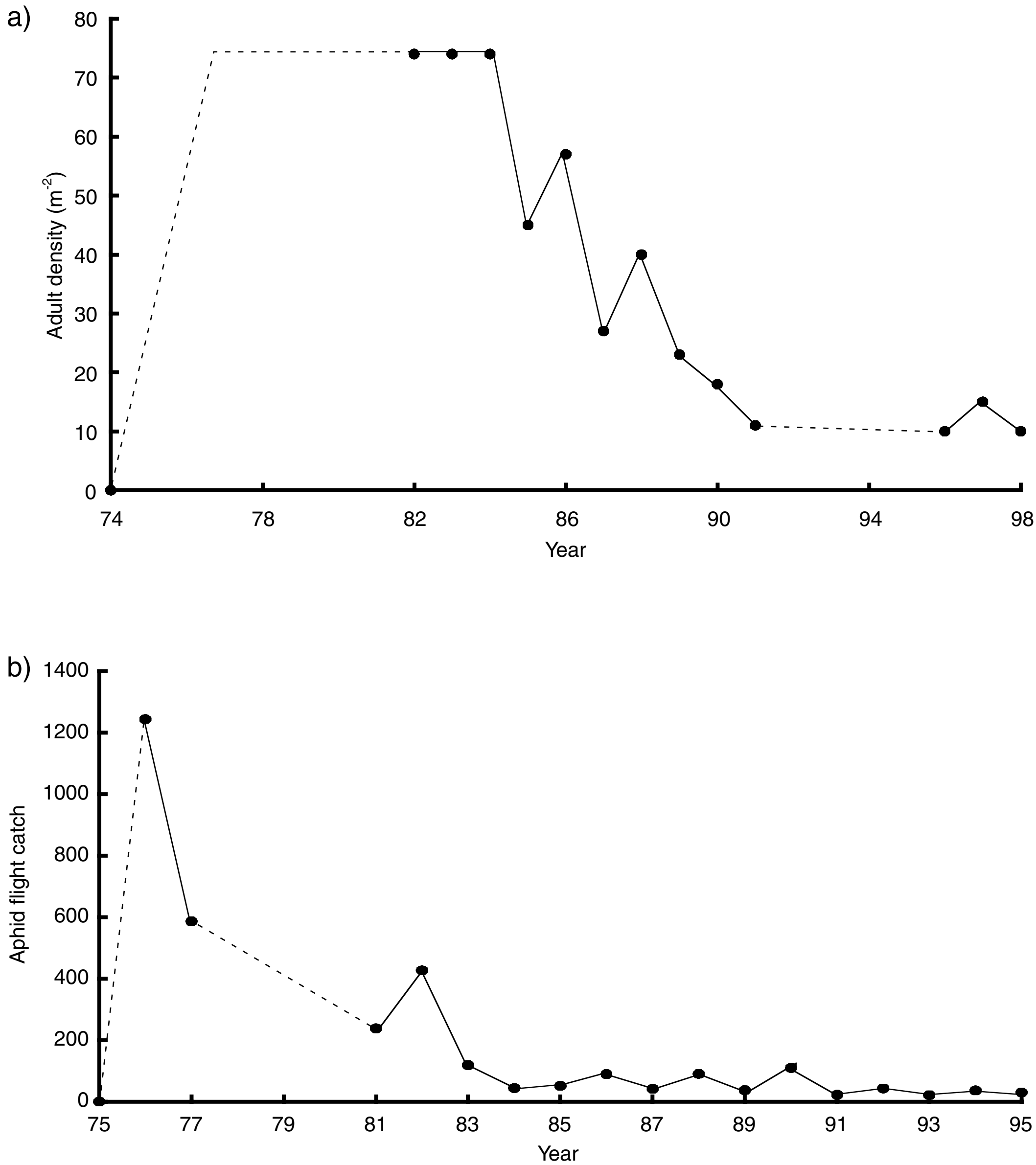

Figure 12.1 a) Local densities of *Sitona discoideus* at a site in Darfield, Canterbury, South Island, over time following its invasion in 1974 (dotted lines indicate interpolated but unknown densities). b) Regional densities over time of the blue-green lucerne aphid, measured as suction trap catches of flying adults at Lincoln, South Island, since its invasion in 1975/6 (dotted lines indicate interpolated but unknown densities).

12.4.2.1 Background

Social insects present a well-recognized invasive potential, and within New Zealand the impact of wasps is now evident and increasing. This may foreshadow an equal but as yet unrealized impact from the Argentine ant. The common wasp *Vespula vulgaris* is the most widespread and damaging of the four wasp species definitely established, which also include the German wasp *V. germanica* and two species of paper wasp, the Australian *Polistes humilis* and the Asian *P. chinensis*. The German wasp arrived in 1943, and the common wasp was recognized in Dunedin in 1983 but had probably been present in New Zealand since the 1970s. The Australian paper wasp was abundant in the northern North Island in the late 1880s, while the Asian paper wasp was first found in Auckland in 1979; since then it has spread throughout the northern half of the North Island after a failed attempt was made at eradication.[35] More recently, the yellow flower (or scoliid) wasp *Radumeris tasmaniensis* was found in the far north of the North Island in February 2000, and the government is commissioning a survey to establish the extent of its range and its host preferences.[36] This large species (30 to 40 mm long) is native to Australia and Papua New Guinea. All of these wasps attack native invertebrates, and the yellow flower wasp parasitizes the larvae of scarab beetles, of which New Zealand has a number of native species.

Moller[37] suggested that the following factors contribute to the success of social wasps as invasive species: the tendency of queens to seek hibernation refuges in places that may be associated with transported goods; the increase in global movement of such goods; the ability of queens to produce thousands of workers immediately in the absence of males (once fertilized in the autumn); a short generation time and a high reproductive rate, leading to rapid initial population increase; a high dispersive ability, even in the absence of human agency; phenotypic plasticity, for example the ability of German wasp colonies to overwinter in New Zealand, and the ability to succeed in a wide range of habitats; tolerance of extreme ranges in climate; a lack of native natural enemies; and a broad diet and opportunistic feeding behaviour.

Some of these apparent advantages may be more theoretical than real. First, although an overwintered immigrant queen has a very high reproductive potential and does not depend on mate-finding, in practice there appears to be a high probability of early nest failure,[37a] perhaps because of the combined energetic demands of nest-building, foraging, and raising the initial brood. It is estimated that 99% of overwintered queens fail to found new nests, at least among established wasp populations. The success rate may be higher in invading populations when there is little or no competition or aggression between queens in spring. Second, the rapid initial population increase still represents only one colony throughout the first year, which is potentially vulnerable to catastrophic failure by events such as flooding. Finally, although conditions in the nest render colonies largely independent of climate once the nest is established, with the exception of relatively rare catastrophes like flooding, climate may nevertheless exert an effect during the very early period of nest-building. Thus, Barlow, Beggs, and Barron[37b] show that spring rainfall has a significant negative effect on population increase of the common wasp in New Zealand honeydew beech forests, perhaps because the rain limits the wasps' foraging at this critical time. In spite of these factors that temper the ability of social wasps to invade and establish, the net effect is that in comparison with most other invertebrates they are extremely well equipped to do so.

12.4.2.2 The invasion

In terms of process, and the factors predisposing the success of invasions and an area's vulnerability to them, the displacement of German wasps by the common wasp in New Zealand can be considered as a model for displacement of a native species by an invader. By 1991 the German wasp was present over most of New Zealand, the common wasp was spreading, and there was evidence that the latter was replacing the former in some habitats but not in others. Indeed, in many habitats the German wasp was more abundant. Harris et al.[38] considered two key questions arising from these observations. First, what would eventually happen in habitats where there was no evidence of displacement of German wasps by common wasps, and why? Second, what is the mechanism for competitive displacement, where this appears to be occurring, that enables the common wasp to successfully invade in these environments?

Considering the first question, Harris et al.[38] suggested two future scenarios: eventual coexistence through habitat segregation, in which each species dominates in different habitats; or eventual replacement of German wasps everywhere through displacement by common wasps, which might occur at different rates in the different habitats. An argument against the second is the fact that in no other country is there evidence that one species can exclude the other from regions where there is a mixture of habitats.[38]

At the present time, both species are present over much of New Zealand, and Leathwick et al.[39] suggested that the common wasp is almost invariably more abundant. For example, in the Manawatu area in the southwest of the North Island, 74% of nests collected between 1992 and 1997 were of the common wasp,[39] but because no estimates of nest density were made prior to its arrival, it is not possible to tell whether the German wasp has been displaced. If competition was occurring, there was no evidence for it in terms of impaired growth or size of German wasp colonies where densities of the common wasp were high.[39] Thus the answer to the first question appears to be that there is evidence of a slow increase in the proportion of common wasps everywhere, but no information to suggest either that this will continue and lead to the eventual demise of the German wasp, or that there is active competition between the two species.

In terms of the second question, regarding competitive displacement, in one environment there is a classic instance of this via a mechanism that now appears to be understood. Populations of the common wasp attain some of their highest recorded densities in beech forests (*Nothofagus* spp.) on the South Island. These high densities, which on occasion exceed 30 nests per ha, are associated with a copious supply of honeydew secreted at the ends of waxy anal filaments or threads by the scale insects *Ultracoelostoma assimile* and *U. britanni* on trunks and branches of the beech trees.[40] The trees typically then become blackened with sooty mould. It is estimated that these "honeydew beech forests," where the scale insects are widespread, cover about 1 million ha of New Zealand, representing about 15% of the total forest cover.[41] In the northwest of the South Island, where honeydew beech forest is widespread, the common wasp appears to have competitively displaced the German wasp in an invasion front during 1986–87.[38,42] Moller et al.[42] suggested that the common wasp invasion front spread at a rate of 30 to 70 km/year, based on percentages of nests counted that were common wasps rather than German wasps. The progress of the front is shown in Figure 12.2, in terms of a 50% common wasp contour. This contour is necessarily approximate and is based on percentages of common wasp nests in the total nest populations in various locations (data from Moller et al.[42]). In these honeydew beech forests, there is evidence that the increased percentage of common wasp nests is accompanied by a decline in that of German wasp nests, to the extent that the latter species became extinct in many sites where it was formerly common.[42] The invasion of common wasps is graphically illustrated by the Waitahu site (Figure 12.3), where representation of this species in the wasp population increased from a trace in 1987 to 99% in 1989 (data from Moller et al.[42]).

So why have common wasps displaced German wasps in honeydew beech forests? Both species are generalist feeders,[38] but they appear to differ in their foraging habits for protein food, with the common wasp searching on shrubs and saplings and the German wasp in forest litter.[42] If anything, therefore, there appears to be a niche separation in terms of protein food. However, two other possibilities remain. One is direct interaction, in which queens of the common wasp usurp German wasp nests in spring,[43] although it would seem unlikely that they would be more successful at doing this than German wasp queens attacking common wasp nests, given the observations below. The second possibility is competition for honeydew, and there is some evidence for this. First, honeydew is in short

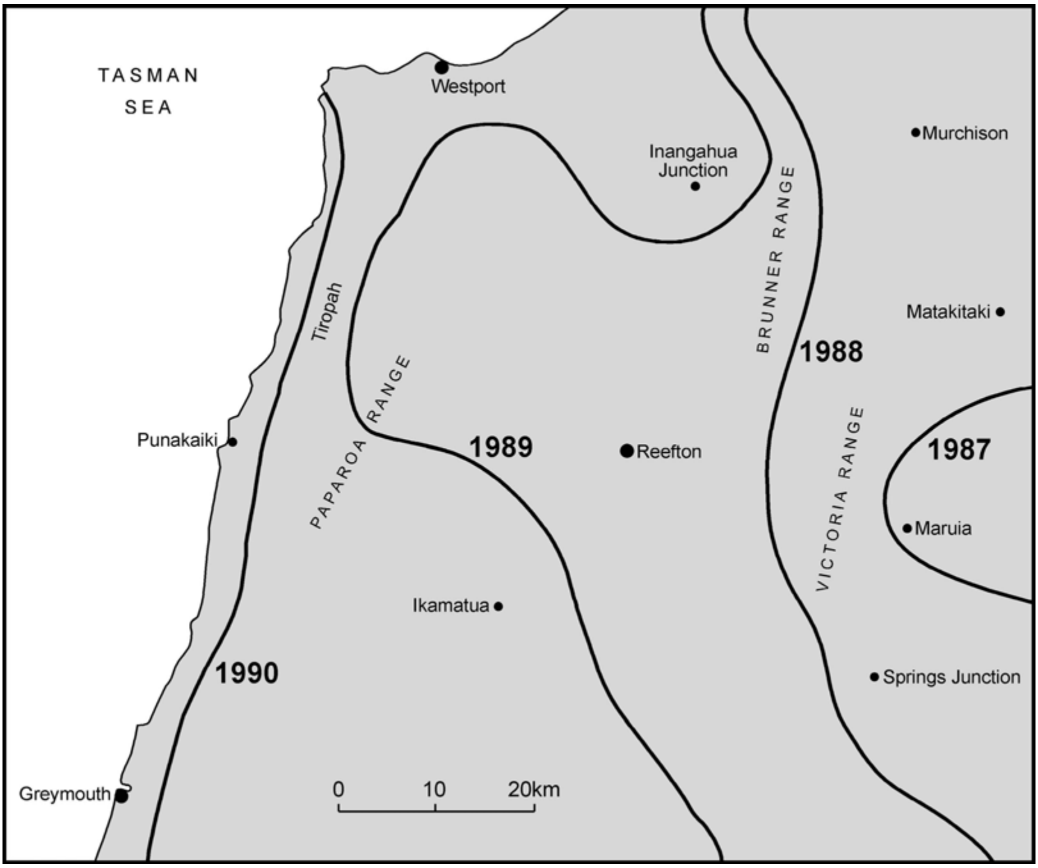

Figure 12.2 The invasion front and replacement front of German wasps by common wasps, on the west coast of New Zealand's South Island, measured as the approximate position of points at which 50% of nests sampled were of common wasps rather than German wasps.

supply in late summer, when wasp numbers reach their peak. Second, the common wasp is a more efficient forager than the German wasp.[38,43] Third, this advantage may allow the common wasp to produce queens that are of higher "quality" than those of the German wasp, and Archer[44] suggested a positive relationship between queen "quality" and the ability to successfully found colonies in spring. The latter concept is supported by evidence that smaller queens result when food is withheld from larvae, and that small queens are underrepresented among the queens that successfully found colonies in the spring.[45]

Harris et al.[38,43] offer some explanation for the superior foraging ability of common wasps over German wasps. When wasp numbers are high, the trunks of beech trees tend to harbor either species rather than both, suggesting that the more abundant species may be able to monopolize the honeydew resource.[38] Fourteen direct encounters observed between common and German wasps all resulted in victory for the German worker (cited in Harris et al.[38]), so the mechanism is clearly not direct antagonistic behavior. However, common wasps can remove honeydew at a faster rate than German wasps, and they spend less time feeding on fermenting honeydew on the tree trunks, which causes German wasps to become lethargic.[43] Thus the competitive displacement of German wasps by common wasps in honeydew beech forests appears to be due to the their superior foraging ability, while in other habitats throughout New Zealand the common wasp may be more abundant

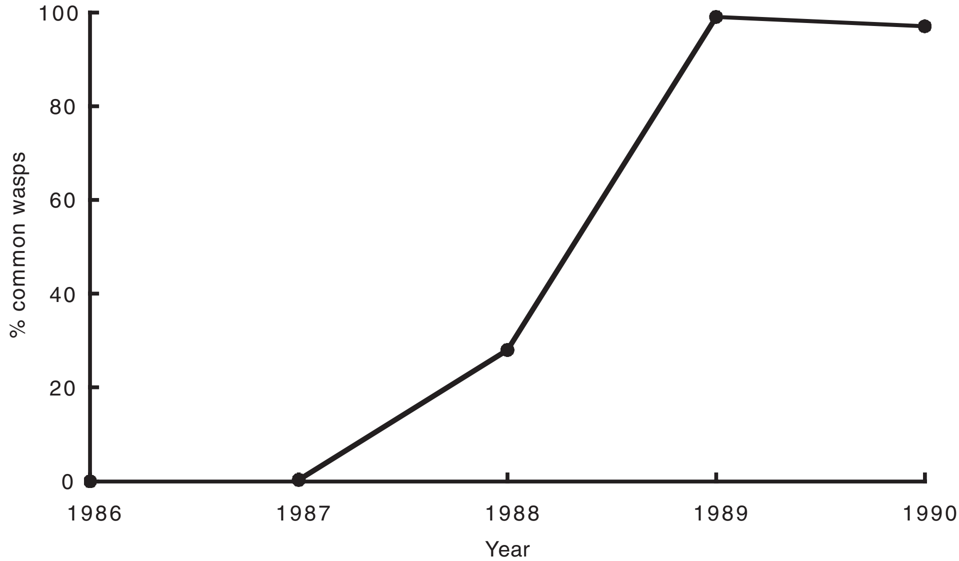

Figure 12.3 The replacement of German wasps by common wasps at a single site (Waitahu), indicated by the rapidly increasing proportion of nests sampled that were of common as opposed to German wasps.

than the German, but the two may be coexisting and there is no evidence of competitive displacement.

12.4.2.3 Ecological impacts

Beggs[41] estimated that honeydew beech forests, representing about 1 million ha of land, carried a wasp density of 10,000 workers per ha, or a biomass of 3.8 kg/ha at the time of peak numbers in summer. These high wasp populations have at least three key ecological effects. First, they compete with native birds and invertebrates for food. Second, they prey on native invertebrates, which affects their populations — and may also impact on insectivorous native birds, if the most vulnerable insect species to wasps are also the preferred food sources for birds. Third, by removing honeydew, wasps affect nutrient cycling and possibly the health of forest trees.

Wasps remove about 70% of the annual production of honeydew[41] and reduce the standing crop by 99% for about 4 months, and by 90% for a further 2 months.[46] Among the bird species affected are the kaka *Nestor meridionalis meridionalis*, a native parrot, and two honeyeaters, the *tui Prosthemadera novaeseelandia*e and the bellbird *Anthornis melanura*. When honeydew is available, kaka obtain most of their daily energy needs from it. However, it was calculated that it would be unprofitable energetically for the kaka to harvest the honeydew when the energy yield declines below 1.4 J per drop, and when wasps are present it is often near or below this value.[47] Field observations support this suggestion that wasps impact kaka feeding, since the birds were observed to avoid highly infested areas of beech forest until the honeydew reached an energy value of 4 J per drop.[47] Using the known rates of honeydew renewal, Beggs[41] estimated an approximate "ecological damage threshold" for wasps in terms of their effect on kaka behaviour. It corresponds to a frequency of wasp visits to honeydew threads of one every 400 min. Similarly, tui and bellbirds reduced their feeding on honeydew, or, in the case of tui, left beech forest reserves altogether, when the standing crop of honeydew fell below 2500 J/m^2, which corresponds to a cropping rate of no more than one drop every 180 min. Thus the ecological damage threshold above which the behavior of nectar-feeding birds is affected is a cropping rate of one honeydew drop every 180 to 400 min.[41] The extent to which these effects on behavior translate into impacts on survival or reproduction is unknown, and is confounded by the effects of introduced predators, such as stoats, *Mustela erminea*.[41]

The second major impact is predation on native invertebrates. Depending on their density, wasps remove up to 8.1 kg of invertebrates per ha per year.[48] It has been suggested that wasp population changes may be partly dictated by interaction with an invertebrate food source, since this is one of the few conceivable reasons for observed cycles in wasp abundance.[37b] However, wasps are largely generalist predators, so their impact on any one invertebrate species is independent of its abundance. Under these circumstances, the predation pressure can be maintained as the density of that species declines, although it is also possible that switching to other prey may occur under these circumstances. Among items that could be identified, the diet of wasps in honeydew beech forests consisted of 30% spiders, 20% caterpillars, 20% ants and bees, 15% flies, and 15% other.[48] It is estimated that the predation rate on some species, such as the caterpillars of the moth *Uresiphita polygonalis maorialis* and the orb-web spider *Eriphora pustulosa*, is so high that the probability of an individual surviving through the wasp season is virtually nil, suggesting that such populations may be seriously at risk.[41] Again, an approximate ecological damage threshold has been estimated at about 10 to 20% of average extant wasp densities, necessary for the conservation of such vulnerable native invertebrate species.[49]

The third impact is on nutrient cycling, through a reduction of carbon flow through honeydew to microorganisms in the soil. It is likely that the scale insects actually produce more honeydew in the presence of wasps, since the largely passive flow is stimulated by removal. In the absence of removal, the honeydew becomes more viscous, and the anal threads may become plugged.[41] Under these circumstances, the drain of carbon and nitrogen from the tree may well be exacerbated by wasp activity. As Beggs[41] points out, if New Zealand beech honeydew affects the metabolism of microorganisms, hence nutrient cycling, in the same way as aphid honeydew does in the Northern Hemisphere, then given the magnitude of removal (70% of annual production), it is highly likely that wasps negatively affect microorganisms around beech trees, and therefore possibly the trees themselves. At the same time, microorganisms in the vicinity of wasp nests may be positively affected. These complex interactions are particularly difficult to measure and quantify. In spite of its obvious magnitude, and largely because of funding constraints, the full ecological impact of wasps remains very poorly understood.

12.4.2.4 Management response

Considerable effort has been devoted to methods of wasp control, because of both the threat they pose to conservation and their impact on tourism. Protein-based poison baits can kill 80 to 100% of the wasp colonies within a site, but reinvasion is a major problem,[49] given that queens may fly as far as 70 km before establishing a nest.[42] Moreover, wasps do not respond to protein baits at some times of the year, particularly early in the season, before about January, possibly because of the abundance of prey.[41] It is not possible to use carbohydrate baits because of the risk of poisoning honeybees, and in any case these baits would have to compete with the abundant natural honeydew supply. Biological control has been attempted but so far has proved unsuccessful. A wasp parasitoid from Europe, *Sphecophaga vesparum vesparum*, was released in 1987–88 in many parts of New Zealand, but it established in only two sites. Both parasitoid and wasp populations have been monitored at one of these sites for 13 years, and there has been no detectable effect on the wasps, with parasitism levels averaging around 10% or less.[50,51] Earlier modeling of the interaction[52] suggested little chance of the parasitoid increasing its impact, partly based on a very low maximum yearly rate of increase of around 1.3% locally.[52] Two other strains of *S. vesparum* have also been released, but so far neither has established. The pathogenic fungus *Beauveria bassiana* is showing promise,[53] and work is in progress to incorporate it into a bait. It has the advantage over toxins that less would have to be taken into the nest.[41] There is also the potential to genetically modify wasp gut bacteria to make them pathogenic.[53a] The ideal control would probably be one that sterilizes queens, since the existing sterilized queens would compete with immigrant ones for nest sites, and this would reduce the reinvasion problem.

12.4.3 The clover root weevil

The clover root weevil case study exemplifies an early response to a recent and particularly serious economic threat.

12.4.3.1 Background

The genus *Sitona* comprises more than 100 species worldwide. The clover root weevil *Sitona lepidus* is distributed throughout Europe; it has been found in North Africa and occidental Asia; and it is now established in North America south of 50°N.[54] In Finland, *S. lepidus* has been found as far north as 66–67°N.

12.4.3.2 The invasion

S. lepidus was first recognized in New Zealand in March 1996.[55] The species is sufficiently similar in appearance to *Sitona discoideus*, however, for it to have been overlooked, probably for some years, and retrospective examination of stored samples indicated that the species had definitely been present since May 1994.[55] Effectively, *S. lepidus* was discovered by accident, because its phenology was inconsistent with that of *S. discoideus*; until then, specimens of *S. lepidus* recovered were presumed to be *S. discoideus*. Once recognized, *S. lepidus* was found to be distributed over more than 200,000 ha, suggesting that it had probably been in New Zealand for some time.[56] Such an establishment area precluded any chance of the weevil's eradication. Any ambiguity about the pest status of *S. lepidus* in New Zealand was quickly resolved. Soon after the weevil was discovered, it was recognised that *S. lepidus* larvae were reaching far higher densities than those found in Europe, and its damage effects were patently obvious. Willoughby and Addison[57] reported enormous peak populations of *S. lepidus* larvae as dense as 1400 larvae per m^2, which they calculated to represent 50 to 60 larvae per clover plant; this compared to the larval population peaks of 34 to 90 larvae per m^2 found in England.[58]

Not surprisingly, at the same time there was increasingly strident New Zealand grower reaction to the problem, and researchers embarked on a series of initiatives to try to raise funds for research, initially into the weevil's phenology and the patterns of damage. An early initiative was a series of public meetings held throughout the affected areas, plus two postal surveys, which were designed to assess district-wide patterns of damage and grower perceptions.[59] In another early initiative, Willoughby and Addison[57] assessed the rate of dispersal of *S. lepidus*, which they concluded was about 35 km/year. Thereafter, such surveys were discontinued, as costs rose geometrically as the radius of distribution increased.

Although most *Sitona* spp. tend to be univoltine,[60] and *S. lepidus* appears to be generally univoltine in Europe,[61] intensive population studies at representative sites showed that the weevil in New Zealand was bivoltine,[57] as has also been found by Mowat and Clawson[62] in Northern Ireland. The studies also confirmed the potential for *S. lepidus* to reach very high and destructive populations, although the 1400 larvae per m^2 initially observed by Willoughby and Addison[57] tended to be the exception rather than the rule. Indeed, Addison et al.[63] found that densities of 300/m^2 were more the norm. Probably the most important finding of this work was the effect of climate, and moisture in particular, on population densities. Dry conditions, as in spring 1996, led to restricted ovarian development and flights of the adults out of the pastures.[63] This in turn resulted in low numbers of larvae (50/m^2) and adults in the next generation during the southern summer of 1997. Conversely, wet spring conditions in 1997 led to a summer larval population in 1998 five to ten times greater.[63] This influence of spring moisture conditions was subsequently confirmed by Willoughby and Hardwick,[64] who examined the effect of irrigation on population buildup.

In general, wet spring conditions result in more severe summer damage and high numbers of adults and larvae entering the overwintering phase. Such high overwintering populations followed by further wet spring conditions lead to very damaging impacts. These findings have had important implications for the scheduling of irrigation.

12.4.3.3 Economic impact

While not an obligate feeder on white clover (*Trifolium repens*), *S. lepidus* strongly favors this legume.[65] Typical of *Sitona* spp., the adults leave crescent-shaped notches in the leaves, and the larvae feed on the root nodules and roots. The larvae are far more damaging, affecting both sustainability and productivity.[66,67] Initial attempts to understand the biology of *S. lepidus* were based on the international literature, but this tended to be vague, if not contradictory, in its declaration of the weevil's pest status. Researchers in Northern Ireland concluded that the weevil plays a significant role in reducing clover persistence[68]; elsewhere, statements about pest status in established pasture were more ambiguous,[58] although the destructive effects of the weevil in seedling establishment has long been widely recognized.[68] Mowat and Shakeel[69] concluded that the weevil is a pest of white clover in Ireland and England,[70] but in general, the species does not appear to be a major problem elsewhere.[71] In New Zealand, Gerard et al.[72] showed that the adult weevil has no impact on white clover seed production. However, given the enormous populations that can clearly occur in New Zealand, the threat to established clover pastures is very real.

Any threat to white clover has immediate and very serious implications for New Zealand's grazing industries because of its importance to those industries. For example, clover comprises 20 to 60% of the composition of New Zealand's intensive dairying pastures, and as such contributes an average of 210 kg of nitrogen per ha per year to such a system.[73] More broadly, Caradus et al.[74] have calculated that this represents 1.57 million tons of nitrogen fixation, with a value of more than $1.49 billion a year. Additionally, white clover–based fixation of atmospheric nitrogen usefully enhances the protein levels in the animal diet,[75] reducing the need for feed supplements. The alternative approach, using nitrogenous fertilizers, can cause eutrophication of waterways, nitrification of sources of drinking water, and adverse impacts on wildlife. While targeted use of such fertilizers can improve the situation, there remains the potential for farmer error or simply for inappropriate use. There has also been a major move away from using animal-based protein feed supplements in response to the appearance of spongiform encephalopathies. In response to this threat, the use of legumes is seen as natural and sustainable, and therefore very acceptable to the expanding organic market. Given that the economic cost of $300 million per year for clover root weevil, as estimated in Section 12.2.2 above, is based solely on the loss of nitrogen fixation, this figure is probably fairly realistic.

12.4.3.4 Management response

Soon after the weevil outbreak was discovered, a British entomologist with experience working with *Sitona lepidus* was invited to New Zealand. He immediately confirmed the notable differences in population behavior shown by the species in New Zealand compared with that observed in Europe, and he assisted in the design and development of an appropriate research program. Part of this comprised population dynamic studies, while the rest focused on three forms of control: short-term management strategies, for immediate application to affected farming systems; clover resistance and tolerance; and biological control systems, in the form of either classical agents or pathogens. It was recognized that the latter two objectives were of necessity medium- or long-term in their scope.

Short-term management. One of the useful features that arose from the series of farmers' meetings held shortly after *S. lepidus* had been discovered was the wide range of practical suggestions that were made to try to limit its impact. These included: examination of the value of cultivation and fallow periods in limiting the pest's impact; analysis of the rates of reinvasion by the weevil after its removal; the testing of insecticidal exclusion barriers around the perimeter of fields; methods of clover reestablishment through overdrilling; the value of removing clover from pastures using selective herbicides; the usefulness of coating clover seed with pesticides to expedite seedling reestablishment; and approaches to slow down the weevil's spread throughout New Zealand. Regarding the latter, one obvious question related to how long adult weevils are able to survive in hay bales.

While much of this work was investigated, few of the results have been published; rather, the information has been promulgated through technology transfer processes such as field days. In short, reinvasion was found to be very rapid after cultivation, as the weevils seem to be particularly attracted to seedling clover plants, and barriers in no way precluded reinvasion.

Clover resistance and tolerance. A medium- to long-term program was established within 6 months after the arrival of the clover root weevil to examine the effect of soil type and grazing intensity on damage. Work was also established to seek a genetic basis for clover resistance or tolerance to weevil attack. All of these field-based approaches were confounded by the impact of weather, which resulted in highly variable weevil numbers. In view of this, the work became more protracted than expected.

A broad screening program for resistance was developed to seek white clover plant tolerance and resistance to the weevil.[59] This involved the replicated planting of 10,000 lines of white clover germplasm, from which 250 lines were selected and seedlings planted in weevil-infested pastures. The plants were then field-tested for 3 years under conventional farming practices.[59] Initial results have shown some indication of genetic resistance or tolerance to the weevil, but the findings are as yet incomplete and are likely to be commercially sensitive.

Biological control. Based on success in suppressing the weevils *S. discoideus* in lucerne with *Microctonus aethiopoides*[32] and *Listronotus bonariensis* in ryegrass pasture with *Microctonus hyperodae*,[76] there was good reason to pursue suitable biological control agents for *S. lepidus*, including searches for classical control agents (e.g., parasitoids) and useful pathogens.

Initial screening of New Zealand's existing populations of *Microctonus* spp. demonstrated no potential whatsoever for the control of clover root weevil.[71] Consequently, in 1998 a search for suitable agents was begun in Europe and the United States. In the United States no parasitoids were identified, although a putative disease of the female reproductive tract was noted.[77] Conversely, in Europe very active ecotypes of *M. aethiopoides* were identified in Germany and Switzerland. In addition, *Allurus* spp. (Hymenoptera) (C.B. Phillips, unpublished data), *Perlitus sp.* (Hymenoptera), and *Microsoma exiguum* (Diptera) were noted as potentially useful parasitoids. The initial phases of this search employed an agent to search on behalf of the program; this established the value of searching for parasitoids in Europe. Thereafter, as the work became more complex, it was found to be cost-effective to send New Zealand staff to work in Europe. This approach was taken in conjunction with the USDA European Biological Control Laboratory in Montpellier, France, the Institute of Grassland and Environmental Research at North Wyke, England, and a number of collaborating European groups. As a result, understanding of the distribution of *M. aethiopoides* ecotypes was greatly enhanced, with parasitoids being collected from four zones in France and one zone each of Spain, Switzerland, Germany, Romania, Finland, Norway, Sweden, England, Wales, Scotland, and Ireland. The collection of such a wide range of biological material is highly consistent with the view that ecotypes can play a major part in the success of biological control programs.[76] Additionally, New Zealand staff were able to research the ecology and phenology of the potential control agents more effectively. Recently, much of the material recovered has been remitted to New Zealand quarantine, where it is now being examined further for its potential to control *S. lepidus*.

The widespread search for control agents also includes screening of potential pathogens to complement material tested in New Zealand. Species under consideration include *Beauveria bassiana, Bacillus thuringiensis*, a species of *Enterobacter*, and two species of *Serratia*.[59] Connected to this work is the opportunity to search for aggregation pheromones in *S. lepidus*, such as those found in *S. lineatus*,[78,79] and incorporate them into an "attract-and-infect" technology.

12.5 Conclusions

Experience shows that invasive invertebrates pose a major threat to New Zealand, and the threat is becoming well recognized by the appropriate government agencies, research funding bodies, and the public at large. The invasions of wasp species occurred sufficiently long ago that they offer little in terms of lessons for safeguarding against future incursions. On the other hand, the clover root weevil invasion occurred at a time of relative enlightenment about the threats of invading species and the pathways of invasion, so it is worth considering some lessons from this incursion.

From the perspective of protecting New Zealand's clover production, work on the weevil was begun belatedly, because the pest was simply not recognized. More precise interception systems and more widespread monitoring systems would have at least increased the chances of eradicating the weevil, although the species' soil-dwelling larval stages, and the presence of all stages throughout the year, would still have made this very difficult. The overseas literature was of very limited value, as the phenology and damage potential of *S. lepidus* in New Zealand proved to be significantly different from what had been observed elsewhere. Had decisions been made entirely from the literature, quite erroneous approaches might well have been undertaken.

Developing any kind of research program was difficult. Very little was known regarding the extent of the problem, so care had to be taken not to exaggerate the importance of the pest to both growers and funding bodies. The former looked to scientists for reassurance, or at least information (neither of which were really available), while the latter were wary of scientists overselling the threat in order to obtain new funding. Some resolution was found through a district-wide "road-show," in which the scientists were able to explain honestly the limited extent of the understanding of the problem, while the growers were able to assist in making a case to the government for additional funding, as well as providing their own contribution. Thanks to an established dialogue with the growers, the development of a mutually acceptable work program was relatively easy. Regular meetings thereafter have proved very helpful, particularly in defining research approaches to the short-term alleviation of impacts. In effect, it took four seasons to properly understand the phenology of the weevil, and this point needed to be made repeatedly during early consultations with the growers. Understanding the phenology immediately indicated management options around the scheduling of irrigation.

The visit by a British entomologist was valuable in confirming the interpretations made by entomologists in New Zealand, and it provided a useful basis for subsequent collaborative research. The very rapid implementation of a delimiting survey of the clover root weevil was useful in that it precluded further debate about the merits of trying to eradicate the pest. Conversely, subsequent surveys of the weevil's distribution became progressively more expensive and were not cost-effective. Nonetheless, targeted surveys to measure extremes of the distribution are likely to be valuable.

In terms of biological control agents (both classical agents and pathogens), events showed the wisdom of eventually setting up research capability in the primitive habitat of the weevil. Collaboration with other agencies throughout Europe has helped enormously, and it highlights the need for international reciprocity and goodwill, not only in the pursuit of biological control agents, but also in the sharing of information and the development of databases on key invasive species. In this respect, the clover root weevil offers a salutary lesson about the apparent uniqueness of species impacts, and about the importance of context. The hope must be that the uniqueness is more apparent than real, and that there are, in fact, "rules" that govern invading species, and all we have to do is find them. Perhaps, for example, the potential pest status of the clover root weevil in New Zealand was obvious, on the simple basis of abundance of the clover resource.

Acknowledgments

We are grateful to David Teulon, Craig Phillips, and Marlon Stufkens for access to unpublished data, to Gary Barker, Travis Glare, Dave Leathwick, and Richard Watson for helpful discussion, and to George Gill and the Ministry of Agriculture and Forestry for information on species interceptions. We also thank Craig Phillips for helpful comments on the manuscript. Funding was provided by the Foundation for Research, Science and Technology.

References

1. Ministry for the Environment, *The State of the New Zealand Environment* Report, 1997.
2. Ministry of Agriculture and Forestry, Terms of reference for the development of a biosecurity strategy for New Zealand, *Biosecurity,* MAF, Wellington, New Zealand, 24, 15, 2000.
3. Gibbs, G.W., Butterfly blues, *Forest and Bird,* 18, 18, 1987.
4. Gibbs, G.W., New Zealand Butterflies: Identification and Natural History, Collins, Auckland, 1980.
5. Bertram, G., The impact of introduced pests on the New Zealand Economy, in *Pests and Weeds: A Blueprint for Action*, Hackwell, K. and Bertram, G., Eds., New Zealand Conservation Authority. Wellington, New Zealand, 45, 1999.
6. Prestidge, R.A., Barker, G.M., and Pottinger, R.P., The economic cost of Argentine stem weevil in pastures in New Zealand, in *Proc. 44th NZ Weed and Pest Control Conf.*, 165, 1991.
7. Goldson, S.L., Proffitt, J. R., and Baird, D.B., Establishment and phenology of the parasitoid *Microctonus hyperodae* Loan in New Zealand, *Env. Entomol.,* 27, 1386, 1998.
8. *New Zealand Official Yearbook*, Statistics NZ, David Bateman Ltd, Auckland NZ, 2000
9. Vlassoff, A. and McKenna, P.B., Nematode parasites of economic importance in sheep in New Zealand, *NZ J. Zool.,* 21, 1, 1994.
10. Heath, A.C. G., Ectoparasites of livestock in New Zealand, *NZ J. Zool.,* 21, 23, 1994.
11. Bisset, S.A., Helminth parasites of economic importance in cattle in New Zealand, *NZ J. Zool.*, 21, 9, 1994.

12. Brunsdon, R.V., The economic impact of nematode infection n sheep: implications for future research and control, in *The Economic Importance of Parasites of Livestock in New Zealand*, Heath, A.C. G., Ed., NZ Soc. Parasitology Misc. Pub. 1, 4, 1988.
13. Carter, P.C. S., Risk assessment and pest detection surveys for exotic pests and diseases which threaten commercial forestry in New Zealand, *NZ J. For. Sci.*, 19, 353, 1989.
13a. Gill, G., personal communication.
14. Barker, G.M., Naturalised terrestrial molluscs in New Zealand: origins and establishment, in *Proc. 41st Ann. Conf. NZ Ent. Soc.*, 1992.
14a. Watson, R.N., personal communication.
15. Willoughby, B.E. and Hardwick, S., Tropical grass webworm (*Herpetogramma licarsisalis*) (Walker)(Lepidoptera: Pyralidae): a short-term establishment in Northland, New Zealand? in *Proc. 7th Australasian Conf. Grassland. Invert. Ecol.*, Matthiessen, J. N., Ed., CSIRO, Wembley, WA, Australia, 52, 1999.
16. Hardwick, S. and Davis, L.T., Field control of *Herpetogramma licarsisalis* (Walker) in northern Northland pastures, *NZ Plant Protection*, 53, 360, 2000.
17. Parliamentary Commissioner for the Environment, *New Zealand Under Siege: A Review of the Management of Biosecurity Risks to the Environment*, Office of the Parliamentary Commissioner for the Environment, Wellington, New Zealand, 2000.
18. Moeed, A. and Meads, M.J., Seasonality of pitfall trapped invertebrates in three types of native forest, Orongorongo Valley, New Zealand. *NZ J. Zool.*, 12, 17, 1985.
19. Ramsay, G.W., et al., *Research on Terrestrial Insects of New Zealand,* Wildlife Research Liaison Group Review No. 10., 1988.
20. Goldson, S.L. and Frampton, E.R., Managing incursions: New Zealand case studies and analysis, in *Plant Health in the New Global Trading Environment: Managing Exotic Insects, Weeds and Pathogens*, McRae, C.F. and Dempsey, S.M., Eds., Department of Agriculture, Fisheries and Forestry, Australia, 143, 1999.
21. Frampton, R., Synthetic pheromone to help in painted apple moth programme, *Biosecurity*, MAF, Wellington, New Zealand, 24, 19, 2000.
22. Ministry of Health, *Exclusion and Control of Exotic Mosquitoes of Public Health Significance*. Report to the Minister of Biosecurity, Ministry of Health, Wellington, New Zealand, 1997.
23. Oldroyd, B.P., Coevolution while you wait: *Varroa jacobsoni*, a new parasite of western honeybees, *TREE*, 14, 312, 1999.
24. Belton, D., How the varroa management programme works, *Biosecurity,* MAF, Wellington, New Zealand, 21, 5, 2000.
25. Bolger, P., Varroa update, *Biosecurity*, MAF, Wellington, New Zealand, 24, 12, 2000.
26. Green, E.E., The wanderings of a gigantic African snail, *Zoologist* (London), 15, 41, 1911.
27. Caughley, G. and Lawton, J. H., Plant-herbivore systems, in *Theoretical Ecology,* May, R.M., Ed., Blackwell Scientific Publications, Oxford, UK, 132, 1981.
28. Mead, A.R., *The Giant African Snail: A Problem in Economic Malacology,* University of Chicago Press, Chicago, USA, 1961.
29. Kean, J. M. and Barlow, N.D., A spatial model for the successful biological control of *Sitona discoideus* by *Microctonus aethiopoides. J. Appl. Ecol.*, 38, 162, 2001.
30. Goldson, S.L., et al., The seasonal biology of *Sitona discoideus* Gyllenhal (Coleoptera: Curculionidae), an introduced pest of New Zealand lucerne. *Bull. Ent. Res*., 74, 249, 1984.
31. Kain, W.M. and Trought, T.E., Insect pests of lucerne in New Zealand, in *Lucerne for the Eighties: Agronomy Society of New Zealand Special Publication 1*, Wynn-Williams, R.B., Ed., 49, 1982.
32. Goldson, S.L., Proffitt, J. R., and McNeill, M.R., Seasonal biology and ecology in New Zealand of *Microctonus aethiopoides* (Hymenoptera: Braconidae), a parasitoid of *Sitona* spp. (Coleoptera: Curculionidae) with special emphasis on atypical behaviour, *J. Appl. Ecol.*, 27, 703, 1990.
32a. Muscroft-Taylor, K., personal communication.
33. Leathwick, D.M. and Winterbourn, M.J., Arthropod predation on aphids in a lucerne crop, *NZ Entomologist*, 8, 75, 1984.

34. Dunbier, M.W., Wynn-Williams, R.B., and Burnett, P.A., Lucerne in the 70s, in *Lucerne for the Eighties: Agronomy Society of New Zealand Special Publication 1*, Wynn-Williams, R.B., Ed., 3, 1982.
35. Clapperton, B.K., Moller, H., and Sandlant, G.R., Distribution of social wasps (Hymenoptera: Vespidae) in New Zealand in 1987, *NZ J. Zool*., 16, 315, 1989.
36. Pascoe, A., Yellow flower (scoliid) wasp: an unwelcome visitor?, *Biosecurity,* MAF, Wellington, New Zealand, 25, 10, 2001.
37. Moller, H., Foreword, *NZ J. Zool.,* 18, i, 1991.
37a. Leathwick, D.M., personal communication.
37b. Barlow, N.D., Beggs, J.R., and Barron, M.C., Dynamics of common wasps in New Zealand beech forests: a model with density-dependence and weather. *J. Anim. Ecol*., in press.
38. Harris, R.J., Thomas, C.D., and Moller, H., The influence of habitat use and foraging on the replacement of one introduced wasp species by another in New Zealand, *Ecol. Entomol.,* 16, 441, 1991.
39. Leathwick, D.M., et al., Comparative growth and seasonality of *Vespula germanica* (F.) and *Vespula vulgaris* (L.)(Hymenoptera: Vespidae) colonies in the Manawatu region of New Zealand, *NZ J. Zool*., 26, 27, 1999.
40. Gaze, P.D. and Clout, M.N., Honeydew and its importance to birds in beech forests of South Island, New Zealand. *NZ J. Ecol.,* 6, 33, 1983.
41. Beggs, J. The ecological consequences of social wasps (*Vespula* spp.) invading an ecosystem that has an abundant carbohydrate resource, *Biol. Conserv.,* 99, 17, 2001.
42. Moller, H., et al., *Ecology and Control of Wasp Populations in New Zealand*. Unpublished contract report to Department of Conservation, Wellington, New Zealand, 1990.
43. Harris, R.J., Moller, H., and Winterbourn, M.J., Competition for honeydew between two social wasps in South Island beech forests, New Zealand, *Ins. Soc.*, 41, 379, 1994.
44. Archer, M.A., Population dynamics of the social wasps *Vespula vulgaris* and *Vespula germanica* in England. *J. Anim. Ecol.,* 54, 473, 1985.
45. Moller, H., et al., Effect of introduced social wasps on the standing crop of honeydew in New Zealand beech forests. *NZ J. Zool*., 18, 171, 1991.
46. Beggs, J. R. and Wilson, P.R., The kaka, *Nestor meridionalis*, a New Zealand parrot endangered by introduced wasps and mammals. *Biol. Cons*., 56, 23, 1991.
47. Harris, R.J., and Beggs, J. R., Variation in the quality of *Vespula vulgaris* (L.) queens (Hymenoptera: Vespidae) and its significance in wasp population dynamics. *NZ J. Zool*., 22, 131, 1995.
48. Harris, R.J., Diet of the wasps *Vespula vulgaris* and *V. germanica* in honeydew beech forest of the South Island, New Zealand. *NZ J. Zool*., 18, 159, 1991.
49. Beggs, J. R., et al., The difficulty of reducing introduced wasp (*Vespula vulgaris*) populations for conservation gains. *NZ J. Ecol*., 22, 55, 1998.
50. Beggs, J. R., Harris, R.J., and Read, P.E. C., Invasion success of the wasp parasitoid S*phecophaga vesparum vesparum* (Curtis) in New Zealand. *NZ J. Zool*., 23, 1, 1996.
51. Barlow, N.D., Beggs, J. R., and Moller, H., Spread of the wasp parasitoid Sphecophaga vesparum vesparum following its release in New Zealand. *NZ J. Ecol*., 22, 205, 1998.
52. Barlow, N.D., Moller, H., and Beggs, J. R., A model for the effect of *Sphecophaga vesparum vesparum* as a biological control agent of the common wasp in New Zealand. *J. Appl. Ecol*., 33, 31, 1996.
53. Harris, R.J., et al., Susceptibility of *Vespula vulgaris* and *V. germanica* (Hymenoptera: Vespidae) to generalist entomopathogenic fungi and their potential for wasp control. *J. Invert. Path*., in press, 2001.
53a. Glare, T., personal communication.
54. Bright, D.E., Revision of the genus Sitona (Coleoptera: Curculionidae) of North America. *Ann. Ent. Soc. Am*., 87, 277, 1994.
55. Barratt, B.I. P., Barker, G.M., and Addison, P.J., *Sitona lepidus* (Coleoptera: Curculionidae), a potential clover pest new to New Zealand, *NZ Entomologist*, 19, 23, 1996.
56. Barker, G.M., et al., *Sitona lepidus* Gyllenhal newly established in New Zealand: assessment of distribution in the North Island, *Proc. 49th NZ Plant Prot. Conf*., 266, 1996.

57. Willoughby, B. and Addison, P., Clover root weevil (*Sitona lepidus*) – a threat to sustainability of white clover in New Zealand pastures?, *Proc. NZ Grassland Assoc.*,59, 23, 1997.
58. Murray, P.J. and Clements, R.O., Distribution of three species of *Sitona* weevils in grassland England, *Ann. Appl. Biol.*, 127, 229, 1995.
59. Willoughby, B.E., et al., Clover root weevil (*Sitona lepidus*): the New Zealand response to a major biosecurity breach, *Proc. 7th Australasian Conf. Grassland. Invert. Ecol.*, Matthiessen, J. N., Ed., CSIRO, Wembley, WA, Australia, 35, 1999.
60. Moulden, J., The biology of Sitona species with particular reference to S. humeralis, Report, Agronomy Branch, Department of Agriculture, South Australia no. 44, 11pp, 1973.
61. Gerard, P.J., et al., Establishment of the invader: insights into life history and biology of *Sitona lepidus* in the Waikato region of New Zealand, *Proc. 7th Australasian Conf. Grassland. Invert. Ecol.*, Matthiessen, J. N., Ed., CSIRO, Wembley, WA, Australia, 43, 1999.
62. Mowat, D.J. and Clawson, S., Oviposition and hatching of the clover root weevil *Sitona lepidus* Gyll. (Coleoptera: Curculionidae), *Grass and Forage Sci.*, 51, 418, 1996.
63. Addison, P.J., et al., Clover root weevil: observations on differences between 1997 and 1998 summer populations in the Waikato, *Proc. 51st NZ Plant Prot. Conf.*, 1, 1998.
64. Willoughby, B.E. and Hardwick, S., Effect of summer irrigation on population dynamics and reproductive status of adult *Sitona lepidus* in the Waikato, *Proc. 52nd NZ Plant Prot. Conf.*, 245, 1999.
65. Murray, P.J. and Clements, R. O, Investigations of the host feeding preferences of Sitona species commonly found on white clover (*Trifolium repens*) in the UK, *Ent. Exp. Appl.*, 71, 73, 1994.
66. Mowat, D.J. and Shakeel, M.A., The effect of some invertebrate species on persistence of white clover in ryegrass swards, *Grass and Forage Sci.*, 44, 117–124, 1989.
67. Care, D.A., et al., Interaction between clover root weevil and clover root type, *NZ Plant Prot.*, 53, 420, 2000.
68. Murray, P.J. and Clements, R.O., Studies on the feeding of *Sitona lineatus* L. (Coleoptera: Curculionidae) on white clover (*Trifolium repens* L.) seedlings, *Ann. Appl. Biol.*,121, 233, 1992.
69. Mowat, D.J. and Shakeel, M.A., The effect of pesticide application on the establishment of white clover in a newly-sown ryegrass/white clover sward, *Grass and Forage Sci.*, 43, 371, 1988.
70. Clements, R.O. and Murray, P.J., Pest and disease damage to white clover at widespread sites in England and Wales, *Proc. Brit. Grassland Soc. Symp. Strategies for Weed, Disease and Pest Control in Grassland*, 10.21, 1991.
71. Barratt, B.I. P., Evans, A.A., and Ferguson, C.M., Potential for biocontrol of *Sitona lepidus* by *Microctonus* spp., *Proc. 50th NZ Plant Prot. Conference*, 37, 1997.
72. Gerard, P.J., Eden, T., and Tozer, W., Preliminary investigation into impact of *Sitona lepidus* adults on white clover seed production, *Proc. 52nd NZ Plant Prot. Conf.*, 250, 1999.
73. Ledgard, S.F., et al., Nitrogen losses from grazed dairy pasture, as affected by nitrogen fertiliser, *Proc. NZ Grasslands Assoc.*, 57, 21, 1996.
74. Caradus, J. R., Woodfield, D.R., and Stewart, A.V., White clover: New Zealand's competitive edge, in *Proceedings of a Joint Symposium, Lincoln University, New Zealand, 21–22 November, 1995*, Woodfield, D.R., Ed., Agronomy Society of New Zealand Special Publication No. 11, Grassland Research and Practice Series No. 6, 1,1996.
75. Bax, J. A. and Schills, R.L. M., Animal responses to white clover, FAO/REUR Technical Series 29, White clover in Europe: State of the Art, 7, 1993.
76. Phillips, C.B., et al., Progress in the search for biological control agents of the clover root weevil, *Sitona lepidus* (Coleoptera: Curculionidae), *NZ J. Agric. Res.*, 43, 541, 2000.
77. Blight, M.M., et al., Male-produced aggregation pheromone in *Sitona lineatus*. *Naturwissenschaften*, 71, 1984.
78. Blight, M.M. and Wadhams, L.J., Male-produced aggregation pheromone in the pea and bean weevil, *Sitona lineatus*, *J. Chem. Ecol.*, 13, 733, 1987.

chapter thirteen

The impact of exotic insects in New Zealand

Angus Cook, Philip Weinstein, and Alistair Woodward

Contents

13.1 Introduction

For much of its history, New Zealand has been relatively free of insect incursions that could threaten its vulnerable ecosystems. Isolation and a temperate climate have limited the establishment of many pests that are prevalent in other countries. However, in more recent times this relative refractivity to exotic insects (and arachnids, for the purposes of this chapter) has been compromised, primarily as a result of human activities.

0-8493-0836-4/02/$0.00+$1.50

In particular, the process of pastoral transformation (the establishment of pastureland) over the last two centuries has provided opportunities for many unwanted pests to enter the country and become established. More recently, the global mass movement of passengers and cargo has placed the country at further risk of insect and arachnid invasions. This chapter will explore New Zealand's transition from "virgin soil" status to participant in the global exchange of organisms. The impacts of these incursions will be explored, and the associated costs of such events will be illustrated using three recent case studies.

13.2 Historical and current perspectives

13.2.1 The New Zealand ecosystem prior to human settlement

Following its separation from the ancient supercontinent of Gondwana, New Zealand embarked on an 80-million-year period of geographical isolation. The islands remained remote from the major changes in species composition that were occurring in Eurasia, Africa, and the Americas. With minimal opportunity for reciprocation of continental organisms, even mammals failed to become established, excepting two species of bat. The large distances required for migration and ocean surroundings prevented incursions of insects, even from "near" neighbors such as Australia (Figure 13.1). New Zealand was therefore left with a depauperate fauna, and the only potential human or crop pests present were those that evolved *insight*. Before human colonization around 1000 years ago, there was thus a virtual absence of insects that would be considered pests today. As will be discussed, New Zealand's isolation has become increasingly compromised since the arrival of humans and the vast range of new plants and animals that accompanied them (Figure 13.2). A number of insects also took advantage of these migrations, and were successful stowaways on the boats (and later aircraft) that landed in New Zealand.[1]

The transfer of exotic insect species from their former habitats has therefore occurred via "long-distance jumps."[2] Compared with gradual insect migrations or movements with geological change, these species transfers occurred rapidly. Once the natural barriers had been surmounted, the country found itself exposed to the fauna of far-distant locations.

13.2.2 The introduction of insects with Polynesian migration

A range of insects accompanied the migration of Polynesians across the Pacific. However, a major factor that limited insect spread was the great distances separating islands in the Pacific, and the long travel times necessary to bridge the expanses of ocean. As with other pests, some filtering out of stowaway species occurred during the process of movement across the scattered islands of the Pacific. By current standards, passenger numbers were small, and travel irregular and infrequent. Despite the existence of trade routes between some island groups, the goods transported tended to be of low volume. Thus, despite centuries of human migration, there was limited transmission of exotic insects from the Asian mainland to the Pacific island groups, particularly those which were most remote. New Zealand, one of the last island groups to be colonized, was therefore spared from large-scale invasion, even with Polynesian arrival. In addition, the temperate climate may have limited the establishment of any tropical species that accompanied the early navigators.

However, a few hardy species did appear to survive this often interrupted, centuries-long progress from Asia to New Zealand. It is believed that these included fleas, lice, and scabies mites, which could endure the long distances and cramped conditions of canoe voyages by settling upon human and animal hosts (including a breed of rat, *kiore*, and dog, *kuri*, from Polynesia). It is possible that plant pests may also have been carried with

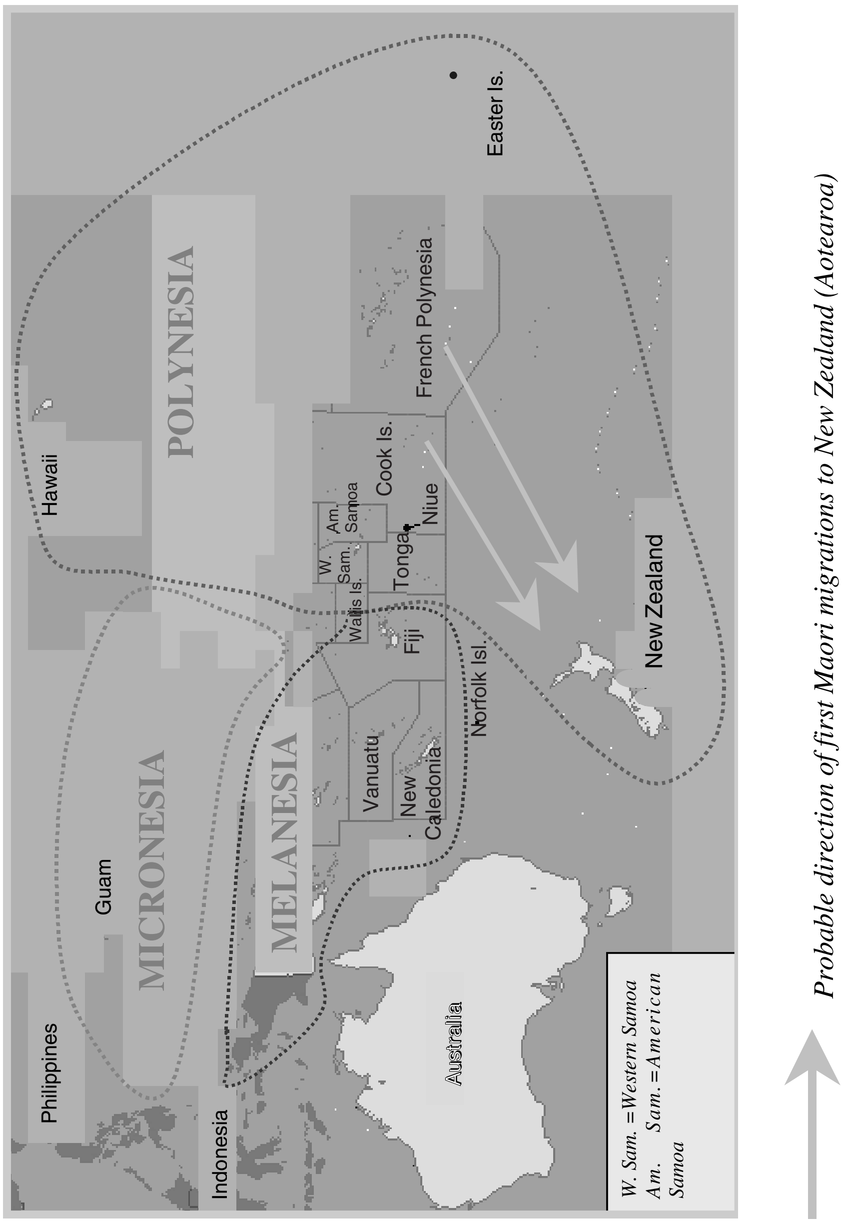

Figure 13.1 New Zealand and the South Pacific.

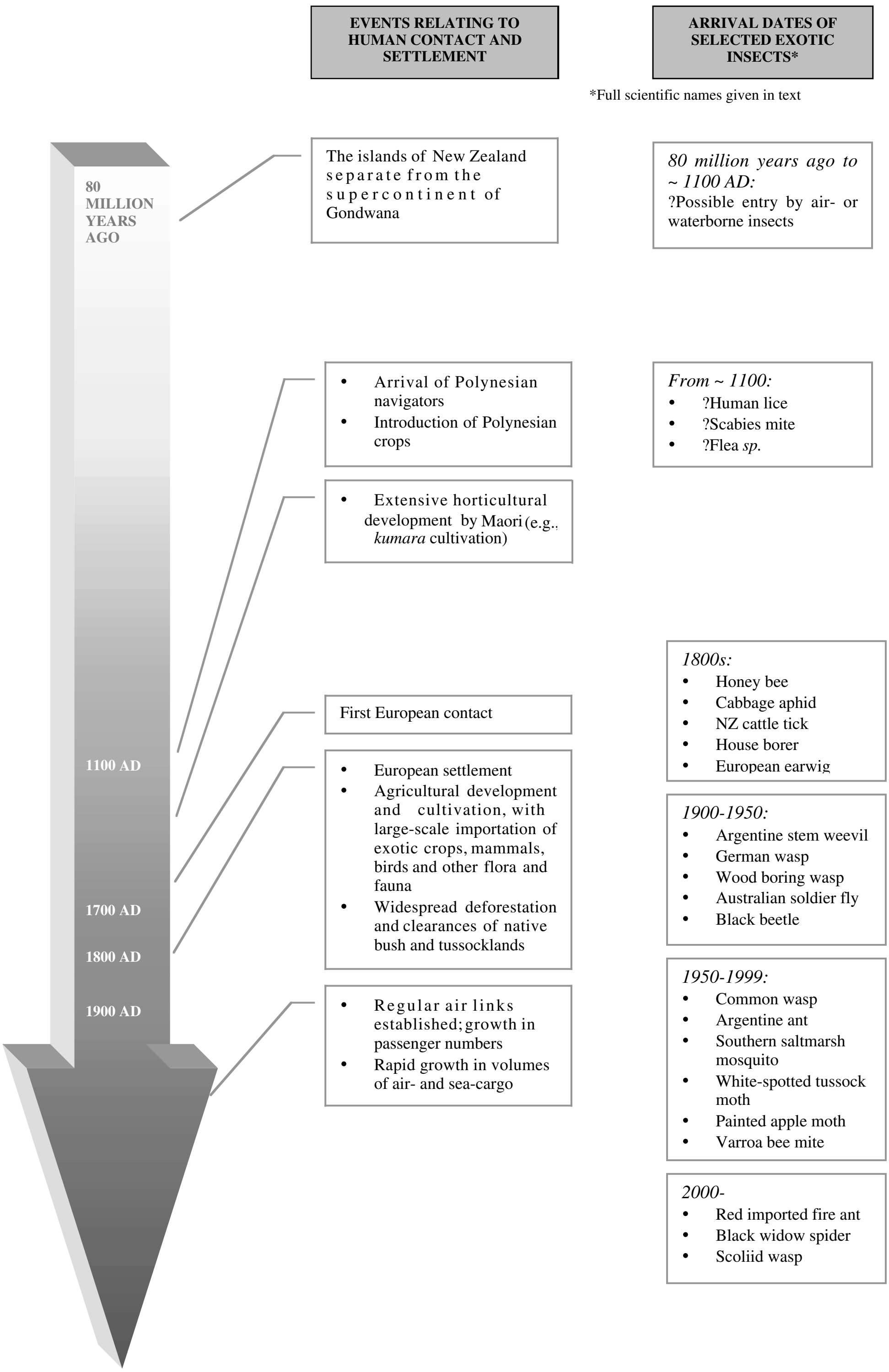

Figure 13.2 Chronology of human and exotic insect arrivals in New Zealand.

the crops the Maori bought with them, such as *kumara*, the New Zealand sweet potato. However, any experiments at transplanting and establishing certain tropical plants, and their associated pests, may have failed in New Zealand's temperate climate.

13.2.3 The consequences of European contact and settlement

After centuries of relative isolation from exposure to exotic animals, New Zealand was suddenly exposed to a new threat: the arrival of European traders and settlers. The arrival of Europeans, and the new range of plants and animals that accompanied them, accelerated a process of insect introduction that continues today. Larger, faster sailing vessels had the potential to carry insects all the way to New Zealand from their home countries, as well as from any trading ports in between, including those across the Americas, Asia, and Australia. The earlier canoe-based carriage of relatively few species, filtered out through time and distance, had ended.

Humans were not the only arrivals that carried insect stowaways. Exotic mammalian species accompanied the travelers, including the livestock and domestic animals that were to help populate the new farms. Although some pigs and goats were released in the 1700s, the majority of domesticated animals arrived in the following century.[3] Stock numbers increased substantially, such that they now outnumber humans by a ratio of more than 20:1. Currently New Zealand has an estimated stock population of more than 60 million, of which sheep and cattle predominate. Pets and undesirable animals, including new rat species, also arrived. The most common arthropods found on introduced mammals and birds included fleas, mites, and ticks. Sheep, the most common domesticated animal, brought with them a range of external parasites, including itchmites, body lice such as *Bovicola ovis*, and sheep keds (*Melophagus ovinus*). Many species of fly that are associated with livestock, such as *Lucilia sericata*, have also become widespread.

The past two centuries of European settlement have also seen an accelerated rate of pastoral transformation, degradation of indigenous ecosystems, and interruption of waterways. As Moran notes, "the burning of tussock grassland in the South Island was standard practice for over a century," while in the North Island "forests were ruthlessly fired, countless timber trees were destroyed, and rates of erosion on hill country increased until almost out of control."[4] All of these processes have provided new niche opportunities for exotic insects. Many species accompanied imported plants and were transplanted into an ecosystem that was often devoid of the usual predators. The massive depopulation of many native New Zealand bird species also removed an effective form of insect control.

Consequently, insect pests have taken advantage of almost every plant- or crop-based industry established in New Zealand. The codling moth *Cydia pomonella*, for example, is a pest native to southeastern Europe. Since its introduction, it has spread widely throughout the country, causing substantial damage to apples, pears, and stone fruit; in unsprayed orchards, more than 80% of the fruit may be damaged.[5] Other imported insects include numerous varieties of aphids, which cause plant damage by direct feeding or by acting as vectors for destructive viruses. Their introduction commenced in colonial times — for example, the cabbage aphid, *Brevicoryne brassicae* was recorded in the 1860s soon after the introduction of cruciferous plants[5] — and continues into the modern era, with the arrival of such species as the blue alfalfa aphid *Acyrthosiphon kondoi* and rose grain aphid *Metopolophium dirhodum*, a major vector for the spread of barley yellow dwarf virus (BYDV). Some introduced species may have both beneficial and adverse attributes, such as the European earwig *Forficula auricularia*, which attacks fruit, flowers, and foliage, but may help to control populations of wooly apple aphids (*Erisoma lanigerum*).

Many insect species have been introduced from nearby neighbors, such as Australia. One example is the soldier fly *Inopus rubriceps*, a sugarcane pest from Queensland that

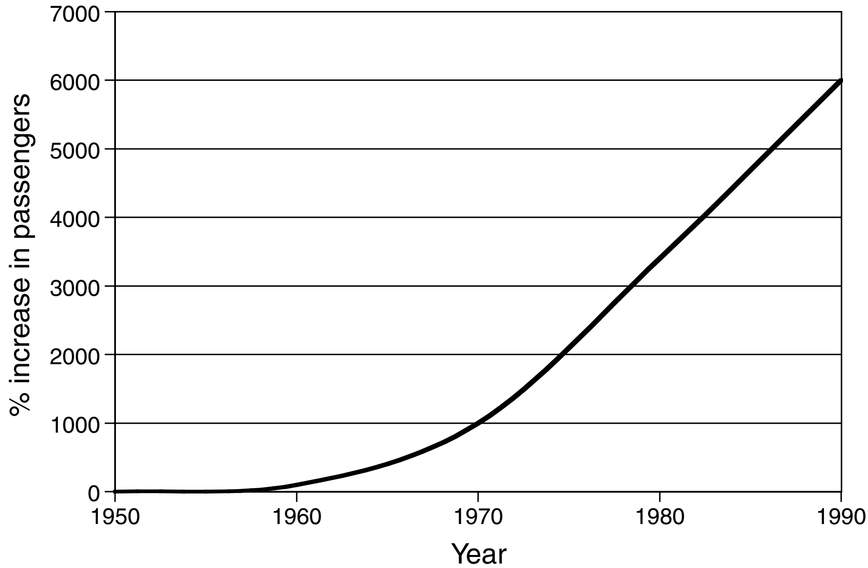

Figure 13.3 Percentage increase in passengers carried on international flights to and from New Zealand: 1950–1990.

was first observed in New Zealand in 1944.[5] The fly has since spread throughout much of the upper North Island, where it causes major damage to cereal and fodder crops, such as millet, oat, and maize. Habitat opportunities were also provided with the transportation and establishment of Australian plants, such as gum trees (*Eucalyptus*). For example, the eucalyptus tortoise beetle *Paropsis charybdis* was first detected on the South Island in 1916, and has since spread to affect eucalyptus plantations nationally.

13.2.4 Insect arrivals with the rise of global transport

The 20th century heralded other events that exposed New Zealand to further invasion by new species of insects. A major influence was the trend toward mass transport of people and goods across the national borders. This transition was ushered in by the two World Wars, which were associated with the mobilization of troops and equipment on an unprecedented scale. In subsequent decades, this was followed by new waves of immigrants and incoming goods, further diminishing New Zealand's isolation.

This increasing scale of movement was also accompanied by the greater efficiency and speed of transport. By the mid 20th century, seaborne cargo and passenger lines were being superseded by air travel. Globally, there has been an almost fiftyfold increase in air travel since 1949, with 1.1 billion air travelers a year by 1990. For New Zealand, total passengers on international flights to and from the country exceeded 3 million by 1990[6] (see Figure 13.3). New Zealand's inclusion on global air routes reduced the travel time to the country from days or weeks to mere hours. This, in turn, has provided opportunities for the rapid transport of insect pests, without the quarantining effect of a prolonged sea journey.

The quantity of both sea and air cargo unloaded has grown significantly in the same period, reflecting New Zealand's diversifying range of trading partners and imported goods. The removal of trade barriers has resulted in differing patterns of importation. For example, ideal insect "receptacles" — such as used vehicles — have flooded into the country. Despite the threat of insect carriage, it is only possible to inspect a small percentage of incoming cargo and containers.

An insect group that has taken advantage of increased levels of international travel and trade is the mosquito. Incursions by mosquito species raise the likelihood of an eventual outbreak of vector-borne human disease. For example, one of the best vectors of dengue fever in the world, *Aedes albopictus*, the Asian tiger mosquito, is easily transported internationally. In New Zealand, *Ae. albopictus* has been intercepted in border checks on numerous occasions, often in used tires that had been exposed to rainwater and mosquitoes in their Asian ports of origin. Although the species failed to become permanently established,[7] it is capable of adapting to cooler climates. One mosquito vector for another arbovirus, Ross River virus, has already penetrated the country's borders: The southern saltmarsh mosquito, *Ochlerotatus [Aedes] camptorhynchus*, is well established in the Hawke's Bay, Gisborne, and Northland regions.

Other recent arrivals include aggressive species of exotic ants. One pest that has proliferated widely is the Argentine ant, *Linepithema humile*, first identified in 1990, (although it may have been present prior to this date). The ants are well established in Auckland and have been transported to other locations within trash and plant material from nurseries. Infestation sites have now been identified on both main islands and on offshore sanctuaries (see Figure 13.4).[8] These ants, well known as major household and garden pests in other countries, form multi-nest colonies and are aggressive foragers, often replacing or killing other types of native invertebrates. In Australia, the species damages crops such as citrus, tomatoes, and grapes, indicating the potential threat posed to New Zealand horticulture and viticulture. The ants also place native birds at risk by their capacity to attack fledglings in the nest and by competing for food resources, such as honeydew and nectar. In addition to the Argentine ant invasion, a nest of red imported fire ants (*Solenopsis invicta*) was detected at Auckland International Airport in 2001. In contrast to ants already present in New Zealand, which have little direct impact on human health, fire ants tend to swarm when disturbed and are capable of inflicting painful stings.

13.2.5 Exotic insects in an era of climatic change

The establishment of insects in New Zealand in the past century is not only the product of large-scale and rapid travel. In the past few decades there has been greater awareness of the effects of climatic change. It has been suggested that altered weather patterns — including gradual warming secondary to greenhouse gas emissions — may allow the expansion of the ecological niches of a number of insect species. Together with the pathways for introduction described above, climatic change may allow the expansion of the ranges of existing pests, or may usher in exotic insects from which the country was once protected by virtue of its climate.

One identified trend over the past 30 years is that of increasing average global temperatures. Recent forecasts from the Intergovernmental Panel on Climate Change (IPCC) suggest that the trend will continue. Although New Zealand is experiencing less warming than most regions owing to its position in the South Pacific, it is vulnerable to any climate-related changes in its ecosystem.

For example, *increasing average temperatures* would allow range expansion for temperature-dependent species. It is anticipated that certain imported mosquitoes, ticks, and ants would broaden their range to higher latitudes and higher altitudes. For some species, including *Aedes* mosquitoes, breeding times are also shorter at higher temperatures.[9]

Increasing average rainfall would allow range expansion for rainfall-dependent species. A number of insects, particularly mosquitoes, would benefit from this trend. Disease patterns in Australia and the Pacific islands indicate that the heavy summer rainfalls are associated with breeding of *Aedes* and *Culex*, thus causing increased rates of Ross River virus disease and dengue fever.

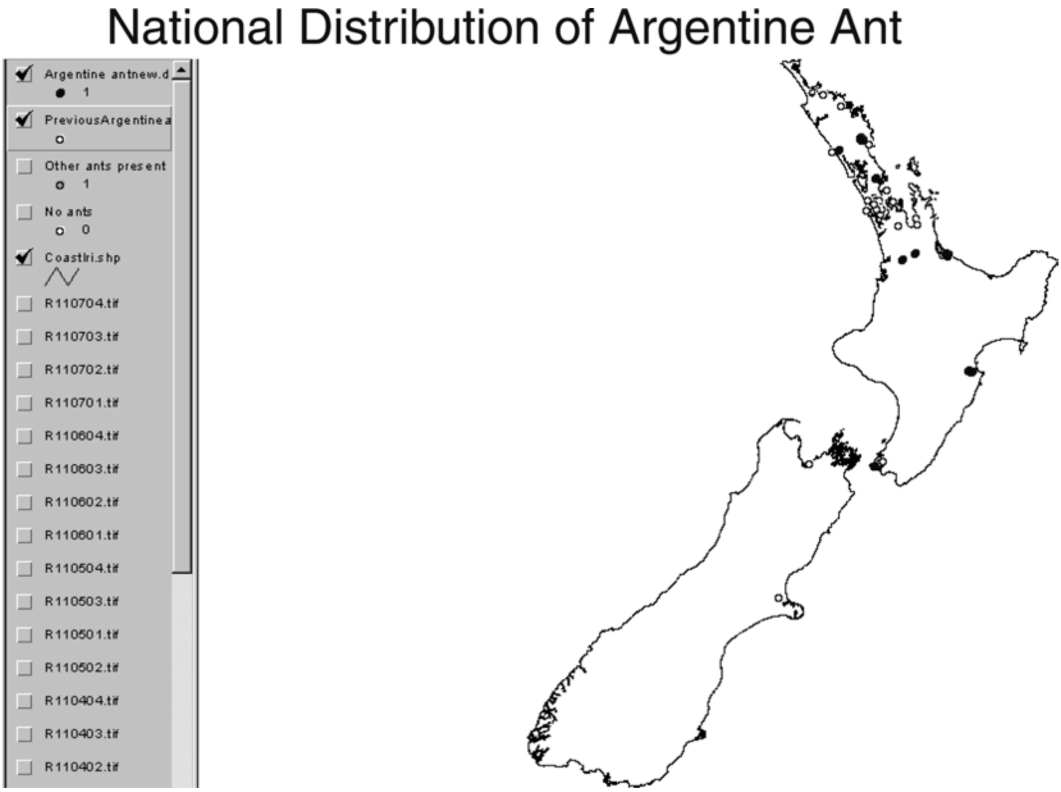

Figure 13.4 Extent of Argentine ant distribution for the period 1990–2001. (Map provided courtesy of Amelia Pascoe, Ministry of Agriculture and Forestry, Wellington.)

Coastal habitats may be further modified by *rising sea levels*, another probable consequence of global warming. Thus, low-lying parts of New Zealand may ultimately be vulnerable to flooding from two sources: increased rainfall and rising sea levels. Although this would result in habitat loss detrimental to some insect species, others — including, once again, some species of mosquito — may reap the benefits of such cumulative changes.

Insect habitats are also influenced by *any increases in the frequency of extreme climatic events.* While such variability is partly related to trends in global warming, periods over which the climate may be more extreme have also been linked to the El Niño Southern Oscillation, which is driven by changes in oceanic surface temperatures. When El Niño conditions prevail, increasing aridity in some areas of New Zealand may result; a particularly vulnerable region is the upper South Island, which suffered from drought and prolonged damage to its agricultural sector in the late 1990s. The flip-side climatic pattern is La Niña, which in New Zealand results in increased rainfall and floods, especially during summer. Either climatic consequence — aridity (and thus water stagnation) or flooding — could provide new niche opportunities for exotic insects.

Figure 13.5 provides an overview of the factors that, often in combination, may lead to the arrival and establishment of exotic insects.

13.3 Assessing the impact of exotic insects

Assessing the impacts, and thus the inferred costs, of insect invasions is a complex task. Estimates of the impacts of invasion, even for single species, may be difficult to quantify

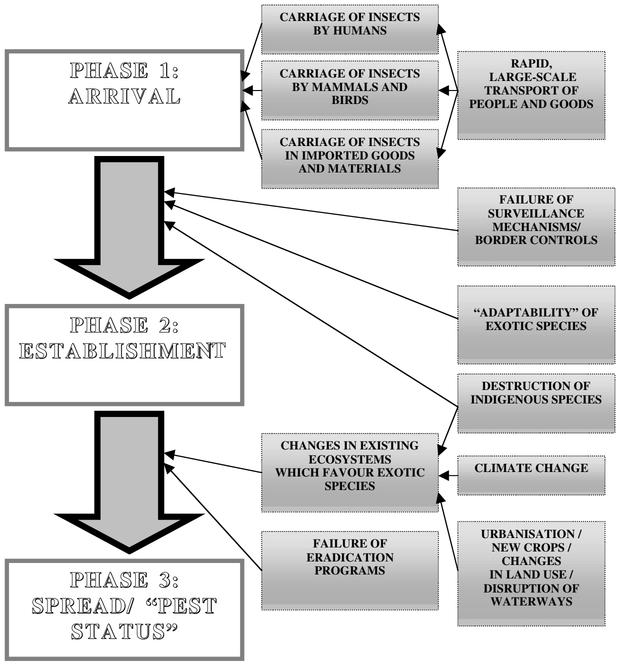

Figure 13.5 Factors contributing to insect arrival, establishment, and spread.

and may vary widely. For all imported pests (plants, insects, mammals, etc.), recent estimates put the economic losses at $NZ440 million per year, and costs of control and eradication at $400 million per year.[1] (Monetary sums in this chapter are expressed as New Zealand dollars.) Therefore, it is probable that the detrimental costs of insect incursions alone have cumulatively cost the country billions of dollars in the past decade. However, such impact analyses are often limited in their scope, and may often exclusively consider effects on particular crops or the costs of eradication. In fact, invasions may have wide-ranging effects, including damage to indigenous ecosystems and effects on tourism and human health.

In this section, the process of impact assessment will be illustrated using three recent case studies of insect incursions in New Zealand. The presence of these selected species has diverse consequences: the southern saltmarsh mosquito incursion has implications for human health and quality of life; the painted apple moth threatens the horticultural sector and indigenous ecosystems; and the varroa bee mite has the potential to compromise a range of agricultural, apicultural, and horticultural industries.

However, although the specifics of each invasion vary, there are also a number of commonalities. The incursions illustrate the shortfalls and problems in the current surveillance mechanisms for incoming pests. All three species have penetrated border controls to establish niches in New Zealand, and their discovery has often resulted from chance identification, such as by members of the public.

Once the exotic insect is identified, and its range elucidated, additional challenges emerge in attempting to limit further spread. The ideal of complete eradication has often proved to be difficult or impractical. Controversy also arises regarding the best strategy for educating the public, and those in affected industries, about the risks posed both by the species and by any proposed control programs, such as aerial spraying of populated areas with insecticides. Finally, each case study highlights the uncertainties inherent in estimating current and projected costs of an insect incursion, even if the impact assessment is limited to a single species.

13.3.1 Human health and quality of life

In terms of compromising human health, the manner in which insects may act is diverse. One common mechanism is by causing stings and envenomation. In New Zealand, stings by bees and wasps represent the most significant health consequences, including life-threatening anaphylactic shock. Although some native bee and wasp species are capable of stinging humans, the usual aggressors are introduced Hymenoptera species. These include honeybees, bumblebees, and Australian paper wasps, all of which arrived in the 19th century. The 20th century also saw the introduction of a number of new species, such as the German wasp (*Vespula germanica*), which was well established by 1945, while common wasps (*Vespula vulgaris*) spread across most of New Zealand through the 1980s. The most recent arrival is the Asian paper wasp (*Polistes chinensis*), which is particularly concentrated in the Auckland–Northland area, although its range is extending further south. New Zealand has the highest recorded densities of wasps in the world,[10] with all four species peaking in abundance over the late summer and autumn months of February to May. Dymock[11] estimated that at least 850 people in New Zealand seek medical attention for wasp stings annually, although the actual number of sting injuries is likely to be between two- and tenfold greater because of underreporting.

Human contact with imported arthropods may also result in envenomation. One arachnid that has gained some notoriety in New Zealand is the Australian red-back spider, *Latrodectus hasselti*. The presence of this arachnid species represents a risk to health superseding that posed by the native, reclusive katipo (*Latrodectus katipo and atritus*). Red-back spider bites may result in painful inflammation, and occasionally even death. In 2001, the importation of table grapes from the United States was suspended following repeated discoveries of black widow spiders (*Latrodectus mactans*) in supermarket produce.

Apart from inflicting stings and bites, insects may cause infestations. Lice and scabies probably all arrived with the first Polynesian settlers, and even today infestation still commonly results in illness, including severe skin irritation. The frequency of human infestation with the scabies mite, *Sarcoptes scabiei*, has fluctuated over the past century, with massive outbreaks during World War II,[12] a decline during the 1950s and ’60s, and a slow increase since that time.

Although it is probable that fleas also accompanied the Maori immigrants, the subsequent introduction of European animals significantly increased the opportunity for infestation. The most common species to breed in residences and affect humans is the cat flea, *Ctenocephalides felis felis*,[12] although fleas from dogs, poultry, and wild birds may also bite humans. Fleas may also act as vectors for disease. Historically, this mode of transmission was of concern. In the early 1900s, New Zealand was affected by periodic outbreaks of bubonic plague, transmitted by rat fleas. In the same period, cases of typhus and relapsing fever transmitted by lice were also reported. Although such diseases have faded from prominence for almost a century, there is potential for vector-borne diseases to reemerge. The threat is primarily posed by exotic mosquitoes.

Other parasites from imported birds and mammals may also affect health. Mites from introduced pigeons, starlings, and poultry may cause skin rashes in humans. Widespread introduction of mammals and birds also increased the opportunity for contact with ticks, the most common of which is *Haemaphysalis longicornis*. Ticks commonly infest sheep, cattle, and dogs,[13] but they may be passed to humans, particularly onto uncovered skin. Some movement of ticks has also occurred across the Tasman Sea, with certain Australian species of tick, including *Ixodes holocyclus*, having been identified in New Zealand. In Australia, this species may cause paralysis in dogs and humans.

Human health is not only compromised by the insects directly, but also by agents used for the eradication and control of pests. The widespread use of insecticides has resulted in direct toxicity to humans, as well as to non-target organisms, soil, and waterways. The use, transport, and disposal of toxic pesticides all present opportunities for exposure. Popular chemicals have included the organophosphates, which can produce severe poisoning in humans. A further impact on human quality of life is the degree of public concern that is generated during attempts at eradication. For example, the use of sprays to control insects, such as the aerial spraying program over Auckland that was designed to eradicate the white-spotted tussock moth (*Orgyia thyellina*) in 1999, may meet with widespread public fear and opposition.

The health consequences of insect invasions must also be considered in much broader terms, beyond the consideration of insects merely as causative agents in specific illnesses. Economic impacts also arise because of the cost of instituting preventive campaigns to preserve health, such as in providing public education regarding health risks posed by insects or exposure avoidance. Even for insects that do not create disease directly, indirect effects may impinge on health. Widespread devastation of crops, particularly in an economy heavily dependent on primary resources such as New Zealand's, may destroy livelihoods and raise the general level of regional poverty.

13.3.2 Agriculture

Agricultural production is the major industry in New Zealand, and numerous animals have been introduced for commercial purposes since European settlement. These include sheep, beef and dairy cattle, poultry, pigs, goats, and deer. More recently, an array of new pastoral arrivals has been introduced from abroad, ranging from ostriches to alpacas. With the overwhelming reliance on agricultural activities, exotic arthropods that compromise animal health pose a major threat. Such parasites may irritate, weaken, or even kill animals; damage may also affect the wool or hides, thus detracting from their commercial value.

Flies, particularly introduced blowflies such as *Lucilia sericata*, may result in more severe disease or death by causing flystrike, particularly among lambs. In this condition, maggots and larvae proliferate in the soiled fleece and may burrow into the flesh; the effects range from wool damage to distress or death of the animal.[20] It has been estimated that up to 3% of the national sheep flock is stricken each year; lamb deaths may reach 180,000 annually.[5]

The other groups of arthropods of agricultural importance in New Zealand are ticks, lice, and mites. Ticks, although a major problem in other countries, are of lesser importance in New Zealand, and only one species, *Haemaphysalis longicornis*, commonly affects livestock. However, severe tick infestation may result in anemia, loss of condition, low milk production, and damaged hides. Other ectoparasites, such as lice and mites, do not typically result in severe mammalian or avian illness, although severe local inflammation may be very distressing and may result in failure to thrive.

A recent arrival that has the potential to cause significant economic loss is the honeybee mite, *Varroa jacobsoni*. This pest may disrupt not only the apicultural industry, but also pastoral and horticultural industries that rely on bees for crop pollination.

13.3.3 Horticulture, arable crop production, and forestry

Given the large amount of land under cultivation in New Zealand, the impact of exotic insects on the agrarian economy is of major concern. In the years since human habitation, and particularly with the pastoral transformation that occurred with European settlement, few crops or plant-based industries have escaped the effects of introduced pests. The large-scale destruction of native bush and tussocklands, and replacement by imported plant species, has been associated with a proliferation of new species. Certain native species have also taken advantage of the new dietary opportunities as well, such as the grass grub, *Costelytra zealandica*, which was once localized only to tussock land but now feeds on a wide range of pastureland, grain, fodder crops, and nursery plants.

Because of New Zealand's reliance on livestock industries, the maintenance of pastures and the production of cereal, forage, and seed crops are of particular importance. Introduced pests with significant economic impacts include the Australian soldier fly; the Argentine stem weevil *Listronotus bonariensis*, which destroys maize and rye crops; and the black beetle *Heteronychus arator*, which originated in South Africa but probably traveled to New Zealand via New South Wales, Australia, and which feeds on pasture grasses. Insect pests have also established themselves with each new horticultural development, including the production of pipfruit, citrus fruit, vegetables, berries, and grapes.

Forestry is another major industry in New Zealand. Extensive planting has taken place in the past century, notably in areas of the North Island where the soil is less suitable for farming. As of June 1998, commercial forests covered around 6% of the country's land area, or 1.6 million hectares. The most popular introduced tree is *Pinus radiata*, a native of North America that thrives in New Zealand's mild, moist climate. This tree species alone, which has been planted extensively since the 1920s, provides a suitable habitat for dozens of accidentally introduced insects, such as burnt pine longhorns (*Arhopalus ferus*, transported from Europe in the 1950s in cable drum battens) and the wood boring wasp *Sirex noctilio* (introduced from Europe in the 1920s and first associated with severe outbreaks in New Zealand in the 1940s).

Wood products are also threatened by the presence of a number of imported pests. Household borers, such as *Anobium punctatum*, are well established and were probably introduced with the first European settlers.[5] More recently, Australian subterranean termites (*Coptotermes acinaciformis*) have been identified in North Island localities. It is believed that this species may have arrived in power-pole timber imported from Australia in the 1930s.[8] Termites potentially feed on a variety of wood products, including wooden houses and fence posts. Their potential for widespread damage in New Zealand, where wood is an extremely popular building material, remains uncertain.

13.3.4 Indigenous ecosystems

In the past millennium of human habitation, the impact of alien insect incursions on the original ecosystem of New Zealand has been profound. The end of the country's long isolation was accompanied by the extinction or endangerment of more than 1000 native plants and animals, the destruction of two thirds of the native forest, and the introduction of numerous species, many of which have become pests. Exotic insects, introduced either deliberately (for example, the honeybee) or accidentally, have contributed to this decline in indigenous biodiversity.[8]

Indigenous shrublands and forests may be adversely affected by the feeding patterns of incoming species. For example, the painted apple moth — first identified in New Zealand in 1999 and probably introduced in shipping containers from Australia — is a voracious consumer of a wide range of vegetation. The species poses a potential risk to many indigenous plant species, including the ribbonwood and the kowhai, a flowering tree that is vulnerable to full defoliation by moth larvae.[8]

Effects on the viability of native species may be less direct, as illustrated by introduced wasp penetration of indigenous beech (*Nothofagus*) forests. For 5 months of the year, wasps — predominantly the German and common species — reduce the standing crop of aphid-secreted honeydew by more than 90%. As a consequence, the wasp population is now in direct competition with many native birds, such as kaka (*Nestor meridionalis*), tui (*Prosthemadera novaeseelandiae*), and bellbirds (*Anthornis melanura*), that rely upon honeydew as a ready source of carbohydrates. Wasps also compete with a variety of birds for invertebrate food. Recent reports indicate that the foraging opportunities and behaviors of certain native birds have been disrupted as a result.[23]

In some cases, exotic species may directly eliminate indigenous insects. One recent example of this process is the direct parasitism of native scarab beetles by scoliid wasps (*Radumeris tasmaniensis*). The wasps, which originated in Australia, were first identified in 2000 on beaches in the Northland region. The adults sting and paralyze the larvae of scarab beetles, which then serve as a nutrient source for newly hatched wasp larvae.

Finally, methods of insect control may also have pronounced environmental effects. Commonly used insecticides, such as synthetic pyrethroids, may be inadvertently washed into waterways and end up poisoning aquatic animals.

13.3.5 Tourism

New Zealand's tourism industry often portrays the country as a clean, safe destination with a unique environment. The presence of exotic insects poses a clear threat to this image and could compromise tourist numbers. One example of a favorable attribute which may be affected is New Zealand's freedom from many vector-borne diseases. The establishment of mosquitoes capable of carrying arboviruses, such as those causing dengue fever or Ross River virus disease, may act as a deterrent for some travelers. Any further degradation of the country's ecosystem, in which 90% of the native species are unique, may also result in the reduced desirability of New Zealand as a destination.

13.3.6 Integrity of biosecurity systems

The biosecurity system in New Zealand has undergone radical change in the past decade. In its original form, each component of the biosecurity program was the responsibility of a different government department and ministry, with separate administrations. However, a number of insect interceptions and incursions in the 1990s, particularly by the white-spotted tussock moth and the Asian tiger mosquito, contributed to a major policy review. As a result, a new administrative structure emerged, with the appointment of a minister of biosecurity and a Biosecurity Council. One of the council's principal objectives was to permit a more integrated response to biosecurity threats, and from its inception it included representatives from a range of departments and ministries, including the Department of Conservation, the Ministry of Agriculture and Forestry, the Ministry of Fisheries, and the Ministry of Health.

The components of the biosecurity system in New Zealand may be defined as follows:

- Monitoring at the pre-entry phase
- Monitoring at the entry phase (that is, the maintenance of biosecure borders)
- Surveillance of exotic species within New Zealand
- Response capacity to control or eradicate exotic species

The first two components of the system are directed at the exclusion of unwanted insects, along with other pests, from the country. Examples of exclusion strategies include monitoring of insects at the pre-entry phase, such as by inspection and spraying of cargo at the port of origin, and the maintenance of biosecure entry points through border inspections of luggage and cargo. The feasibility of such monitoring is assisted by the fact that New Zealand is an island nation, which prevents direct terrestrial invasion across frontiers with other countries. The wide separation even from near neighbors further restricts opportunities for natural waterborne or airborne insect spread.

However, the heavy reliance on trade and tourism, and the frequent border crossings that result, have tended to overcome these natural barriers. In the past decade, total passenger numbers and the weight of cargo unloaded at New Zealand's ports have both doubled. Despite the greater awareness of biosecurity and the commitment of governmental funding, border inspection facilities are at present being overwhelmed. It is only possible to review a small percentage of the air cargo, mail, and shipping containers that arrive in the country. In 1999–2000, only around 25% of the 360,000 sea containers entering New Zealand were inspected; of those inspected, one fifth were found to be contaminated with plant or animal material. The capacity to prevent incursions by unwanted insects, plants, and animals is thus severely compromised.

Recent examples indicate ways in which strategies for exclusion may be breached. Many incursion events, such as the carriage of Asian gypsy moth eggs in imported used vehicles from Japan, are unintentional. Live insects may also be transported unwittingly in tents or other materials carried by travelers. In other cases, importation may occur illegally. This may have occurred with the recent importation of the parasitic varroa bee mite, which may have spread from a queen bee smuggled into the country.[8]

The other essential components of the biosecurity system — surveillance of exotic insects within New Zealand, and the response capacity for control or eradication — are implemented once an exotic insect has penetrated the borders. Examples of such measures, and an assessment of their efficacy, are described in three case studies in the next section.

13.4 Case studies

13.4.1 The health impact of the southern saltmarsh mosquito

New Zealand is one of only a handful of countries in the world where there is no known transmission of human arboviral disease (arthropod-borne viral disease). Such viruses, which cause a range of diseases, including dengue fever, Ross River virus disease, and Japanese B encephalitis, are chiefly transmitted by mosquitoes. Although New Zealand has 12 endemic mosquito species, they lacked any opportunity at all to be exposed to either humans or human arboviruses until Maori colonization some 1000 years ago. However, it is likely that the Maori were free from arboviral disease because of the spatial and temporal infectious disease filter provided by ocean voyaging. The vector competence of such endemic mosquitoes for human arboviruses is therefore likely to be low, although this has never been investigated.

As described above, the potential for human arboviral transmission in New Zealand rose sharply with European colonization and the introduction of three exotic mosquitoes: *Ochlerotatus [Aedes] notoscriptus*, *Ochlerotatus [Aedes] australis*, and *Culex quinquefasciatus*. Although these species have demonstrated arbovirus vector competence in laboratory settings,[7] there is no evidence of local transmission of arboviruses, and the only infections occurring in New Zealand are from people who have contracted such diseases while abroad. In other words, although potentially capable of arbovirus transmission, these exotic species in reality fail to display very high vector competence. However, this may not be true of invasive species in the future.

13.4.1.1 The incursion and response

In December 1998, the local council in Napier (see Figure 13.6) began receiving complaints from residents who were being fiercely bitten during the day by large numbers of mosquitoes. The Australian southern saltmarsh mosquito, *Oc. camptorhychus*, was identified in the area, the first record of this mosquito in New Zealand. This species is known as a vector of the arbovirus Ross River virus (RRV) in southeastern Australia. Larval sampling and adult light trapping indicated that the infestation was limited to saltmarsh habitat amounting to about 650 ha to the north and south of Napier,[14] and eradication was considered feasible. Initially, *Bacillus thuringiensis var. israelensis* (Bti) was applied to the area from both air and ground, later to be replaced by a pesticide in the form of (S)-methoprene pellets. No adults have been trapped in Napier since April 2000,[15] but additional infestations were detected in Gisborne, Mahia, and Porangahau. Eradication was pursued and extended to include these areas until further infestations were detected at Kaipara and Mangawhai. At that time, the feasibility of eradication became questionable, and the government opted to restrict the effort in the extensive Kaipara and Mangawhai areas to one of containment and control.[15]

It remains unclear whether the infestations outside of Napier represent secondary spread, multiple incursions, or, most likely, long-established populations that were detected neither by surveillance nor by early delimiting surveys. Surveillance for introduced pests of public health significance in New Zealand has historically been limited to periodic surveys that were not targeted according to an assessed risk of mosquito introduction and establishment.[16] Despite the historical establishment of three exotic mosquito species in New Zealand — and repeated interceptions of at least three others, *Aedes albopictus*, *Ochlerotatus [Aedes] japonicus,* and *Culex annulirostris* — there were no well-developed contingency plans for dealing with mosquito incursions when the southern saltmarsh mosquito was detected.[14]

13.4.1.2 The risk and cost

Oc. camptorhynchus is a significant vector of Ross River virus in some coastal regions of Australia; the virus has been isolated from field-collected specimens in Gippsland (Victoria) and the east coast of Tasmania,[17] suggesting that a temperate climate is no impediment to its activity. The presence of this mosquito in New Zealand therefore intensifies the risk of an outbreak of RRV, given that a number of key factors have now converged: (a) there is extensive movement of people across the Tasman, some of whom could transfer the virus from Australia; (b) populations of competent vector mosquitoes have become established; (c) there is the presence of a large non-immune human population; and (d) the climate is compatible with arbovirus outbreak ecology.[18]

In Australia, outbreaks of RRV result in more than 5000 reported cases per year; the disease is characterized by severe joint pains and often is accompanied by rash, fever, fatigue, and/or myalgia. Most cases resolve within 6 to 8 weeks, but symptoms may last longer and sometimes cause relapses for years. A worst-case scenario is that of a "virgin soil" outbreak, such as occurred when the virus was inadvertently introduced into the South Pacific basin, presumably by a viremic traveler; more than 500,000 people were infected, and more than 10% of these suffered seriously debilitating symptoms.[19] The livelihood of some smaller communities was put at risk by the immobilization of several important members, and medical services were under severe pressure.

To quantify the medical and social costs of an outbreak of RRV in New Zealand (including possible losses to the tourist industry) is beyond the scope of this chapter. However, the presence of *Oc. camptorhynchus* in New Zealand was sufficient cause for concern for the government to fund the initial eradication attempt, at an estimated cost of approximately $7.6 million. In addition, the recent commitment of $6 million to ongoing control at newly identified sites[15] brings the current cost of mosquito control alone to $13.6 million. Sadly, *Oc. camptorhynchus* now seems likely to remain in New Zealand, providing an ongoing cost for mosquito control that New Zealand had so far avoided. Incursions of

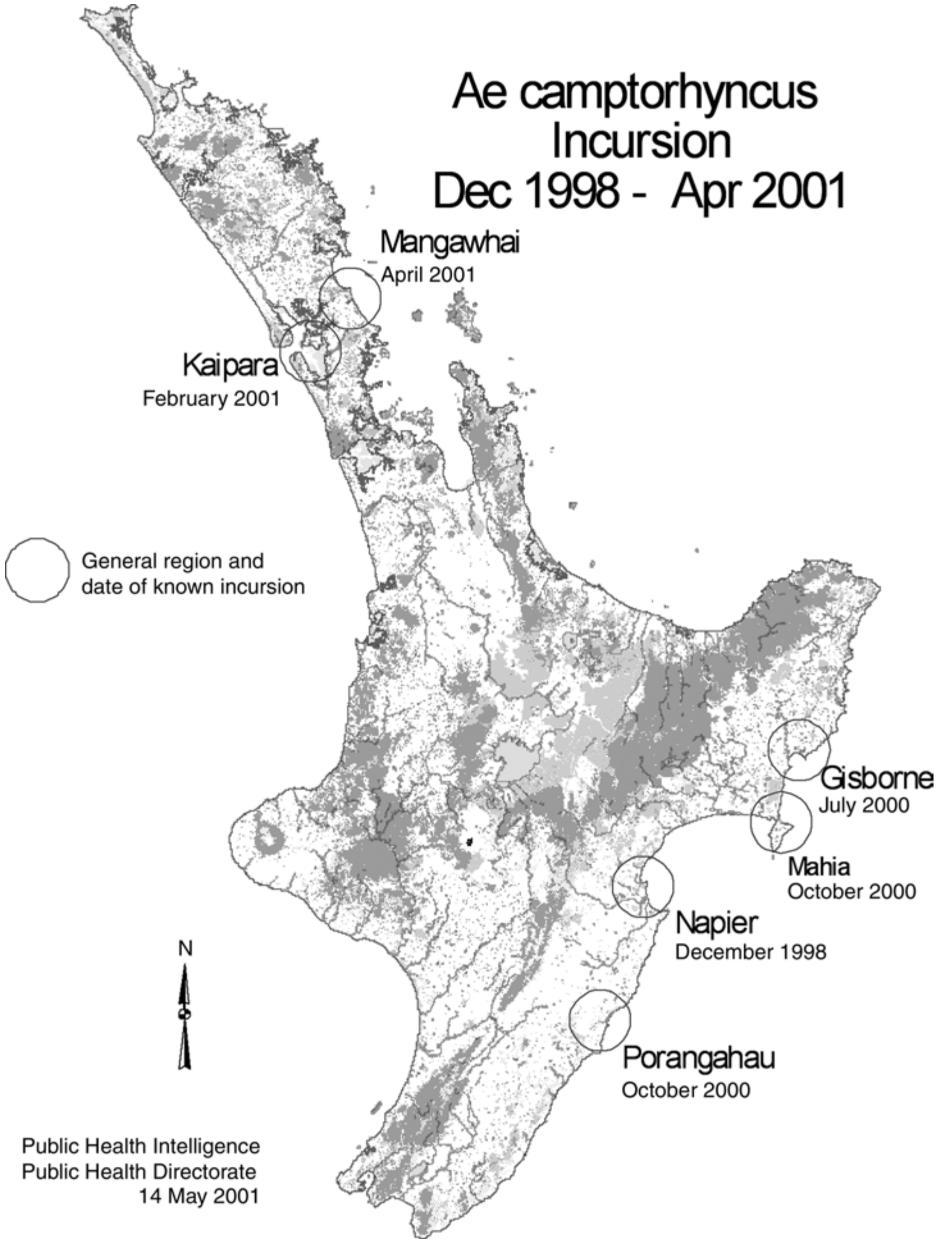

Figure 13.6 *Ochlerotatus* (Aedes) *camptorhynchus* incursions on the North Island for the period December 1998 to April 2001. (Map provided courtesy of Dr. Chris Skelly, Public Health Directorate, Ministry of Health, Wellington.)

further exotic mosquitoes with vector potential can only lead to an escalation of such mosquito control expenditures.

13.4.2 The industrial impact of the honeybee mite

The arrival of the honeybee mite *Varroa jacobsoni* in New Zealand illustrates an unusual situation: An exotic arthropod, the mite, has threatened the viability of a major industry that is based on another exotic arthropod, the honeybee. The honeybee originally arrived in New Zealand in the 19th century, but any further importation of bees has been prohibited since 1954 to prevent disease introductions.

The *Varroa* mite originated in eastern Asia and spread to Russia and Europe. It has since been carried into most other countries that have apiculture industries, excepting Australia. The entire North American continent was affected within 5 years of mite introduction. The mite feeds on bee pupae, either killing them or causing them to grow into deformed adult bees. Usually the entire hive dies once infestation is established. Although the mite is carried naturally to other hives by drifting or robbing bees, the spread of the pest has been considerably accelerated by human practices, including the movement of hives to encourage pollination or to gather honey.

Because it was previously free of the disease, New Zealand established itself as a major exporter of live bees and queens to the Northern Hemisphere.

13.4.2.1 The incursion and response

The honeybee mite is believed to have been introduced in the late 1990s, possibly as a result of the smuggling of an infested queen bee into the country.[8] After the mite's discovery in beehives in Auckland in 2000, the greater Auckland region was declared a controlled area. This measure was designed to restrict the movement of live or dead bees, honey, hives, and beekeeping equipment and appliances within the designated area, or from one area to another. Visits to 3106 apiaries (representing 60,479 hives) in mid-2000 indicated that 309 apiaries (4282 hives) were infected; all were in the upper half of the North Island (see Figures 13.7a and 13.7b). The infested apiaries were highly clustered around Auckland, Helensville, and the Hauraki Plains.[21] At the time of this writing, the lower North Island and South Island had been designated free of the mite.

Unfortunately, the prospects of removing the pest are not favorable; no country has succeeded in eradicating this mite.

13.4.2.2 The risk and cost

The discovery of *V. jacobsoni* in New Zealand poses an immediate threat to the nation's $1.8 million export trade in live queens and other bees, which involves the export of more than 300 million live bees a year. Furthermore, the presence of the mite may also impact trade in honey and other bee products, which have relied on New Zealand's image of "clean" origin.

More widely, there are serious implications for both pastoral agriculture and horticulture sectors. Pastoral farmers, particularly those reliant upon intensive land use, such as for dairying, are dependent on bees for clover pollination. Similarly, pollination of trees and vines is required for most orchard-based industries. Overall, it has been estimated that the total value of disease-free bees to the bee and honey industry, and for pollination for agriculture and horticulture, ranges from $400 million to $900 million per year.

The management options for control of the mite have been reviewed, and it was concluded that eradication was unlikely to be an effective strategy. The government instead approved an interim management program that is aimed at treating high-risk hives and at keeping the South Island *Varroa*-free for as long as possible. In 2001, a total of $7.7 million was committed to the combined costs of surveillance, treatment, and research.

13.4.3 Ecosystem danger: the painted apple moth

The painted apple moth, *Teia anartoides*, is a native of Australia. The moth consumes a wide variety of vegetation, including acacias, pine trees, and fruit crops. The moth is able to pupate and lay eggs on inanimate objects, and thus can be unintentionally transported over long distances. It was first identified in New Zealand in 1999, and is believed to have been transported as eggs laid in shipping containers.

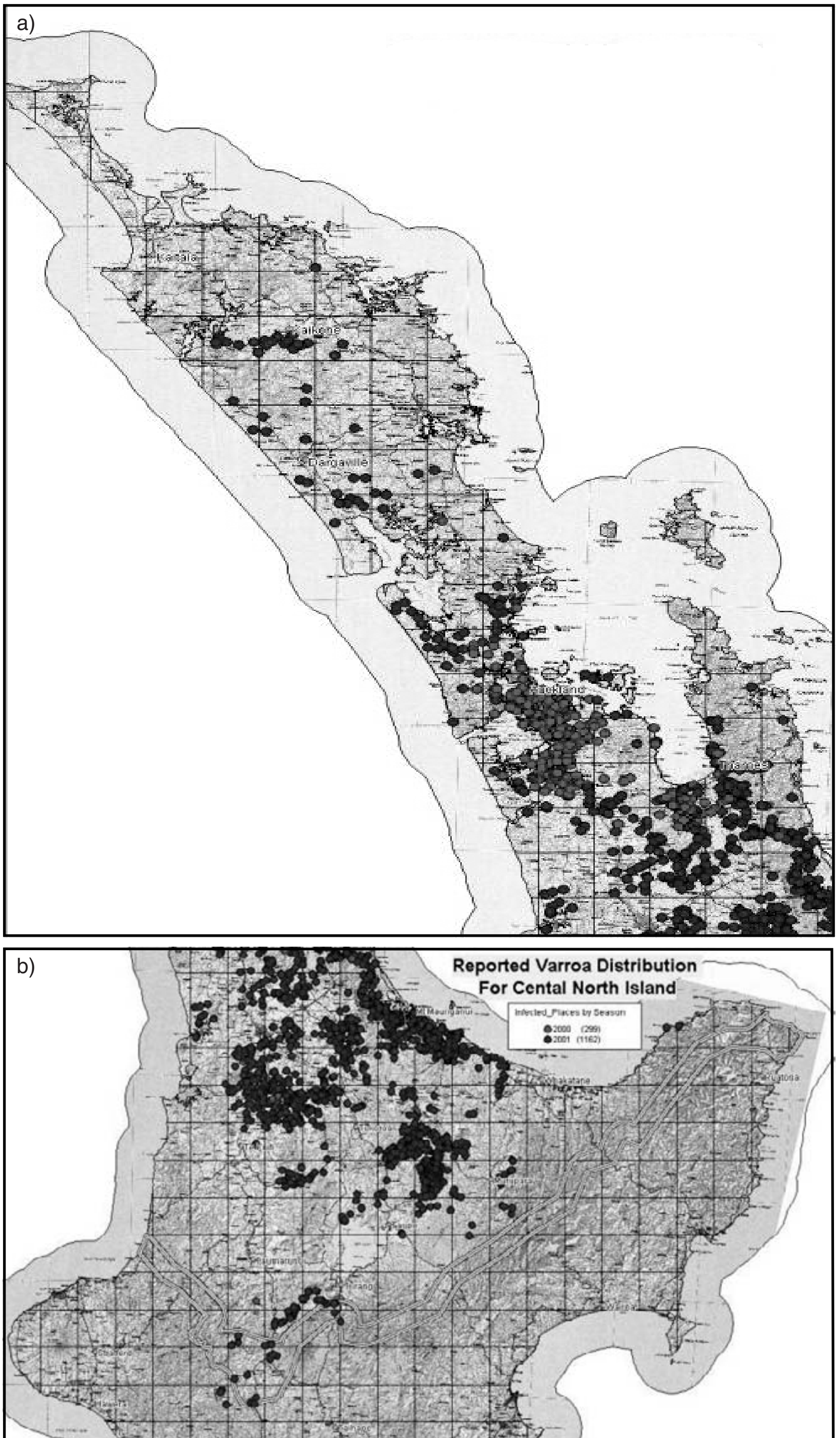

Figure 13.7 a) Sites of Varroa bee mite identification on upper North Island for the period 2000–2001. b) Sites of Varroa bee mite identification on central North Island for the period 2000–2001. (Maps courtesy of the Biosecurity Authority, Ministry of Agriculture and Forestry, Wellington, New Zealand.)

13.4.3.1 The incursion and response

In May 1999, the larvae of painted apple moth were first identified by a member of the public in the west Auckland suburb of Glendene. Five months later, in an incident believed to be unrelated, the moth was identified in another suburb. Based on the developmental stage of the identified moths, it is thought that the species may have been present for at least a year prior to identification.

The extent of infestation was determined by a delimiting survey. Ground spraying of trees, buildings, and shipping containers was conducted using two insecticides, chlorpyrifos and deltamethrin. A ban was placed on removal or transport of vegetation and other risk materials from infested sites. The sites were then checked weekly to determine the effectiveness of spraying. In late 2001 it was concluded that ground spraying had not eradicated the species. Targeted aerial spraying over western suburbs of Auckland was planned using *Bacillus thuringiensis var. kurstaki* (Btk), an organic insecticide containing a naturally occurring soil bacterium.

13.4.3.2 The risk and cost

The painted apple moth consumes a variety of vegetation, including native New Zealand plants such as the kowhai and ribbonwood. According to a Ministry of Agriculture and Forestry appraisal, the moth may "seriously impact on New Zealand's forestry, conservation and horticulture."[8]

An economic impact report published in 2000[22] assessed the potential effect of the moth's establishment in New Zealand. Several major areas of impact have been suggested:

Private and public gardens. Based on the Australian experience, the painted apple moth has the potential to become a common garden pest. Thus, increased spraying would be required for private and public gardens and grounds (including parks, reserves, golf courses, schoolyards, etc.).

Forestry. It is likely that New Zealand's extensive exotic forests may be susceptible to the moth, including radiata pines and eucalypts, both of which are able to host the moth. Although moth infestation is unlikely to result in tree death, growth rates may be affected following extensive defoliation.

Human health. Some people may suffer from skin rashes and respiratory problems after contact with the hairs on painted apple moth larvae. Although the reactions are usually mild, hospitalization may be required for sensitive individuals. Thus the costs of medical treatment and loss of work should be considered, although these are likely to be relatively small.

Trade prospects. In the event of moth establishment, it has been suggested that trading partners that are currently free of the moth may impose import restrictions on New Zealand goods.

The impact report estimated that the costs generated by painted apple moth may reach $47.6 million over the next 20 years. This figure was acknowledged as conservative, and it excludes any effects on indigenous ecosystems. It has been suggested that the moth could potentially cause widespread damage to native forests and shrublands, and may directly compete with native organisms for food.

13.5 Discussion

Over the past centuries, New Zealand has been subject to the importation of numerous exotic insects. Assessing the impacts of these invaders is a complex task. Numerous economic and environmental sectors may be simultaneously threatened, as occurred with the establishment of the *Varroa* bee mite. It may be impossible to provide cost equivalents for outcomes such as the disruption of indigenous ecosystems (for example, the direct parasitism of native scarab beetles by imported scoliid wasps) or impaired quality of life (such as from proliferation of *Vespula* species, which inflict stings). Any estimates of impacts for the immediate future, and related cost projections, are also limited by many uncertainties. For many exotic insects, particularly those introduced in the past decade, it is not clear when, or at what environmental limits, expansion will cease. Equilibrium is unlikely to be reached for some years or decades, and the progressive encroachment of many species suggests that penetration into new niches may be far from over, as illustrated by the spread of the southern saltmarsh mosquito along the country's east coast.

Despite these limitations, it is possible to attempt a partial quantification. For the purposes of this discussion, the analysis will be limited to the three pests described in the case studies. For these species, the incurred and projected costs over the next 20 years have been estimated and aggregated in Table 13.1.

This basic analysis, which uses relatively conservative estimates, indicates that the incurred and projected costs for these three species alone represent a considerable national burden. The value given is equivalent to approximately 0.73% of New Zealand's gross domestic product for one year (using national GDP for 1999). Furthermore, it is likely that sustaining the economic and environmental impacts of exotic insects will become increasingly difficult. The problem of incursions is becoming more acute, with the dramatic reduction in travel times and the large increase in cross-border traffic. Although $38 million was committed to border security in the 2000–01 budget year, such measures, including cargo inspections, are clearly becoming less effective at excluding these surreptitious immigrants.[25] The maintenance of a fully biosecure border is unlikely to be achievable. Other identified factors that may compromise the nation's biosecurity include: difficulties in predicting which exotic species will arrive, and whether they will be viable in the New Zealand environment; limited progress in biosecurity research and policy, and the resulting delays in establishing priorities; and limited funding for managing emergencies.[8]

Biosecurity measures are also contingent on public acceptance. Consensus on the relative risks and benefits posed by new insect invaders may be difficult to achieve. Public perceptions of campaigns to eradicate or control insects often diverge widely, especially if the process involves the application of insecticides over a populated area. In New Zealand, the desirability of eradication also differs between rural and urban populations. City-dwellers (such as those in the port cities, where exotic insects often first arrive) may not perceive any direct benefits from control measures, particularly when the threat an insect pest poses to rural industries lies in other parts of the country. In summary, although the technological capacity may exist for complete eradication, public acceptance of programs, or their associated monetary costs, may be a limiting factor.

Further problems will be created by ongoing environmental transformation and fragmentation. Cultivation is already extensive, and it is likely that continued impingement on indigenous forests and tussocklands will provide additional opportunities for exotic insects. The ecological equilibrium may also potentially be disrupted by climatic change. Although any long-term effects of global warming (or other climatic instabilities) are uncertain, any trend toward a more humid, subtropical climate is likely to be advantageous to many incoming arthropods, including mosquitoes, cockroaches, ticks, and ants.[26] Climate change and colonization by overseas species are not a new phenomenon to the New Zealand landmass, but they are currently occurring on an unprecedented scale.

Table 13.1 **Incurred and Projected Costs of Insect Invasions**

	Costs incurred	Projected costs over the next 20 years, if control measures fail	Totals
Southern saltmarsh mosquito	$13.6 million	$46 million	$59.6 million
Varroa bee mite	$7.7 million	$600 million	$607.7 million
Painted apple moth	$2.2 million	$47.6 million	$49.8 million
			(i) Total costs: $717.1 million
			(ii) Costs as a percentage of New Zealand's GDP in 1999: 0.73%

In response to a growing awareness of biosecurity issues, a number of strategies have been proposed and initiated to limit the impact of exotic invasions in New Zealand. For the purposes of this chapter, these strategies may be categorized as follows:

1. National
2. Trade-affiliated (that is, strategies involving New Zealand's principal trading partners)
3. Global

At a national level, the formation of the Biosecurity Council and allocation of government funds for related research, education, and monitoring indicate the emerging prioritization of biosecurity issues. Despite these advances, recent literature indicates that biosecurity preparedness could be improved at a number of levels[1,25] and that pre-existing quarantine measures should be incorporated into a wider framework by which to manage invasives. Future initiatives are required to anticipate which exotic species may arrive and to determine whether they will be viable in the New Zealand environment, to upgrade pest surveillance, and to manage incursive emergencies.[8]

A major difficulty is presented by New Zealand's commitments to trade liberalization and its engagement with an increasing diversity of trading partners. As with other island states, it is not socially or economically feasible for the country to recover its previous isolation. However, the benefits of incoming passengers and cargo needs to be calibrated against the risk of further erosion of New Zealand's position as a repository for unique flora and fauna, a desirable tourist destination, and a "clean" exporter of (principally) dairy and horticultural produce, meat, fish, and timber. Because biosecurity measures operating at a national level are becoming less effective as a primary means of defence, they need to be complemented by trade-affiliated strategies, which involve New Zealand's principal trading partners. At present, these predominantly include the nations of the Asian Pacific Economic Cooperation group (APEC) and, to a lesser degree, Europe and the Middle East. To minimize the risk of insect carriage from countries within our trading network, regulations are required to ensure compliance with health standards and pre-border evaluation of imports (and the containers in which they are transported). It has been suggested that emphasis would shift from "free trade" to "safe trade," in which the effects of importing patterns on the indigenous ecosystem are considered.[8] Addressing the invasive implications of trade has required states such as New Zealand to adopt a supranational approach, in which solutions extend far beyond our borders. Recent achievements have included the negotiation of bilateral quarantine arrangements with a number of Pacific island nations.

Finally, the issue of incursions must be addressed at a global level The extent of bioinvasion is currently the focus of several international research initiatives. For example, the Global Invasive Species Programme (GISP), in which New Zealand scientists participate, aims to define invasion pathways and impacts, and to explore strategies for the control and management of new species. Multilateral environmental agreements with relevance to biosecurity include the Convention on Biological Diversity and the associated Cartageña Protocol on Biosafety, issued in 2000. It has been acknowledged that such agreements have a potential to conflict with the trade liberalization philosophies of bodies such as the World Trade Organization, and many differences between them await resolution. Another issue with wide implications for biosecurity is that of climate change, including its effect on the establishment of exotic insects (particularly mosquitoes). The anthropogenic component of global warming and climatic variability will probably only be reduced through strictly enforced global accords, including those designed to limit greenhouse gas emissions.

In summary, it is evident that the global ecosystem is tending toward a state of increasing homogenization,[27] in which organisms may be transported thousands of miles and presented with niche opportunities once prohibited by natural barriers. Despite its distance from other populated centres, New Zealand has unequivocally become a participant in this global exchange of species. With human assistance, exotic insects have shown themselves to be quite capable of storming the beachheads. A synthesis of national, trade-affiliated, and global strategies will be required to meet the ongoing challenge posed by these new arrivals.

Acknowledgments

The authors would like to thank the following for their invaluable assistance in the preparation of this chapter: Dr. Ruth Frampton and Amelia Pascoe of the Ministry of Agriculture and Forestry Biosecurity Authority, and Dr. Chris Skelly of the Public Health Directorate, Ministry of Health, and Amy Snell, Medical Entomology Unit, Wellington School of Medicine.

References

1. Green, W., *Biosecurity Threats to Indigenous Biodiversity in New Zealand: An Analysis of Key Issues and Future Options*, EcoLogic Conservation Consultants, Wellington, 2000.
2. Suarez, A.V., Holway, D.A., and Case T.J., *Patterns of Spread in Biological Invasions by Long-Distance Jump Dispersal: Insights from Argentine Ants*, Proceedings of the National Academy of Sciences of the United States of America. 98 (3): 1095–1100, 2001.
3. Salmond, A., *Two Worlds: First Meetings between Maori and Europeans 1642–1772*, Viking, Auckland, 1991.
4. Moran, W., Systems of agriculture: regional patterns and locational influences, in *Society and Environment in New Zealand*, Johnson, R.J. (Ed.), Whitcombe and Tombs, Christchurch, 1974.
5. Scott, R.R., *New Zealand Pest and Beneficial Insects,* Lincoln College of Agriculture, Christchurch, 1984.
6. Statistics New Zealand, *New Zealand Yearbook 2000,* David Bateman Ltd, Auckland, 2000.
7. Weinstein, P., Laird, M., and Calder, L., Australian arboviruses: at what risk New Zealand? *Australian and New Zealand Journal of Medicine*., 25, 666, 1995.
8. Office to the Parliamentary Commissioner for the Environment, New Zealand Under Siege: a review of the management of biosecurity risks to the environment. Wellington, 2000.

9. Eberhart-Phillips, J., *Plagues on Our Doorstep*. Tandem Press, Auckland, 1999.
10. Thomas, C.D., et al., The prevalence of introduced Vespula vulgaris wasps in a New Zealand beech forest community, *New Zealand Journal of Ecology*. 13, 63, 1990.
11. Dymock, J.J., Forgie, S.A., and Ameratunga, R., A survey of wasp sting injuries in Urban Auckland from December to April in 1991/2 and 1992/3, *New Zealand Medical Journal.*, 107, 32, 1994.
12. Brock, R.L., *Medical Impact of Insects and Arachnids in New Zealand 1967–1976*, Department of Health Occasional Paper, Wellington, New Zealand, 1982.
13. Blackmore, D.K. and Humble, M.W., *Zoonoses in New Zealand*, Massey University, Palmerston North, 1987.
14. Garner, S., Browne, G., and Taylor, J., Kiwis thump Aussies — a saltmarsh mosquito eradication attempt in New Zealand, A*rbovirus Research in Australia*., 8, 168, 2001.
15. Ministry of Health press release: Government decision on the long-term response to the southern saltmarsh mosquito, Ministry of Health, Wellington, 2001.
16. Hearnden, M., et al., Improving the surveillance of mosquitoes with disease vector potential in New Zealand, *New Zealand Public Health Report.,* 6, 25, 1999.
17. Russell, R.C., *Mosquitoes and Mosquito-Borne Disease in Southeastern Australia*, Dept. of Medical Entomology, Westmead Hospital, Westmead, New South Wales, 1993.
18. Weinstein, P., An ecological approach to public health intervention: Ross River virus in Australia, *Environmental Health Perspectives*., 105, 364, 1997.
19. Marshall, I.D. and Miles, J.A.R., Ross River virus and epidemic polyarthritis, *Current topics in vector Research,* 2, 31, 1984.
20. Bruere, A. and West, D.W., *The Sheep: Health, Disease and Production*, Massey University, Palmerston North, 1993.
21. Benard, H., et al., *The Outbreak of Varroa Destructor in New Zealand Bees: Delimiting Survey Results and Management Options,* 2001 Conference Proceedings of the Food Safety and Biosecurity Branch of the New Zealand Veterinary Association.
22. Ministry of Agriculture and Forestry Policy, Potential economic impact on New Zealand of the painted apple moth, Ministry of Agriculture and Forestry, Wellington, July 2000.
23. Beggs, J., The ecological consequences of social wasps (*Vespula* spp.) invading an ecosystem that has an abundant carbohydrate resource, *Biological Conservation*, 99, 17, 2001.
24. Frampton, R., The Ministry of Agriculture and Forestry Biosecurity Authority, personal correspondence, 2001.
25. Budd, K. and Arts, Anne-Marie, *Review of Biodiversity Influences in the Last Decade*, Report to the Parliamentary Commissioner for the Environment, Wellington, 2000.
26. McMichael, A., *Planetary Overload*, Cambridge University Press, Cambridge, 1993.
27. Low, T., *Feral Futur*e, Viking, Ringwood, 1999.

section seven

South Africa

chapter fourteen

*The economic consequences of alien plant invasions: examples of impacts and approaches to sustainable management in South Africa**

B.W. van Wilgen, D.M. Richardson, D.C. Le Maitre, C. Marais, and D. Magadlela

Contents

Abstract

The invasion of natural ecosystems by alien plants is a serious environmental problem that threatens the sustainable use of benefits derived from such ecosytems. Most past studies in this field have focused on the history, ecology, and management of invasive alien species, and little work has been done on the economic aspects and consequences of invasions. This chapter reviews what is known of the economic consequences of alien plant invasions in South Africa. These economic arguments have been used to successfully launch the largest environmental management program in Africa.

* This chapter appeared in *Environment, Development, and Sustainability*, 3:145–168, 2001. It is reprinted here with permission of Kluwer Academic Publishers.

Ten million ha of South Africa have been invaded by 180 alien species, but their impacts are not fully understood, although they are undoubtedly significant. The indications are that the total cost of these impacts is substantial. Selected studies show that invasions have reduced the value of fynbos ecosystems by more than $11.75 billion (U.S); that the total cost of invasion would be about $3.2 billion on the Agalhas Plain alone; that the net present cost of invasion by black wattles amounts to $1.4 billion; that invasions by red water fern have cost $58 million; and that the cost to clear the alien plant invasions in South Africa is around $1.2 billion. These few examples indicate that the economic consequences of invasions are huge.

One of the unique aspects of invasive plant control programs in South Africa has been the ability to leverage further benefits (mainly through employment) for the expensive control programs from the government's poverty relief budget. This has made it possible to allocate substantial funding to a program that would otherwise have struggled to obtain significant support. Biological control of invasive species also offers considerable benefits, but is often the subject of some debate. We believe that, at least in the case of many invasive alien plant species in South Africa, biological control offers one of the best and most cost-effective interventions for addressing the problem.

14.1 Introduction

In South Africa, thousands of plant species from other parts of the world have been introduced for a range of purposes — as crop species, for timber and firewood, as garden ornamentals, for stabilizing sand dunes, and as barrier and hedge plants. Many of these alien species have become naturalized, surviving in the South African landscape without needing to be tended, and some of these naturalized species have become invasive. Invasive alien species are able to survive, reproduce, and spread, unaided and sometimes at alarming rates, across the landscape. The invasion of newly colonized areas by alien organisms is a global problem of significant and growing proportions,[1] and it can have serious implications for the environment.

In southern Africa, much of the historic concern about invasives has centered on the consequences for the conservation of biodiversity. Southern Africa is an area of remarkable biological diversity. For example, the region is home to some 21,137 species of vascular plants, about 80% of them endemic.[2] Many parts of the region have been affected by alien plant invasions. South Africa has a long colonial history, dating back 350 years, and it also has a well-developed infrastructure, with thriving agricultural and forestry sectors. These factors have contributed significantly to the introduction, establishment, and spread of invasive alien plants.[3] About 750 tree species and around 8000 shrubby, succulent, and herbaceous species have been introduced to South Africa; of these, 161 species (38 herbaceous, 13 succulent, and 110 woody) are regarded as seriously invasive,[4] although many more will become weeds in the future.

Biological invasions have received much attention from South African scientists and land managers in the past (see Macdonald et al.[5] and Richardson et al.[6,7] for recent reviews). However, most attention has been given to the history, ecology, and management of invasive alien species, and little work has been done on the economic aspects and consequences of invasions. The situation is changing as people begin to realize the wide-ranging consequences of invasions. An important breakthrough was the demonstration of the current and potential impacts of invasive alien trees and shrubs on water resources in South Africa.[8–10] This work demonstrated the economic benefits of intervention, and led to the establishment of the Working for Water program, which is aimed at the control of invasive alien plants to protect water resources and ensure the security of water supplies.[11]

The South African government spent more than $100 million on this program between its inception in 1995 and April 1, 2000.

In this chapter we review what is known of the economic consequences of alien plant invasions. Although studies in this field are still in their infancy, there have nonetheless been some significant advances that provide new perspectives on the problem. A review would seem timely, given that significant funds are being invested in the problem. In addition, we review the unique solutions that South Africa has pioneered with respect to dealing with the problem, and we provide a short overview of the economic benefits and sustainability of such approaches.

14.2 Extent of the problem

Several estimates have been made of the spatial extent of alien plant invasions in South Africa. Unfortunately, each study used a different method or concentrated on particular species or areas. For these reasons, and because surveys were done at different times, these results cannot easily be merged to produce a national overview. Richardson et al.[7] reviewed available data on the extent of alien plant invasions in different parts of South Africa. The most comprehensive set of records is the South African Plant Invaders Atlas[12,13]; these data can be summarized as frequencies by quarter-degree (latitude and longitude), but they cannot easily be converted to estimates of the extent of the invaded area. The same applies to the recent farmer surveys for the national desertification audit.[14]

This gap has been partly filled, at least for woody shrub and tree species, by a rapid reconnaissance of the extent of invasions in South Africa undertaken in 1996–97.[15,16] The emphasis of this assessment was on mapping species believed to use more water than native vegetation, so succulent (e.g., Cactaceae), herbaceous (e.g., grasses, annuals), and aquatic invaders were generally excluded. The areas of commercial plantations of the invading species, and invasions in the major urban and metropolitan areas, were also excluded. The data from this survey need to be interpreted with caution. This mapping exercise was aimed at providing a broad-brush estimate of the extent of invasions, and various data sets from other sources were also included in the final database. The estimates derived from this survey have elicited considerable discussion, and opinions on the accuracy of the data are strongly divided, but they are the only national assessment available at present.

According to this national survey, about 10 million ha of South Africa have been invaded by the approximately 180 species that were mapped (Table 14.1). Of South Africa's nine provinces, the Western Cape has the most extensive invasions, followed by the Northern and Mpumalanga provinces. KwaZulu-Natal and the Eastern Cape were not adequately mapped, and the true extent of invasion in these provinces is likely to be closer to the percentage for Mpumalanga, which has similar climate, vegetation, history of colonization, and land-use patterns.[16] Most of the invasions are concentrated in the wetter regions of the country, and this is reflected in the number of species that have been recorded per quarter-degree square. The greatest number of species occurs in the Western Cape and along the eastern escarpment from KwaZulu-Natal through to the Northern Province (Figure 14.1).

The data from the national survey were summarized by province, and not by the country's major biomes. However, it is possible to get indications from the distribution of the invasions and the representation of the biomes in the provinces. The fynbos (a Mediterranean-type shrubland) biome is the best studied, and the most invaded, with extensive dryland invasions in both the mountains and the lowlands, as well as invasions along all the major river systems.[7,16,17] Most of the fynbos biome is located in the Western Cape, but the small portion of this biome in the Eastern Cape (10,300 km^2, or 6% of the

Table 14.1 Areas Invaded by Alien Plants in the Seven Provinces of South Africa, and the Mean Canopy Cover[16]

Province	Major biomes (% of province)	Area (km²)	Total area invaded (km²)	Total area invaded (%)	Mean canopy cover (%)
Eastern Cape	Grassland (40), Nama Karoo (25), thicket (16)	167,398	6720	4.01	22.51
Free State	Grassland (72), Nama Karoo (22), savanna (6)	129,936	1661	1.28	14.56
Gauteng	Grassland (78), savanna (22)	16,519	223	1.35	58.56
KwaZulu-Natal	Savanna (54), grassland (36), thicket (8)	94,596	9220	9.75	27.21
Mpumalanga	Grassland (64), savanna (36)	79,571	12,778	16.06	14.49
Northern Cape	Nama Karoo (54), savanna (30), succulent Karoo (14)	361,981	11,784	3.26	14.10
Limpopo	Savanna (97), grassland (3)	122,143	17,028	13.94	15.45
North West	Savanna (71), grassland (29)	116,010	4052	3.49	13.88
Western Cape	Fynbos (47), Nama Karoo (24), succulent Karoo (24)	129,314	37,274	28.82	16.80
South Africa (including Lesotho)		1,217,467	100,739	8.07	17.23

Data on biomes from Low, A.B. and Rebelo, A.G., V*egetation of South Africa*, Lesotho and Swaziland. Department of Environmental Affairs and Tourism, Pretoria, 1996.

province) is also heavily invaded. The major invaders are *Acacia*, *Hakea*, and *Pinus* species. At the scale of the whole Cape Floristic Region (comprising mainly fynbos vegetation types, but also parts of the succulent Karoo, Nama Karoo, thicket, and forest biomes), the data from two independent assessments of the extent of dense stands are reasonably similar (Table 14. 2), but Versfeld et al.[15] clearly underestimated the extent of the light invasions, and overestimated the extent of the medium invasions, on the Agulhas Plain. Nevertheless, these studies all highlight the extensive invasions in the fynbos biome.

The forest biome has been heavily invaded, but the extent cannot be quantified at present.[7] The grassland and savanna biomes have also been extensively invaded, mainly by acacias, other tree species, and a variety of woody scramblers (e.g., brambles). The worst-affected areas are the grasslands of the Drakensberg escarpment and the moister regions of the savanna biome along the lower escarpment, and in the KwaZulu-Natal midlands and coastal belt. Most of the invasions in these biomes are found along the banks and in the beds of the rivers; there are few, if any, river systems that have not been extensively invaded. Invading trees such as syringa (*Melia azedarach*) and jacaranda (*Jacaranda mimosifolia*) have spread into semiarid savanna by invading along perennial rivers, where the freely available water allows them to survive the seasonal drought. The Nama Karoo (semi-desert shrubland, summer rainfall) is probably the fourth most heavily invaded biome; woody invaders, notably mesquite trees (*Prosopis* spp.), have invaded at least 18,000 km^2 of the low-lying alluvial plains and the seasonal and ephemeral watercourses. Several cacti (*Opuntia* spp.) and saltbushes (*Atriplex* spp.) have invaded large areas of the Nama Karoo and succulent Karoo (winter rainfall) biomes[18] and the thicket biome in the Eastern Cape.[7]

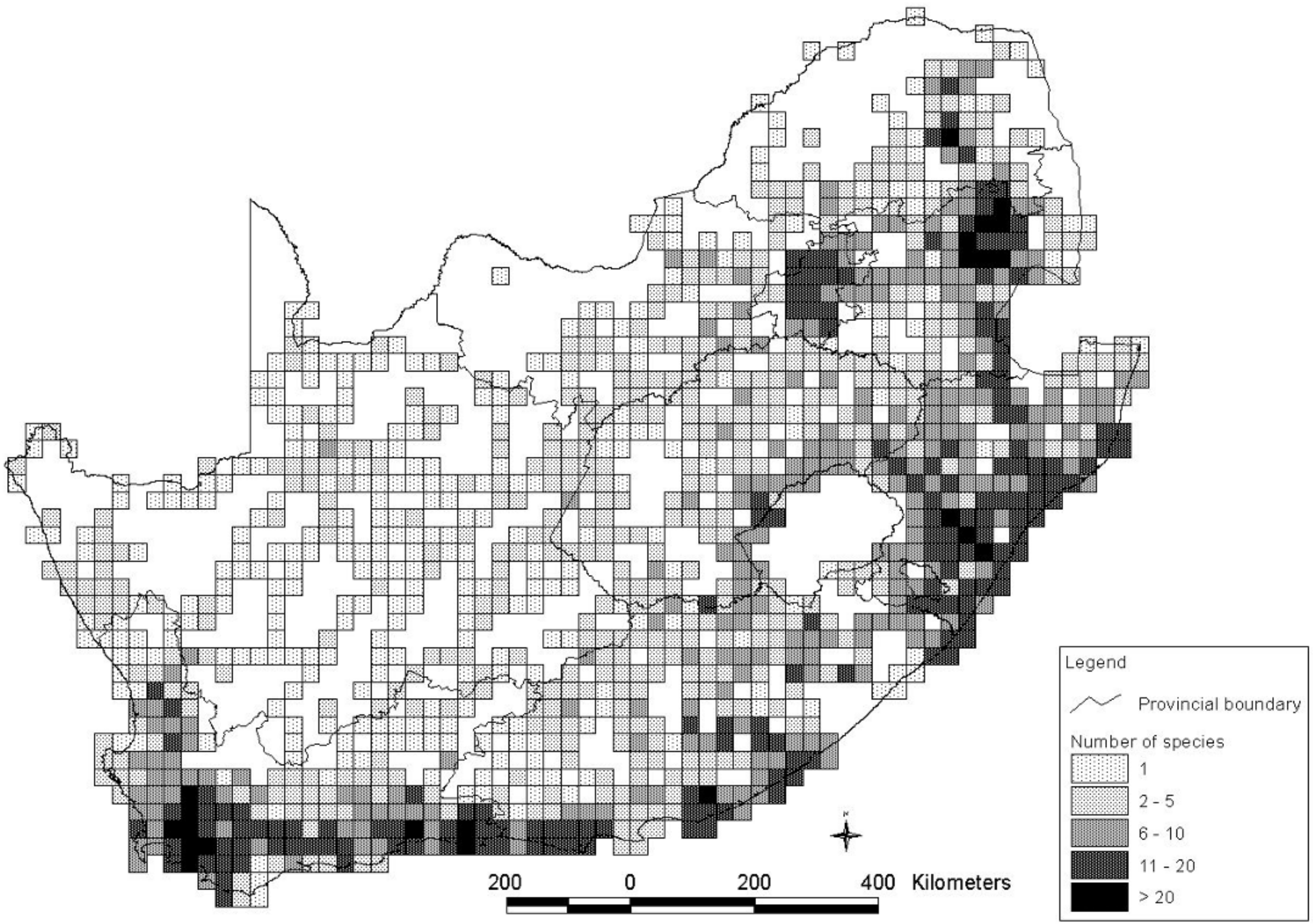

Figure 14.1 Map of South Africa showing number of invasive alien plant species recorded per quarter-degree square. (Data from Henderson, L., South African plant invaders atlas (SAPIA), *Applied Plant Science*, 12, 31, 1998.

Table 14.2 A Comparison of the Extent of Invasions in the Cape Floristic Region (CFR) and Agulhas Plain (areas are in km²)

Area	Source	Total study area	Density class (canopy cover)		
			Light (<25%)	Medium (25–75%)	Dense (>75%)
Cape Floristic Region	15	94393	29575	6968	1226
	17	87892	—	—	1394
Agulhas Plain	17	2377	22	1794	107
	58	2161	1081	167	302

Table uses data from Versfeld et al.[15] for selected catchment areas that fall mainly within the boundaries of the CFR or the Agulhas Plain, respectively. Summary statistics for the light and medium invasions for the CFR were not given by Cowling et al.[17]

Several aquatic weeds have spread over large areas in South Africa, notably water hyacinth (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), Kariba weed (*Salvinia molesta*), parrot's feather (*Myriophyllum aquaticum*), and red water fern (*Azolla filiculoides*). These species, in the absence of natural enemies and in the presence of eutrophic waters, form large, dense mats that degrade aquatic ecosystems and impact on all aspects of water utilization. Four of the five species are effectively under biological control, and the only real problem species is water hyacinth, for which biological control has not been developed to its full potential. Water hyacinth is widespread throughout South Africa and severely impacts rivers in the Western and Eastern Cape, KwaZulu-Natal, and Mpumulanga provinces, and on the Vaal River in the Gauteng and Free State provinces.

14.3 Environmental impacts

The development of an understanding of environmental impacts of invasive alien plants would be extremely useful for the quantification of economic impacts. Unfortunately, no standard system exists for the objective quantification of the many and varied environmental impacts of invasive alien plants worldwide.[19] As in other parts of the world, impacts of plant invasions in South Africa have been measured in numerous ways, making comparisons between biomes within this region, or with other regions or countries, difficult. Many descriptions of impacts are anecdotal or correlative (comparing invaded sites with uninvaded sites, or comparing one site at different times), or are based on the performance of the invader in other parts of the world. Very few detailed studies, and no manipulative experiments, have been done to determine the magnitude, mechanisms of impacts, and implications of impacts of invasions of alien plants in South Africa. Nevertheless, we can draw some conclusions as to the types and magnitude of impacts caused by the most important plant invaders.

Table 14.3 gives examples of the main types of impacts caused by the most widespread and damaging invaders. The types of impacts include effects on individuals (including genetics), effects on the population dynamics of native species, effects on community dynamics (species richness or diversity, trophic structure), and effects on ecosystem processes and functioning.

When considering the impacts of invasive plants, it is useful to consider two categories of invaders, as proposed by Chapin et al.[20] (see also Dukes and Mooney[21]): *discrete-trait invaders* (DTIs) and *continuous-trait invaders* (CTIs). DTIs add a new function, such as nitrogen fixation, to the invaded ecosystem, whereas CTIs differ from natives only in traits, such as litter quality or growth rates, that are distributed continuously among species. Results from several parts of the world show that DTIs generally have greater ecosystem-

Table 14.3 Examples of Environmental Effects of Invasive Alien Plants in South Africa

Invader (life form)	Biomes affected	Disruption	Evidence	References
Acacia cyclops (T)	fy, fo, sa	Δ in seed dispersal dynamics	O	59
		Δ in coastal sediment dynamics	OD	60,61
		Provides nesting habitat for rare African penguins	O	R.J.M. Crawford, unpublished data
		Outcompetes native plants	OD	62
		↑ biomass	OD	63
		↑ litterfall	OD	64
Acacia longifolia (T)	fy, sa	↓ diversity of ground-living invertebrates	OD	65
		↓ stream flow	OD	25
Acacia mearnsii (T)	gr, fy, fo, sa	↓ diversity of ground-dwelling invertebrates	OD	65
		↓ stream flow	OD	25
		Destabilization of stream banks; ↑ erosion	O	7
Acacia saligna (T)	fy, fo, sa	↑ biomass	OD	63
		↑ litterfall	OD	64
		Δ nutrient chemistry in lowland fynbos	OD	26–28,30
		Δ in seed dispersal dynamics	OD	61
		↑ biomass; Δ size and distribution of fuel; ↓ moisture content = Δ fire regime	OD	39
		Attrition of seed banks of native plants over time in dense stands	OD	31,32
Caesalpinia decapetala (CR)	sa, gr, fo	Dense mats cause trees to collapse	O	66

Table 14.3 (Continued) Examples of Environmental Effects of Invasive Alien Plants in South Africa

Invader (life form)	Biomes affected	Disruption	Evidence	References
Chromolaena odorata (SW)	fo, sa	↑ flammability in forest and riverine woodland; forms "ladders" that carry fires into crowns of fire-sensitive trees ↓ biodiversity of ecotones	O	67
Eichhornia crassipes (FFM)	aq	↓ degrades aquatic ecosystems Δ river flows	O	68
Eucalyptus spp. *(T)*	fy, gr, fy, sa, fo	↑ water repellency and soil erosion	OD	40
Hakea sericea (SW)	fy, fo	↑ biomass; Δ in size and distribution of fuel; ↓ moisture content = Δ fire regime;	OD	39
		↑ biomass; results in very intense fires when felled plants are burned (water-repellent soils lead to erosion)	OD	69,70
		Dense stands limit options for fire management	O	71
		Δ vegetation structure leads to ↓ in abundance and diversity of native birds; Δ in arthropod community structure (some taxa ↑; some ↓); ↓ leaf retention and % of seed set in native Proteaceae	OD	72

Lantana camara (SW)	fo, sa, fy, gr	↓ diversity of ground-dwelling invertebrates	OD	63
		↓ suppresses regeneration via allelopathy	OO	73
		poisons livestock (R1.7 million per year)	OD	73
Melia azederach (T)	fo, sa	Outcompetes native plants	O	67
		Δ feeding dynamics of frugivorous birds	O	67
Myriophyllum aqauaticum (FAM)	aq	↓ degrades aquatic ecosystems	O	74
		Δ river flows		
		↑ mosquitoes and diseases		
Nasella trichotoma (AG)	gr	↓ pasture productivity	O	75
Opuntia aurantiaca (SS)	ka, sa	↓ pasture productivity	O	75
Opuntia ficus-indica (SS)	sa, ka, gr, fo, fy	↓ pasture productivity	O	75
Pereskia aculeata (CR)	fo	Dense mats cause trees to collapse	O	66
Pinus pinaster (T)	fy, fo, sa	Outcompetes native plants	OD	62,76
		Dense stands limit options for fire management	O	71
		↓ stream flow	OD	8,35
Pinus radiata (T)	fy	↓ stream flow	OD	24
Prosopis spp. (T/SW)	sa	↑biomass; Δ in vegetation structure; ↓ access, ↓ pasture productivity	O	77
		Outcompetes native plants	O	78
		↓diversity of dung beetle assemblages	OD	79

Table 14.3 (Continued) Examples of Environmental Effects of Invasive Alien Plants in South Africa

Invader (life form)	Biomes affected	Disruption	Evidence	References
Psidium guajava (T)	fo, sa, gr	Outcompetes native plants	O	67
Rubus spp. (SW)	sa, gr, fo	Hybridizes with native *Rubus* sp.	O	80
Salix babylonica (T)	ka, gr	Destabilizes river banks and excludes native plants	O	81
Salvinia molesta (FFM)	aq	↓ degrades aquatic ecosystems Δ river flows ↑ mosquitoes and diseases	O	82
Sesbania punicea (T)	sa, fo, fy, gr	↓ access, ↑ bank erosion, ↓ stream flow	O	83
		Poisoning of stock	O	75
Solanum mauritianum (T)	sa, fo, gr	↓ diversity of ground-dwelling invertebrates	OD	65
		Δ feeding ecology of Rameron pigeon and other native birds	OD	84
		Outcompetes native plants	O	67

Key:

Life forms: T = tree; SW = shrub (woody); SS = shrub (succulent); V = vine; AG = annual grass; CR = creeper; PG = perennial grass; AF = annual forb; PF = perennial forb; FFM = free-floating macrophyte; FAM = floating macrophyte (attached).

Biomes affected: fo = forest; fy = fynbos; gr = grassland; sa = savanna; ka = karoo[7]

Disruption: Δ = change; ↑ = increase; ↓ = decrease.

Evidence: OD = observation with data; O = observation without data; OO = observation in another part of the world.

level impacts than CTIs, although the latter can also bring about such changes, especially if they make up a large proportion of an ecosystem's biomass.

The most dramatic impacts of alien plants in South African systems have clearly been from DTIs. For example, the Australian *Acacia* species (notably *A. cyclops* and *A. saligna*) have radically altered nutrient-cycling regimes in the nutrient-poor systems of lowland fynbos, due to their ability to fix atmospheric nitrogen (no widespread or abundant native species performs this role). These species and several species of *Pinus* that have invaded large areas have added a major new life form (trees) to the tree-poor fynbos. Such invasions have produced many ecosystem-level changes by altering features such as biomass distribution, plant density and vegetation height, leaf-area index, litterfall, and decomposition rates. These invasions are the greatest threat to endangered plant species in the Cape Floristic Region.[22] Widespread tree and shrub invasions in vegetation types that previously had a low tree cover have radically altered habitats for animals. This is shown, for example, by the major changes in distributions of many bird species, including species that invaded the southwestern parts of South Africa from adjacent biomes in response to increased tree cover (e.g., Macdonald[23]). Other DTIs that have caused major impacts in South African ecosystems include *Chromolaena odorata*, *Pereskia aculeata*, *Prosopis* spp., and the suite of alien trees that have invaded watercourses in formerly tree-poor systems.

The most detailed work on assessing the impact of plant invasions in South Africa has focused on tree invasions in the fynbos biome. Several studies have documented the reduction in stream flow caused by plantations of invasive tree species (mainly *Pinus* species[8,24,25]), as well as aspects of the modified nutrient regime[26–30] and impacts on the seed banks of native fynbos species associated with *Acacia saligna* invasions.[31,32] A recent country-wide study made a first attempt at assessing the effect of all woody invaders on surface-water resources. Results from this investigation suggest that woody alien plants may be using as much as 6.7% of the total mean annual surface runoff, or 9.95% of the utilizable surface runoff in South Africa.[15,16]

The above brief review shows that the current understanding of the environmental impacts of invasive alien plants is scattered and patchy. No overall framework for the collection and synthesis of the relevant information has been attempted, and as a result there are few overall syntheses of impacts at the scale of an ecosystem. Nonetheless, the studies that have been done so far could provide a useful starting point for the development of such a framework.

14.4 The economics of invasions

14.4.1 Cost-benefit analyses

Invasions by alien plants in South Africa, as elsewhere in the world, result in many negative impacts. In addition, some invasive plants also have major positive benefits, and these need to be taken into account when assessing the costs resulting from invasions. These impacts and benefits could be expressed in economic terms, although very few studies have attempted to do this until recently. Attempting an objective analysis and summary of the studies that have been done is frustrating, as every study has used a different approach, making an accurate assessment of aggregate impacts impossible. Despite this, several studies have been carried out (Table 14.4), and these can be used to illustrate the known impacts, and to illustrate trends.

Most early assessments of the impacts of alien plants in South Africa concentrated on the environmental impacts, particularly the impacts on biodiversity. It was not until the mid-1990s that scientists began to quantify the impacts of invasion by alien trees and

Table 14.4 Studies on the Economic Impacts of Alien Invasive Plants in South African Ecosystems

Study	Type of impacts or benefits	Value of impacts or benefits	Sources
Impact of alien plant invasions on water yield.	Reduced water yield from invaded watersheds; prevention of water losses through clearing programs.	Clearing of invasive aliens would prevent losses of 30% of Cape Town's water supply at a cost of 1.3 c/m^3.	8.
Costs and benefits of alien plant clearing programs.	Increased water yields resulting from clearing, compared to costs of operating water supply schemes.	Clearing would yield water at 14% of the cost of developing new water supply schemes (1.1 vs. 8.4 c/m^3 respectively).	9,10
Costs and benefits of alien plant clearing programs.	Increased water yields from clearing of invaded watersheds and uneconomic plantations of alien trees.	Clearing yields benefit:cost ratios of between 6:1 and 12:1 for clearing invaded watersheds, and between 360:1 and 382:1 for clearing uneconomic plantations.	34
Relative costs of clearing programs compared to costs of allowing alien plants to invade unchecked.	Increased water resulting from clearing (between 7 and 22% of current runoff), compared to increased losses from continued invasion (from 22 to over 100% if invasions continue unchecked).	Estimated cost of clearing invaded watersheds ranged from US$ 4 to 13 million, and would increase to between US$11 and 278 million if invasions continue unchecked. Value of lost water was not quantified.	35
Broad survey of impacts of alien plants on water resources at a national scale.	Invasions estimated to be using 3300 million m^3 per year (6.7% of the runoff of South Africa).	Value of lost water was not quantified, but estimates of cost to clear infestations range from 412 to 996 million US$.	15,16
Ecological–economic simulation model of mountain fynbos ecosystems	Model assesses consequences of invasion for water production, wildflower harvest, hiker and ecotourist visitation, endemic species, and genetic storage.	Managing alien plants increases value of hypothetical 4 km^2 area from 3 to 50 million US$. This can be achieved by spending a fraction of total value on clearing programs.	47
Cost-benefit analysis of black wattle (*Acacia mearnsii*).	Benefits assessed from commercial crop values and other products; impacts from reduced water yield, increased fire risk, and loss of biodiversity.	Continued cultivation without control programs yields a benefit:cost ratio of 0.4. Continued cultivation with clearing, or with clearing and biological control of seeds yields benefit:cost ratios between 2.4 and 7.5.	36

Cost-benefit analysis of biological control of red water fern (*Azolla filiculoides*).	Serious economic impacts on agricultural sector through loss of water resources and livestock. Total losses estimated at US$58 million in South Africa.	Introduction of a biological control agent has brought the problem under control at a cost of US $ 51,000, yielding a benefit:cost ratio of 1130:1.	M.P. Hill (pers. comm.)
Economic valuation of indigenous vegetation and impacts of alien infestations on the Agulhas Plain.	Benefits from livestock, wildflower harvesting and nature-based tourism. Cost estimates for clearing invasives to prevent erosion of benefits.	Benefits total US$ 3 million annually. Total cost to clear invasives amounts to US$ 5.6 million. Thus, clearing will yield positive benefits after two years.	85
Analysis of the economic consequences of invasion on the use values of fynbos (shrubland) ecosystems.	Values of harvesting of wildflowers, recreational use in protected areas, and of water runoff, in pristine and densely invaded areas respectively.	Reductions due to invasion range from US$ 9.7 to $2.3/ha for harvest values and from $8.3 to $1/ha for recreational use. The value of lost water due to invasion amounted to $163/ha.	37
Cost of controlling jointed cactus (*Opuntia aurantiaca*).	State introduced a subsidized scheme of supplying herbicides to farmers whose land became invaded.	Costs of state-subsidized control efforts exceeded US$12 million over the past 40 years. Cost has dropped by 83.5% following the introduction of a biological control agent.	H.G. Zimmermann (pers. comm.).

Monetary values are in U.S. dollars; where values were published in Rand, we have converted to US$ at a rate of R7 = 1 US$.

shrubs on water resources in South Africa.[8,9,33] These studies were pivotal in persuading the government to take the issue very seriously, while at the same time providing the impetus for further studies on the economics of plant invasions.

The estimates of the impacts of invading alien trees and shrubs on the yield of water from important catchment areas were based on experiments where pines and eucalypts were planted in these areas to determine the effects of plantation forestry on water resources. The results have been extrapolated to other areas, and to estimate water use by other species (Table 14.4). For these reasons, the results must be viewed with caution. However, even if the estimated losses are twice the real losses, the "costs" would still be significant. The studies listed in Table 14.4 have shown that invasive alien plants may be using as much as 6.7% of the country's runoff[15]; that clearing the invasive plants is a good investment simply to prevent water losses[10,34]; and that failure to clear stands of invading trees will result in exponential increases in the costs of clearing as catchment areas become further invaded.[35]

The economic consequences of "lost" water would be even more important. Because water is a limiting resource in South Africa, losses of water will restrict the country's economic growth potential. At least one study[36] has sought to carefully quantify the impacts of lost water for urban, agricultural, and industrial uses, and the results showed that the cost of a clearing program can easily be justified in terms of the economic benefits derived from preventing water losses or restoring water resources to pristine levels.

The significant impacts of invasive alien plants on biodiversity are difficult to evaluate in monetary terms, so this is often not attempted. For example, De Wit et al.'s study[36] on the economic impacts of *Acacia mearnsii* invasions did not take biodiversity into account because of the lack of data or estimates on which to base such evaluations. The economic benefits of biodiversity can nonetheless be significant, both from direct use (such as grazing, or harvesting of natural products), as well as from indirect use (such as recreation and nature-based tourism). Some recent studies have included aspects of the value of biodiversity in their analyses (for example, Turpie and Heydenrych[37] and Higgins et al.[38]; see Table 14.4); these show that invasions have significant economic costs as a result of the impacts on biodiversity. Turpie and Heydenrych[37] estimated the values of the harvesting of wildflowers, and for recreational use in protected areas, and showed that harvest values were reduced from \$9.70 to \$2.30/ha, and recreational use values in protected areas were reduced from \$8.30 to \$1.00/ha, when pristine areas became densely invaded by alien plants.

Invasions by alien plants increase the negative impacts of fires, by increasing fuel loads and fire intensities,[39] thereby making fires more difficult to control and increasing the risk of damage. The more intense fires also cause severe damage to soils,[40] leading to soil loss,[41] severe soil erosion during rainstorms, and damage to infrastructure due to flooding. The economic impacts can be severe, but are difficult to demonstrate (for example, how much worse was a fire because the area was invaded?).

Examples of the costs associated with fires include a wildfire on the Cape Peninsula in March 1999 that created water-repellent conditions in an invaded area that formerly had no overland flow.[42] Flooding followed heavy rains in April 1999 and resulted in cleanup costs of more than \$150,000. This estimate excludes the associated flood damage to 30 dwellings, which probably totaled at least another \$150,000. These impacts did not occur in adjacent areas that were not invaded.

In another example, two wildfires burned 8000 ha on the Cape Peninsula between Jan. 16 and Jan. 20, 2000.[40] Insurance claims arising from this fire amounted to \$5.7 million.[43] Most houses and structures that were damaged were in areas invaded by alien plants, where fire intensities were much higher than in adjacent uninvaded areas. The direct costs of fighting the fire were not documented, but they exceeded \$500,000.[43] After the fires, an average of 147 tons/ha of soil was lost from alien-invaded areas, compared with negligible losses from areas with natural vegetation.[41] The increases in fire intensity also have important negative effects on biodiversity, but these have been poorly studied.

The above negative impacts should not be presented in a one-sided manner, however. Almost all the important crops in South Africa are harvested from alien plants, and the point needs to be made that only a relatively small percentage of these alien plants become invasive. In addition, some invasive alien species have considerable value, despite their negative impacts, and this needs to be taken into account when assessing the costs resulting from invasions. Conflicts of interest arise from time to time in cases where important commercial species become invasive and spread beyond the areas where they are cultivated. These include plantation forestry (*Pinus* sp.[44]), where alien plants provide firewood (many *Acacia* sp.[37]), food (*Opuntia* sp.[45]), fodder (*Prosopis* sp.), or nectar for bees (*Eucalyptus* sp.[46]); and where they have aesthetic or utilitarian value (ornamentals, shade trees, or windbreaks).

A good example is provided by plantation forestry, which is an important part of the South African economy, contributing $300 million, or 2%, to the GDP and employing more than 100,000 people. Downstream industries based on forestry produce timber products worth a further $1.6 billion, much of which is exported and earns valuable foreign exchange. Clearly, these activities are significant. However, a large proportion (38%) of the area invaded by woody alien plants in South Africa is occupied by species used in commercial forestry (especially *Pinus* and *Acacia* sp.). It is thus clear that forestry has been one of the country's major sources of alien infestation.[3,44]

These conflicts have to be dealt with in a sensitive manner if progress is to be made in reducing the significant negative impacts of invading alien plants. Some of the possible approaches for avoiding conflict in South Africa include recognizing the value of a vibrant forest industry and actively managing the spread of plantation trees; making allowance for well-managed woodlots in areas where fuelwood is scarce; using non-invasive species wherever possible, or ensuring that biological control is introduced at the start of new agroforestry projects; using biological control to reduce the invasive potential of otherwise useful species without killing them (for example, by reducing the number of seeds they produce); recognizing potential invaders early and taking precautionary measures; educating people as to the dangers and costs of invasive species; and encouraging the use of alternative, non-invasive species for ornamental and utilitarian purposes.

Arriving at a comprehensive figure for the total costs of invasive plants is not possible at this stage. However, the indications are that the total costs are substantial, and a number of studies can be quoted to support this contention. Some examples are listed below.

1. One of the few detailed studies calculated the value of a hypothetical 4-km^2 (4000-ha) mountain fynbos ecosystem at between $3 million (with no management of alien plants), and $50 million (with effective management of alien plants), based on six components: water production, wildflower harvest, hiker visitation, ecotourist visitation, endemic species, and genetic storage.[47] Given that there are more than 1 million ha of protected fynbos areas in South Africa, the potential reduction in value due to invasion could amount to more than $11.75 billion.
2. Turpie and Heydenrych[37] estimated that the value of lost water amounts to $163/ha on the Agulhas Plain area of South Africa; thus, if 20,000 ha of this area became invaded (20,000 ha is the target area to be incorporated in the proposed Agulhas National Park), the total cost would be in the region of $3.2 billion.
3. In a study on black wattle (*Acacia mearnsii*) invasions, De Wit et al.[36] calculated the economic value of stream flow lost to invasions of black wattle in South Africa using the opportunity-cost approach. First, the value added by water over the

different demand sectors (irrigation, domestic and urban use, mining and industry, the environment, and afforestation) was calculated. Second, the value added by additional water where black wattle was eradicated was estimated. These estimates were adjusted to allow for evaporation and spillage of flood water (33% of additional water was assumed to be unusable), changes in the numbers of downstream water users over the next 20 years, and the degree to which water would contribute to the economic value added in each sector (assumed to be 10% of predicted growth in economic value added). This study revealed a net present cost of $1.4 billion attributed to black wattle invasions. (It should be noted that this study considered only black wattles, and not the many other invasive trees in the country.)

4. In the only detailed study to date on the economic benefits of biological control of invasive alien plants in South Africa, scientists developing biological control solutions have shown that bringing the red water fern (*Azolla filiculoides*) under control has yielded a return on investment of 1130:1.[47a] Red water ferns, introduced from South America, rapidly covered dams and resulted in damage to water pumps, the deaths of livestock, and substantial clearing costs that totalled $58 million. This was compared with a cost of $51,000 to carry out the research that led to the release of the biological control agent, which in turn brought the problem completely under control within 2 years of release.
5. The cost to clear the alien plant invasions in South Africa is estimated to be around $1.2 billion, or roughly $60 million per year for the estimated 20 years that it will take to deal with the problem.[15] This expense is needed to offset the considerable costs due to invasive plants, but the point needs to be made that, should the program not be funded, the costs will grow as invasive plants spread to occupy the full extent of invasible habitats. The country is therefore forced to incur these expenses or face the even worse prospect of increasing impacts.

These few examples indicate that the economic consequences of invasions are huge. The examples cited above represent only portions of the problem for which studies have been done; a thorough analysis would undoubtedly reveal much higher costs.

14.4.2 Dealing with the problem

14.4.2.1 Using the alien plant problem to create benefits

While invasions by alien plants have significant negative environmental and economic impacts, the South African government has used the opportunities offered by the need for labor-intensive clearing programs to generate a range of benefits. By adding these benefits to the obvious environmental and economic advantages, it has been possible to justify the spending of more than $100 million on the program between 1995 and 2000. The funds have been directed through the government's Working for Water program, which engages unemployed people in labor-intensive clearing, follow-up, and rehabilitation projects aimed at bringing invasions of alien plants under control. The program also runs a parallel social development effort that seeks to maximize the opportunities for development of disadvantaged people employed by Working for Water. The social development program has several main components, including: (a) a child-care program allowing women to earn much-needed income to support their families; (b) an HIV/AIDS program, involved HIV/AIDS awareness campaigns and condom distribution among workers; and (c) partnerships that include an ex-offender reintegration program and several others. Working for Water employs the poorest members of communities settled closest to the alien-infested areas. The trend has been to target women, especially single mothers, and encourage them to join teams of about 20 members with supervisors to oversee their productivity.

In early 2000 the Working for Water program employed about 20,000 people, largely from rural areas, in 249 projects around the country. The main social benefits from clearing activities include not only employment, but also the improvement of poor people's livelihoods through increased income, resulting in better nutrition for children, better clothing, and the ability to pay for education. Additionally, the social benefits come with a contribution toward economic empowerment through small-business training and skills development for contractors, leading to better opportunities to earn a living outside the program.

In one of the few detailed studies of the social impacts of the program, Marais[48] quantified benefits in the Western Cape Province during 1996–97. Wages were paid to 2961 people employed as clearing workers, team supervisors, managers, and development, training, and administration officers. The total spent on salaries for the year was \$6.431 million, a mean of \$2175 per person. Most of this money was invested directly into disadvantaged rural communities. Assuming an average family size of five people, and that only one person per family was employed in the project, the direct benefits reached as many as 14,800 people, at \$435 per capita. This injection of funding into disadvantaged communities also had secondary effects in terms of suppliers and service providers. Protective clothing, tools, and mechanical equipment constituted most of the supplies bought by the project, and a number of secondary jobs were linked to the project through procurement.

The project was evaluated in the context of the provincial economy, using a social accounting matrix (SAM) analysis.[49] Since comparatively wealthy taxpayers (the source of the funding) and the workers have vastly different spending habits, the project should be redistributive in nature. Net multiplier effects should show the influence of some contraction in economic activity by taxpayers and an expansion in activity by those persons and businesses receiving project expenditures.

Marais[48] measured direct employment as the number of person-years of employment created from \$250,000 of additional final demand. The estimated number of secondary, indirect, and induced jobs created in industries related to the program was 8.93 jobs for every extra \$250,000 in a project's budget. Based on a 1996–97 expenditure in the Western Cape of \$10.05 million, 359 additional secondary jobs were created. Based on the above, employment benefits for South Africa were estimated using the number of direct jobs and the national expenditure during that year (Table 14.5).

According to Eckert et al.,[49] 67.7% of government revenue from household contributions was drawn from the previously advantaged communities in the Western Cape. At the same time, the percentage of total household spending on unprocessed and processed agricultural commodities by people from previously disadvantaged communities was 73.0% and 68.1%, respectively. This means that the bulk of household spending by these groups was being spent locally and benefited the local economy. On the other end of the scale, the rich were spending money on imports and other expensive items, which did not always benefit the local economy.

Eckert et al.[49] used Gini coefficients to estimate the amount of equality or inequality in the Western Cape province. They found that countries with highly unequal income distributions had Gini coefficients of between 0.5 and 0.7, while the value for countries with relatively equitable distributions was between 0.2 and 0.35.[50] The Gini coefficient for all Western Cape households was 0.509.[49] The total income of the highest income group, including the corporations and the upper- and middle-income groups in the province, declined over the study period by approximately 0.09%. On the other hand, the poorest of the poor showed an increase of 3.08% in their household income, and the Gini coefficient

Table 14.5 Annual Economic Benefits in the 1996/97 Financial Year Associated with the Employment of People in Alien Plant Clearing Programs

Measure of benefit	Western Cape	South Africa
Direct employment (number of people employed)	2961	8386
Approximate number of people supported by direct employment (assuming five to seven per family)	14,800–20,700	41,900–58,700
Indirect employment (number of secondary jobs created)	359	714
Approximate number of people supported by secondary jobs (assuming five to seven per family)	1800–2500	3600–5000
Approximate total number of people supported	16,600–23,200	45,500–63,700
Annual expenditure (millions of US $)	$10.05	$20

Data for the Western Cape are from Marais.[48] Data for South Africa are extrapolations based on Marais[48] and employment data for the Working for Water program at a national level.[86]

was reduced to 0.507.[49] If the size of the project in relation to the provincial GDP is taken into account, this was significant. The total household income in the province before the project started was $10,476,225. After the project it was $10,476,903. The net income change of participants in the program in the Western Cape was $677,750, or 6.74% of the total project expenses for 1996–97. It can be seen as significant that a project of such limited size should have an effect of this magnitude.

14.4.2.2 The economic advantages of biological control programs

The labor-intensive control programs aimed at clearing and rehabilitating the extensive invaded areas are unlikely to be sustained in the long term. The clearing programs should therefore aim to develop components that will ensure that cleared areas do not simply become reinvaded. One of the most cost-effective ways of doing this would be to use biological control, where species-specific organisms are introduced to bring an invasive species under control, or to reduce its invasive potential.

Biological control of invasive alien species is one of the most cost-effective ways of reducing the impacts of such invasions. South Africa has been very successful in finding effective biological control solutions to many invasive weed problems,[7,51] but the true value of these initiatives has not been quantified in any rigorous way. Historically, 103 biological control agents have been released in South Africa against 46 weed species; of these weed species, 22 are now under complete or substantial biological control.[51] The country hopes to build on these successes, and its biological control research program has been expanded to include some species (e.g., *Acacia* and *Pinus* spp.) that were previously excluded from research because of their commercial value.[52,53] In such cases, the approach would be to introduce control agents that reduce seed output (and therefore invasive potential) without affecting the growth potential of the plant. This has been done for several species already, notably Australian *Acacia* species. The total cost of the biological control research initiative between 1997 and 2000 was $3 million. Indications are that these activities represent unprecedented returns on investment.

The example of the economic benefits of bringing the red water fern under control showed a return on investment of 1130:1. Not all research projects will yield such dramatic results. Nonetheless, there are several examples in which biological control has led to significant reductions in costs. For example, the South African government has, for the past 40 years, provided subsidized herbicides to farmers whose land has become infested with jointed cactus (*Opuntia aurantiaca*). The costs of this program have exceeded $12 million over the past 40 years. The recent introduction of a successful biological control agent has seen the annual expenses incurred by government fall by 83.5%.

Another preliminary estimate has been made for the control of the Port Jackson willow (*Acacia saligna*), which has invaded more than 1.8 million ha in South Africa. The introduction of a biological control agent has effectively eliminated the need to proceed with expensive mechanical control programs, yielding a return on investment of $800 for every $1 invested in the research.[54]

The cost of bringing invasive alien trees and shrubs under control in South Africa is estimated to be around $1.2 billion, or roughly $60 million per year for the estimated 20 years that it will take to deal with the problem.[15] It was estimated that by introducing biological control as a factor, clearing costs over 20 years could be reduced to $400 million, or $20 million per year, a far more manageable target for a developing country like South Africa. Given that the clearing program is seen as essential for ensuring water and environmental security, such potential savings are substantial.

14.5 Conclusions

The above review has highlighted the need for a thorough economic assessment of the invasive alien plant problem in South Africa. Almost all of the studies carried out to date have indicated that the impacts of invasive alien plants are severe, and that intervention in the form of control programs is justified. The studies done to date are preliminary, however, and the conclusions of economic analyses are often subject to criticism because of the assumptions on which they are based. Given the indications of the severity of the problem, more studies will be needed in order to understand where the most important economic impacts will lie. Such studies will assist in designing optimal means for funding the control interventions.

One of the unique aspects of invasive plant control programs that has emerged in South Africa to date has been the ability to leverage further benefits from the expensive control programs. Most of the funds for the Working for Water program have been sourced from the government's poverty relief budget, and not just from budgets aimed at protecting water resources, agricultural land, and biodiversity. This leverage has made it possible to allocate substantial funding to a program that would otherwise have struggled to obtain significant support. The links are fragile, however, and the development of convincing economic assessments of the consequences of diverting funds from the clearing program into other necessary interventions, such as education and health care, will go a long way toward maintaining support.

Biological control of invasive species is one solution that appears to offer considerable benefits. Biological control cannot solve invasive-species problems in all cases, but there have been a number of remarkable successes. In each of these cases, the benefits have far outweighed the costs. Biological control is nonetheless often the subject of some debate, since it involves the introduction of yet more alien species into new environments, with some degree of risk.[54] We believe, at least in the case of many invasive alien plant species in South Africa, that biological control offers one of the best and most cost-effective interventions for addressing the problem.

Finally, in arguing a case for the control of invasive plants in South Africa, there are bound to be conflicts. Many invasive species also bring benefits, and with benefits come vested interests. In such cases, the argument for or against control often becomes polarized, with the case being argued from one side or the other, with little balance (see, for example, Johns[55] and Cellier[56]). The education of the broader public in understanding the complexities of the problem, and thus enabling the public to judge the merits of control programs, is an enormous and critical challenge if broad support is to be obtained.

References

1. Kaiser, J., Stemming the tide of invasive species, *Science* 285, 1836–1841, 1999.
2. Cowling, R.M. and Hilton-Taylor, C., Patterns of plant diversity and endemism in southern Africa: an overview, in *Botanical Diversity in Southern Africa,* Huntley, B.J. (Ed.), National Botanical Institute, Pretoria, 1994, pp 31–52.
3. Richardson, D.M., et al., Vectors and pathways of biological invasions in South Africa - Past, future and present, in *Pathways of Biological Invasions,* Carlton, J. and Ruiz, G. (Eds.), Island Press, Washington, D.C., in press.
4. Van Wilgen, B.W. and van Wyk, E., Invading alien plants in South Africa: impacts and solutions, in *People and rangelands: Building the Future*, Proceedings of the VI International Rangeland Congress, Eldridge, D. and Freudenberger D. (Eds.), Townsville, Australia, 1999, 566–571.
5. Macdonald, I.A. W., et al., *The Ecology and Management of Biological Invasions in Southern Africa,* Oxford University Press, Cape Town, 1986.
6. Richardson, D.M., et al., Plant and animal invasions, in *The Ecology of Fynbos*, Cowling R.M. (Ed.), Oxford University Press, Cape Town, 1992, 271–308.
7. Richardson, D.M., et al., Alien plant invasions, in *Vegetation of Southern Africa,* Cowling, R.M., Richardson, D.M., and Pierce, S.M. (Eds.), Cambridge University Press, Cambridge, 1997, 534–570.
8. Le Maitre, D.C., et al., Invasive plants and water resources in the western Cape Province, South Africa: modelling the consequences of a lack of management, *Journal of Applied Ecology* 33, 161–172, 1996.
9. Van Wilgen, B.W., Cowling, R.M., and Burgers, C.J., Valuation of ecosystem services - A case study from South African fynbos ecosystems, *BioScience* 46, 184–189, 1996.
10. Van Wilgen, B.W., et al., The sustainable development of water resources: History, financial costs, and benefits of alien plant control programmes, *South African Journal of Science* 93, 404–411, 1997.
11. Van Wilgen, B.W., Le Maitre, D.C., and Cowling, R.M., Ecosystem services, efficiency, sustainability and equity: South Africa's Working for Water Programme, *Trends in Ecology and Evolution,* 13, 378, 1998.
12. Henderson, L., South African plant invaders atlas (SAPIA), *Applied Plant Science*, 12, 31, 1998.
13. Henderson, L., The southern African Plant Invaders Atlas (SAPIA) and its contribution to biological weed control, *African Entomology Memoir* 1, 159–163, 1999.
14. Hoffman, M.T., et al., *Land Degradation in South Africa*, National Botanical Institute, Kirstenbosch, South Africa, 1999.
15. Versfeld, D.B., Le Maitre, D.C., and Chapman, R.A., *Alien Invading Plants and Water Resources in South Africa: A Preliminary Assessment*, Report TT99/98, Water Research Commission, Pretoria, 1998.
16. Le Maitre, D.C., Versfeld, D.B., and Chapman, R.A., The impact of invading alien plants on surface water resources in South Africa: a preliminary assessment, *Water* SA 26, 397–408, 2000.
17. Cowling, R.M., et al., *Framework for a Conservation Plan for the Cape Floristic Region*. Institute for Plant Conservation, University of Cape Town, 1999.
18. Milton, S.J., Zimmermann, H.G., and Hoffmann, J.H., Alien plant invaders of the karoo: attributes, impacts and control, in *The Karoo — Ecological Patterns and Process*, Dean, W.R.J. and Milton, S.J. (Eds.), Cambridge University Press, Cambridge, 1999, 274–287.
19. Parker, I.M., et al., Impact: toward a framework for understanding the ecological effect of invaders, *Biol. Invasions* 1, 3–19, 1999.
20. Chapin, F.S. III, et al., The functional role of species in terrestrial ecosystems, in *Global Change and Terrestrial Ecosystems*, Walker, B. and Steffen, W. (Eds.), Cambridge University Press, Cambridge, 1996, 403–428.

21. Dukes, J.S. and Mooney, H.A., Biological invaders disrupt ecosystem processes in western North America, in *Disruptions and Variability in Ecosystem Processes and Patterns,* Bradshaw, G.A., Mooney H.A., and Alaback, P. (Eds.), Springer-Verlag, New York, in press.
22. Hall, A.V., Threatened plants in the fynbos and karoo biomes, *Biological Conservation* 40, 29–52, 1987.
23. Macdonald, I.A. W., Range expansion in the Pied Barbet and the spread of alien tree species in southern Africa, *Ostrich* 57, 75–94, 1986.
24. Dye, P., Climate, forest and streamflow relationships in South African afforested catchments, *Commonwealth Forestry Review* 75, 31–38, 1996.
25. Prinsloo, F.W. and Scott, D.F., Streamflow responses to the clearing of alien trees from riparian zones at three sites in the Western Cape, *South African Forestry Journal* 185, 1–7, 1999.
26. Musil, C.F. and Midgley, G.F., The relative impact of invasive Australian acacias, fire and season on the soil chemical status of a sand plain lowland fynbos community, *South African Journal of Botany* 56, 419–427, 1990.
27. Witkowski, E.T. F., Effects of invasive alien Acacias on nutrient cycling in the coastal lowlands of the Cape Fynbos, *Journal of Applied Ecology* 28, 1–15, 1991.
28. Musil, C.F., Effect of invasive Australian acacias on the regeneration, growth and nutrient chemistry of South African lowland fynbos, *Journal of Applied Ecology* 30, 361–372, 1993.
29. Stock, W.D. and Allsopp, N., Functional perspectives of ecosystems, in *The Ecology of Fynbos; Nutrients, Fire and Diversity,* Cowling R.M. (Ed.), Oxford University Press, Cape Town, 1992, 241–259.
30. Yelenik, S., Ecosystem-level feedbacks of N-fixing invasive aliens in lowland fynbos, MSc dissertation, University of Cape Town, 2000.
31. Holmes, P.M. and Cowling, R.M., The effects of invasion by *Acacia saligna* on the guild structure and regeneration capabilities of South African fynbos shrublands, *Journal of Applied Ecology* 34, 317–332, 1997.
32. Holmes, P.M. and Cowling, R.M., Diversity, composition and guild structure relationships between soil-stored seed banks and mature vegetation in alien plant-invaded South African fynbos shrublands, *Plant Ecology* 133, 107–122,1997.
33. Van Wilgen, B.W., Bond, W.J., and Richardson, D.M., Ecosystem Management, in T*he Ecology of Fynbos; Nutrients, Fire and Diversity,* Cowling, R.M., Ed., Oxford University Press, Cape Town, 1992, 345–371.
34. Hosking, S.G. and Du Preez, M., A cost-benefit analysis of removing alien trees in the Tsitsikamma mountain catchment, *South African Journal of Science* 95, 442–448, 1999.
35. Le Maitre, D.C., et al., Invasive alien trees and water resources in South Africa: case studies of the costs and benefits of management, *Forest Ecology and Management,* in press.
36. De Wit, M., Crookes, D., and van Wilgen, B.W., Conflicts of interest in environmental management: Estimating the costs and benefits of black wattle (*Acacia mearnsii*) in South Africa, *Biological Invasions,* in press.
37. Turpie, J. and Heydenrych, B., Economic consequences of alien infestation of the Cape Floral Kingdom's Fynbos vegetation, in *The Economics of Biological Invasions*, Perrins, C., Williamson, M., and, Dalmazzone S. (Eds.), Edward Elgar, Cheltenham, 2000, 152–182.
38. Higgins, S.I., et al., A dynamic ecological-economic model as a tool for conflict resolution in an invasive alien, biological control and native-plant scenario, *Ecological Economics* 22, 141–154, 1997.
39. Van Wilgen, B.W. and Richardson, D.M., The effects of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands: A simulation study, *Journal of Applied Ecology* 22, 955–966, 1985.
40. Scott, D.F., Prinsloo, F.W., and Le Maitre, D.C., The role of invasive alien vegetation in the Cape Peninsula fires of January 2000, Report ENV-S-C 2000–039, Division of Water, Environment and Forestry Technology, CSIR, Stellenbosch, 2000.
41. Euston-Brown, D., The influence of vegetation type and fire severity on catchment stability after fire: a case study from the Cape Peninsula, South Africa, Working for Water Programme, Department of Water Affairs and Forestry, Cape Town, 2000.

42. Scott, D.F., Report on an inspection of the burn site between Fish Hoek and Glencairn, Monday 10 May, 1999, Ad hoc report prepared for Gibb Africa, Cape Town, 1999.
43. Kruger, F.J., et al., A review of the veld fires in the Western Cape during 15 to 25 January 2000, Department of Water Affairs and Forestry, Pretoria, 2000.
44. Richardson, D.M., Forestry trees as invasive aliens, *Conservation Biology* 12, 18–26, 1998.
45. Brutsch, M.O. and Zimmermann, H.G., The prickly pear (*Opuntia ficus-indica* [Cactaceae]) in South Africa: Utilization of the naturalized weed, and of the cultivated plants, *Economic Botany* 47, 154–162, 1993.
46. Johannesmeier, M.F., Beeplants of the South-Western Cape, Department of Agriculture, Pretoria, 1985, 59 pp.
47. Higgins, S.I., et al., An ecological economic simulation model of mountain fynbos ecosystems: Dynamics, valuation and management, *Ecological Economics* 22, 155–169, 1997.
47a. Hill, M.P., personal communication.
48. Marais, C., An economic evaluation of invasive alien plant control programmes in the mountain catchment areas of the Western Cape Province, South Africa, PhD dissertation, University of Stellenbosch, 1998.
49. Eckert, J.B., Liebenberg, G.F., and Troskie, D.P., The Macroeconomics of Western Cape Agriculture: Analysis with a Social Accounting Matrix, Department of Agriculture of the Western Cape, Cape Town, 1997.
50. Whiteford, A. and Mcgrath, M., Distribution of Income in South Africa, Human Sciences Research Council, Pretoria, 1994.
51. Olckers, T. and Hill, M.P., Biological control of weeds in South Africa (1990–1998), *African Entomology, Memoir No. 1*, 1999.
52. Stubbings, J.A., A case against controlling introduced Acacias (A.C.A.C.I.A.), *South African Forestry Journal* 102, 8–14, 1977.
53. Pieterse, P.J. and Boucher, C., A.C.A.C.I.A. (A Case Against Controlling Introduced Acacias) — 19 years later, *South African Forestry Journal* 180, 37–44, 1997.
54. Van Wilgen, B.W., et al., Big returns from small organisms: developing a strategy for the biological control of invasive alien plants in *South Africa, South African Journal of Science* 96, 148–152, 2000.
55. Johns, M., Are all trees green? The spotlight on forestry, *Africa Environment and Wildlife* 1(3), 77–85, 1993.
56. Cellier, S., Are all trees green? The forest industry replies, *Africa Environment and Wildlife* 2(1), 79–85, 1994.
57. Low, A.B. and Rebelo, A.G., Vegetation of South Africa, Lesotho and Swaziland. Department of Environmental Affairs and Tourism, Pretoria, 1996.
58. Cole, N.S., et al., Framework for a conservation plan for the Agulhas Plain, Cape Floristic Region, South Africa. Institute for Plant Conservation, University of Cape Town, 2000.
59. Avis, A.M., A review of coastal dune stabilization in the Cape Province of South Africa, *Landscape and Urban Planning* 18, 55–68, 1989.
60. Fraser, M.W., Foods of Redwinged Starlings and the potential for avian dispersal of *Acacia cyclops* at the Cape of Good Hope Nature Reserve, *South African Journal of Ecology* 1, 73–76, 1990.
61. Knight, R.S. and Macdonald, I.A. W., Acacias and korhaans: an artificially assembled seed dispersal system, *South African Journal of Botany* 57, 220–225, 1991.
62. Higgins, S.I., et al., Predicting the landscape-scale distribution of alien plants and their threat to plant diversity, *Conservation Biology* 13, 303–313, 1999.
63. Milton, S.J. and Siegfried, W.R., Above-ground biomass of Australian Acacias in the Southern Cape, South Africa, *Journal of South African Botany* 47, 701–716, 1981.
64. Milton, S.J., Litterfall of the exotic Acacias in the South Western Cape, *Journal of South African Botany* 47, 147–155, 1981.
65. Samways, M.J., Caldwell, P.M., and Osborn, R., Ground-living invertebrate assemblages in native, planted and invasive vegetation in South Africa, *Agriculture, Ecosystems and Environment* 59, 19–32, 1996.

66. Geldenhuys, C.J., le Roux, P.J., and Cooper, K.H., Alien invasions in indigenous evergreen forest. in *The Ecology and Management of Biological Invasions in Southern Africa*, Macdonald, I.A.W., Kruger F.J., and Ferrar A.A. (Eds.), Oxford University Press, Cape Town, 1986, 119–131.
67. Macdonald, I.A. W., Alien trees, shrubs and creepers invading indigenous vegetation in the Hluhluwe–Umfolozi Game Reserve Complex in Natal, *Bothalia* 14, 949–959, 1983.
68. Hill, M.P. and Cilliers, C.J., A review of the arthropod natural enemies, and factors that influence their efficacy, in the biological control of water hyacinth, *Eichhornia crassipes* (Mart.) Solms-Laubach (Ponterderiaceae), in South Africa, *African Entomology Memoir* 1, 103–112, 1999.
69. Richardson, D.M. and Van Wilgen, B.W., The effects of fire in felled *Hakea sericea* and natural fynbos and implications for weed control in mountain catchments, *South African Forestry* Journal 139, 4–14, 1986.
70. Breytenbach, G.J., Alien control: can we afford to slash and burn hakea in fynbos ecosystems?, *South African Forestry Journal* 151, 6–16, 1989.
71. Seydack A.H. W., Fire management options in fynbos mountain catchment areas, *South African Forestry Journal* 161, 53–58, 1992.
72. Breytenbach, G.J., Impacts of alien organisms on terrestrial communities with emphasis on communities of the south-western Cape in *The Ecology and Management of Biological Invasions in Southern Africa,* Macdonald, I.A.W., Kruger F.J., and Ferrar A.A. (Eds.), Oxford University Press, Cape Town, 1986, 229–238.
73. Baars, J-R. and Neser, S., Past and present initiatives on the biological control of Lantana camara (Verbenaceae) in South Africa, *African Entomology Memoir* 1, 21–33, 1999.
74. Cilliers, C.J., Biological control of parrot's feather, *Myriophyllum aquaticum* (Vell.) Verd. (Haloragaceae), in South Africa, *African Entomology Memoi*r 1, 113–118, 1999.
75. Stirton, C.M., *Plant Invaders, Beautiful but Dangerous*, Cape Provincial Administration, Cape Town, 1978.
76. Richardson, D.M., Macdonald, I.A. W., and Forsyth, G.G., Reductions in plant species richness under stands of alien trees and shrubs in the fynbos biome, *South African Forestry Journal* 149, 1–8, 1989.
77. Harding, G.B. and Bate, G.C., The occurrence of invasive *Prosopis* species in the north-western Cape, South Africa, *South African Journal of Science* 87, 188–192, 1991.
78. Brown, C.J. and Gubb, A.A., Invasive alien organisms in the Namib Desert, Upper Karoo and the arid and semi-arid savannas of western southern Africa in *The Ecology and Management of Biological Invasions in Southern Africa*, Macdonald, I.A.W., Kruger F.J., and Ferrar A.A. (Eds.), Oxford University Press, Cape Town, 1986, 93–108.
79. Steenkamp, H.E. and Chown, S.L., Influence of dense stands of an exotic tree, *Prosopis glandulosa,* Benson, on a savanna dung beetle (Coleoptera: Scarabaeinae) assemblage in southern Africa, *Biological Conservati*on 78, 305–311, 1996.
80. Stirton, C.H., Notes on the taxonomy of Rubus in southern Africa. *Bothalia* 13, 331–332, 1981.
81. Henderson L., Alien invasive Salix spp. (willows) in the Grassland Biome of *South Africa, South African Forestry Journal* 157, 91–95, 1991.
82. Bromilow, C. *Problem Plants of South Africa*. Briza Publications, Pretoria, 1995.
83. Hoffmann, J. H. and Moran, C.M., The invasive weed, *Sesbania punicea* in South Africa and prospects for its biological control, *South African Journal of Science* 84, 740–742, 1988.
84. Oatley, T.B., Exploitation of a new niche by the Rameron Pigeon *Columba arquatrix* in Natal, *Proceedings of the Fifth Pan-African Ornithological Congress*, Southern African Ornithological Society, Johannesburg, 1984, pp 323–330.
85. Heydenrych, B., An investigation of land-use practices on the Agulhas Plain (South Africa), with emphasis on socio-economic and conservation issues, MSc dissertation, University of Cape Town, 1999.
86. Department of Water Affairs and Forestry, *The Working for Water Programme 1996/97 Annual Report*, Department of Water Affairs and Forestry, Pretoria, 1997.

section eight

United States

chapter sixteen

Environmental and economic costs associated with non-indigenous species in the United States*

David Pimentel, Lori Lach, Rodolfo Zuniga, and Doug Morrison

Contents

* This chapter appeared in *BioScience,* 50, 53–65, 2000. It is reprinted here with permission.

16.1 Introduction

In the history of the United States, approximately 50,000 non-indigenous (non-native) species are estimated to have been introduced. Introduced species, such as corn, wheat, rice, and other food crops, and cattle, poultry, and other livestock, now account for more than 98% of the production of the U.S. food system, at a value of approximately $800 billion per year.[1] Other exotic species have been introduced for landscape restoration, biological pest control, sport, pets, and food processing. Some non-indigenous species, however, have caused major economic losses in agriculture, forestry, and several other segments of the U.S. economy, in addition to having harmed the environment. One recent study reported approximately $97 billion in damage from 79 exotic species during the period from 1906 to 1991.[2]

Estimating the full extent of the environmental damage caused by exotic species and the number of species extinctions they have led to is difficult, because little is known about the estimated 750,000 species in the United States, half of which have not even been described.[3] Nonetheless, about 400 of the 958 species that are listed as threatened or endangered under the Endangered Species Act are considered to be at risk primarily because of competition with and predation by non-indigenous species.[4,5] In other regions of the world, as many as 80% of the endangered species are threatened because of the pressures of non-native species.[6] Many other species worldwide that are not listed as endangered are also negatively affected by alien species and ecosystem changes caused by alien species.

Estimating the economic impacts associated with non-indigenous species in the United States is also difficult; nevertheless, enough data are available to quantify some of the impacts on agriculture, forestry, and public health. In this chapter we assess as much as possible the magnitude of the environmental impacts and economic costs associated with the diverse non-indigenous species that have become established within the United States. Although species translocated within the United States can also have significant impacts, this assessment is limited to non-indigenous species that did not originate within the United States or its territories.

16.2 Environmental damages and associated control costs

Most plant and vertebrate animal introductions have been intentional, whereas most invertebrate animal and microbe introductions have been accidental. In the past 40 years, the invasion rates and the risks associated with biotic invaders have increased enormously because of human population growth, rapid movement of people, and alteration of the environment. In addition, more goods and materials are being traded among nations than ever before, thereby creating increased opportunities for unintentional introductions.[1,7]

Some of the approximately 50,000 species of plants and animals that have invaded the United States cause many different types of damage to managed and natural ecosystems (Table 16.1). Some of these damages and control costs are assessed below.

16.2.1 Plants

Most alien plants now established in the United States were introduced for food, fiber, or ornamental purposes. An estimated 5000 introduced plant species have escaped and now exist in U.S. natural ecosystems,[8] compared with a total of about 17,000 species of native U.S. plants.[9] In Florida, of the approximately 25,000 alien plant species that have been imported, mainly as ornamentals for cultivation, more than 900 have escaped and have become established in surrounding natural ecosystems.[10–12] More than 3000 plant species

Table 16.1 Estimated Annual Costs Associated with Some Non-Indigenous Species Introduction in the United States (see text for details and sources) (× millions of dollars)

Category	Non-indigenous species	Losses and damages	Control costs	Total
Plants	25,000			
Purple loosestrife		—	—	$45
Aquatic weeds		$10	$100	110
Melaleuca tree		NA	3–6	3–6
Crop weeds		24,000	3000	27,000
Weeds in pastures		1000	5000	6000
Weeds in lawns, gardens, golf courses		NA	1500	1500
Mammals	20			
Wild horses and burros		5	NA	5
Feral pigs		800	0.5	800.5
Mongooses		50	NA	50
Rats		19,000	NA	19,000
Cats		17,000	NA	17,000
Dogs		250	NA	250
Birds	97			
Pigeons		1100	NA	1100
Starlings		800	NA	800
Reptiles & Amphibians	53			
Brown tree snake		1	4.6	5.6
Fish	138	1000	NA	1000
Arthropods	4500			
Imported fire ant		600	400	1000
Formosan termite		1000	NA	1000
Green crab		44	NA	44
Gypsy moth		NA	11	11
Crop pests		13,900	500	14,400
Pests in lawns, gardens, golf courses		NA	1500	1500
Forest pests		2100	NA	2100
Mollusks	88			
Zebra mussel		—	—	100
Asian clam		1000	NA	1000
Shipworm		205	NA	205
Microbes	20,000			
Crop plant pathogens		21,000	500	21,500
Plant pathogens in lawns, gardens, golf courses		NA	2000	2000
Forest plant pathogens		2100	NA	2100
Dutch elm disease		NA	100	100
Livestock Diseases		9000	NA	9000
Human Diseases		NA	6500	6500
Total				**$137,232.1**

have been introduced into California, and many of these have escaped into the natural ecosystem as well.[13]

Some of the 5000 non-indigenous plants established in U.S. natural ecosystems have displaced several native plant species.[8] Non-indigenous weeds are spreading and invading approximately 700,000 ha/year of the U.S. wildlife habitat.[14] One of these pest weeds is the European purple loosestrife (*Lythrum salicaria*), which was introduced in the early 19th century as an ornamental plant.[15] It has been spreading at a rate of 115,000 ha/year and is changing the basic structure of most of the wetlands it has invaded.[16] Competitive stands of purple loosestrife have reduced the biomass of 44 native plants and therefore are threatening endangered wildlife species, including the bog turtle (*Clemmys muhlenbengil*) and several duck species, that depend on these native plants.[17] Loosestrife now occurs in 48 states and costs $45 million per year in control costs and forage losses.[18]

Many introduced plant species established in the wild are having an effect on U.S. parks as well.[19] In Great Smoky Mountains National Park, 400 of approximately 1500 vascular plant species are exotic, and 10 of these are currently displacing and threatening other species in the park.[19]

The problem of introduced plants is especially significant in Hawaii. Hawaii has a total of 2690 plant species, 946 of which are non-indigenous species.[20] About 800 native species are currently endangered.[21]

Sometimes a single non-indigenous plant species will competitively overrun an entire ecosystem. For example, in California, yellow star thistle (*Centaurea solstitalis*) now dominates more that 4 million ha of northern California grassland, resulting in the total loss of this once productive grassland.[22]

Similarly, European cheatgrass (*Bromus tectorum*) is dramatically changing the vegetation and fauna of many natural ecosystems. This annual grass has invaded and spread throughout the shrub–steppe habitat of the Great Basin in Idaho and Utah, predisposing the invaded habitat to fires.[23–25] Before the invasion of cheatgrass, fire burned once every 60 to 110 years, and shrubs had a chance to become well established. Now, fires occur about every 3 to 5 years; shrubs and other vegetation are diminished, and competitive monocultures of cheatgrass now exist on 5 million ha in Idaho and Utah.[26] The animals that are dependent on the shrubs and other original vegetation have been reduced in numbers or eliminated.

An estimated 138 non-indigenous tree and shrub species have invaded native U.S. forest and shrub ecosystems.[27] Introduced trees include salt cedar (*Tamarix pendantra*), eucalyptus (*Eucalyptus* spp.), Brazilian pepper (*Schinus terebinthifolius*), and Australian melaleuca (*Melaleuca quenquenervia*).[2,28,29] Some of these trees have displaced native trees, shrubs, and other vegetation types, and populations of some associated native animal species have in turn been reduced.[2] For example, the melaleuca tree is competitively spreading at a rate of 11,000 ha/year throughout the vast forest and grassland ecosystems of the Florida Everglades,[22] where it damages the natural vegetation and wildlife.[2]

Exotic aquatic weeds are also a significant problem in the United States. For example, in the Hudson River Basin of New York, there are now 53 exotic aquatic weed species.[30] In Florida, exotic aquatic plants such as hydrilla (*Hydrilla verticillata*), water hyacinth (*Eichhornia crassipes*), and water lettuce (*Pistia straiotes*) are altering fish and other aquatic animal species, choking waterways, altering nutrient cycles, and reducing recreational use of rivers and lakes. Active control measures of the aquatic weeds have become necessary.[2] For instance, Florida spends about $14.5 million each year on hydrilla control.[31] Nevertheless, hydrilla infestations in just two Florida lakes have caused an estimated $10 million worth of recreational losses annually.[31] In the United States as a whole, a total of $100 million is invested annually in aquatic weed control targeting non-indigenous species.[2]

16.2.2 Mammals

About 20 species of mammals have been introduced into the United States; these include dogs, cats, horses, burros, cattle, sheep, pigs, goats, and deer.[32] Several of these species escaped or were released into the wild; many have become pests by preying on native animals, grazing on vegetation, or intensifying soil erosion. For example, goats (*Capra hirus*) introduced on San Clemente Island, California, are responsible for the extinction of eight endemic plant species there and the endangerment of another eight native plant species.[23]

Many small mammals have also been introduced into the United States. These species include a number of rodents, including the European (black or tree) rat *Rattus rattus*, the Asiatic (Norway or brown) rat *Rattus norvegicus*, the house mouse *Mus musculus*, and the European rabbit *Oryctolagus cuniculus*.[32]

Some introduced rodents have become serious pests on farms, in industries, and in homes.[32] Rats and mice are particularly abundant and destructive on farms. On poultry farms, there is approximately one rat for every five chickens.[33,34] Using this ratio, the total rat population on U.S. poultry farms may easily number more than 1.4 billion.[35] Assuming that the number of rats per chicken has declined because of improved rat control since these observations were made, we estimate that the number of rats on poultry and other farms is now approximately 1 billion. With an estimated additional one rat per person in homes and related areas,[36] there are an additional 250 million rats in the United States.[1]

If we assume, conservatively, that each adult rat consumes or destroys stored grains[37,38] and other materials valued at $15/year, then the total cost of destruction by introduced rats in the United States is more than $19 billion per year. In addition, rats cause fires by gnawing through electrical wires; they pollute foodstuffs; and they act as vectors of several diseases, including salmonellosis and leptospirosis and, to a lesser degree, plague and murine typhus.[39] They also prey on some native invertebrate and vertebrate species, such as birds and bird eggs.[40]

One of the first cases of the failure of biological control was the use of the Indian mongoose (*Herpestes auropunctatus*). It was first introduced into Jamaica in 1872 for biological control of rats in sugarcane.[41] It was subsequently introduced to the territory of Puerto Rico, other islands in the West Indies, and Hawaii for the same purpose. The mongoose controlled the Asiatic rat but not the European rat, and it preyed heavily on native ground-nesting birds.[41,42] It also preyed on beneficial native amphibians and reptiles, causing at least seven amphibian and reptile extinctions in Puerto Rico and on other islands in the West Indies.[43] In addition, the mongoose emerged as the major vector and reservoir of rabies and leptospirosis in Puerto Rico and other islands.[44] Based on public health damage, the killing of poultry in Puerto Rico and Hawaii, extinctions of amphibians and reptiles, and destruction of native birds, we estimate that the mongoose is causing approximately $50 million in damage each year in Puerto Rico and the Hawaiian Islands.[45,46]

Introduced cats have also become a serious threat to some native birds and other animals. There are an estimated 63 million pet cats in the United States,[47] plus as many as 30 million feral cats.[48] Cats prey on native birds,[49] plus small native mammals, amphibians, and reptiles.[50] Estimates are that feral cats in Wisconsin and Virginia kill more than 3 million birds in each state per year.[48] Based on the Wisconsin and Virginia data, we assume that five birds are killed per feral cat per year; McKay[51] reports that pet cats kill a similar number of birds per capita as feral cats. Thus, about 465 million birds are killed by cats each year in the nation. Each adult bird can be valued at $30. This cost per bird is based on the literature that reports that a bird-watcher spends $0.40 per bird observed, a hunter spends $216 per bird shot, and specialists spend $800 per bird reared for release; in addition, it should be noted that the EPA fines polluters $10 per fish killed, including small, immature fish.[52] Therefore, the total damage to U.S. bird populations is approximately $17 billion per year. This figure does not include small mammals, amphibians, and reptiles that are killed by feral and pet cats.[50]

Like cats, most dogs introduced into the United States were introduced for domestic purposes, but some have escaped into the wild. Many of these wild dogs run in packs and kill deer, rabbits, and domestic cattle, sheep, and goats. Carter[53] reported that feral dog packs in Texas cause more than $5 million in livestock losses each year. Dog packs have also become a serious problem in Florida.[32] In addition to the damage caused by dogs in Texas, and conservatively assuming $5 million for all damage in the other 49 states combined, total losses in livestock kills by dogs per year would be approximately $10 million per year.

Moreover, an estimated 4.7 million people are bitten by feral and pet dogs annually, with 800,000 cases requiring medical treatment.[54] The Centers for Disease Control and Prevention estimates that medical treatment for dog bites costs $165 million a year, and the indirect costs, such as lost work, increase the total costs of dog bites to $250 million a year.[55] In addition, dog attacks cause between 11 and 14 deaths per year, and 80% of these victims are small children.[56]

16.2.3 Birds

Approximately 97 of the 1000 bird species in the United States are exotic.[57] Of the approximately 97 introduced bird species, only 5%, including chickens, are considered beneficial. Most, though (56%), are considered pests.[57] Pest species include the pigeon, which was introduced into the United States for agricultural purposes.

Introduced bird species are an especially severe problem in Hawaii. A total of 35 of the 69 non-indigenous bird species introduced between 1850 and 1984 in Hawaii are still extant on the islands.[58,59] One such species, the common myna (*Acridotheres tristis*), was introduced to help control pest cutworms and armyworms in sugarcane.[23] However, it became the major disperser of seeds of an introduced serious weed, *Lantana camara*.

In the continental United States, the English or house sparrow (*Passer domesticus*) was introduced in 1853 to control the canker worm.[60,61] By 1900, the sparrows had become pests, because they damage plants around homes and public buildings and they consume wheat, corn, and the buds of fruit trees.[60] Furthermore, English sparrows harass native birds, including robins, Baltimore orioles, yellow-billed cuckoos, and black-billed cuckoos, and they displace native bluebirds, wrens, purple martins, and cliff swallows from their nesting sites.[61,62] They are also associated with the spread of about 29 human and livestock diseases. [63]

The single most serious pest bird in the United States is the exotic common pigeon (*Columba livia*) that exists in most cities of the world, including those in the United States.[64] Pigeons are considered a nuisance because they foul buildings, statues, cars, and sometimes people, and they feed on grain.[62,65] The control costs of pigeons are at least $9 per pigeon per year.[66] Assuming one pigeon per ha in urban areas,[67] or approximately 0.5 pigeons per person, and using potential control costs as a surrogate for losses, pigeons cause an estimated $1.1 billion per year in damage. These control costs do not include the environmental damage associated with pigeons; the birds serve as reservoirs and vectors for more than 50 human and livestock diseases, including parrot fever, ornithosis, histoplasmosis, and encephalitis.[62,63]

16.2.4 Amphibians and reptiles

Amphibians and reptiles introduced into the United States number about 53 species. All of these non-indigenous species occur in relatively warm states — Florida is now host to 30 species, and Hawaii to 12.[68,69] The negative ecological impacts of several of these exotic species have been enormous.

The brown tree snake (*Boiga irregularis*) was accidentally introduced to the snake-free U.S. territory of Guam immediately after World War II when military equipment was moved onto Guam.[70] Soon the snake population reached densities of 100 per ha, and dramatically reduced native bird, mammal, and lizard populations. Of the 13 species of native forest birds originally found on Guam, only 3 still exist[70]; of the 12 native species of lizards, only 3 have survived.[71] The snake eats chickens, eggs, and caged birds, causing major problems to small farmers and pet owners. It also crawls up trees and utility poles and has caused power outages on the island. One island-wide power outage caused by the snake cost the power utility more than $250,000.[72] Local outages that affect businesses are estimated to cost from $2000 to $10,000 per commercial customer.[73] Since the island experiences about 86 snake-related outages per year,[74] our estimate of the cost of these incidents is conservatively $1 million per year.

In addition, the brown tree snake is slightly venomous and has caused public health problems, especially when it has bitten children. At one hospital emergency room, about 26 people per year are treated for brown tree snake bites.[2] Some bitten infants require hospitalization and intensive care, at an estimated total cost of $25,000 per year.[75]

The total costs of endangered-species recovery efforts, environmental planning related to snake containment on Guam, and other programs directly stemming from the snake's invasion of Guam reach more than $1 million per year; in addition, up to $2 million per year is invested in research to control this serious pest.[76] The brown tree snake has also invaded Hawaii but thus far has been exterminated there. Hawaii's concern about the snake, though, has prompted the federal government to invest $1.6 million per year in brown tree snake control.[77] The total cost associated with the snake is therefore more than $5.6 million per year.

16.2.5 Fish

A total of 138 non-indigenous fish species have been introduced into the United States.[78–80] Most of these introduced species have been established in states with mild climates, such as Florida (50 species) [80] and California (56 species).[81] In Hawaii, 33 non-indigenous freshwater fish species have become established.[82] Forty-four native species of fish are threatened or endangered in the United States by non-indigenous fish species.[83] An additional 27 native fish species are also negatively affected by introductions.[83]

Introduced fish species frequently alter the ecology of aquatic ecosystems. For instance, the grass carp (*Ctenopharyngodon idella*) reduces natural aquatic vegetation, while the common carp (*Cyprinus carpio*) reduces water quality by increasing turbidity. These changes have caused the extinctions of some native fish species.[84]

Although some native fish species are reduced in numbers, driven to extinction, or hybridized by non-indigenous fish species, alien fish do provide some economic benefits in the improvement of sport fishing. Sport fishing contributes $69 billion to the economy of the United States.[1,85] However, even taking into account these economic benefits, based on the more than 40 non-indigenous species that have negatively affected native fishes and other aquatic biota, a conservative estimate puts the economic losses due to exotic fish at more than $1 billion annually.

16.2.6 Arthropods and annelids

Approximately 4500 arthropod species (2582 species in Hawaii and more than 2000 in the continental United States) have been introduced to the United States. Also, 11 earthworm species[86] and nearly 100 aquatic invertebrate species have been introduced.[2] About 95% of these introductions were accidental, with many species gaining entrance via plants or through soil and water ballast from ships.

For example, the accidentally introduced balsam woolly adelgid (*Adelges piceae*) inflicts severe damage in balsam–fir natural forest ecosystems.[87] According to Alsop and Laughlin,[88] this aphid is destroying the old-growth spruce–fir forest in many regions. Over the past two decades it has spread throughout the southern Appalachians, where it has destroyed up to 95% of the Fraser firs.[89] Alsop and Laughlin[88] report the loss of two native bird species and invasion by three other bird species as a result of adelgid-mediated forest death.

Other introduced insect species have become pests of livestock and wildlife. For example, the red imported fire ant (*Solenopsis invicta*) kills poultry chicks, lizards, snakes, and ground-nesting birds.[90] A 34% decrease in swallow nesting success, as well as a decline in the northern bobwhite quail populations, was reported as attributable to these ants.[91] The estimated damage to livestock, wildlife, and public health caused by fire ants in Texas is estimated to be $300 million a year. An additional $200 million is invested in control per year.[92,93] Assuming similar damages in other infested southern states, such as Florida, Georgia, and Louisiana, the fire ant causes damage totaling more than $1 billion per year. Southern states are also affected by another insect, the Formosan termite (*Coptotermes formosanus*), which is reported to cause structural damage totaling approximately $1 billion per year in the southern United States, especially in the New Orleans region.[94]

The European green crab (*Carcinus maenas*) has been associated with the demise of the softshell clam industry in New England and the Maritime Provinces of Canada.[95] It also destroys commercial shellfish beds and preys on large numbers of native oysters and crabs,[95] causing an annual estimated economic impact of $44 million per year.[95]

16.2.7 Mollusks

Eighty-eight species of mollusks have been both intentionally and accidentally introduced and established in U.S. aquatic ecosystems.[2] Two have become serious pests: the zebra mussel (*Dreissena polymorpha*) and the Asian clam (*Corbicula fluminea*).

The zebra mussel was first found in Detroit's Lake St. Clair after it had gained entrance via ballast water released in the Great Lakes from ships that had traveled from Europe.[96] It has now spread into most of the aquatic ecosystems in the eastern United States and is expected to invade most freshwater habitats throughout the nation within approximately 20 years.[96] Large mussel populations reduce the available food and oxygen for native fauna. In addition, zebra mussels have been observed completely covering native mussels, clams, and snails, thereby further threatening their survival.[96,97] Mussel densities have reached 700,000 per m^2 in some locations.[98] Zebra mussels also invade and clog water intake pipes and water filtration and power generating plants; it was estimated that they will cause $100 million per year in damage to these facilities and associated control costs by the year 2000.[99]

Although the Asian clam grows and disperses less rapidly than the zebra mussel, it too is causing significant fouling problems and is threatening native species. Costs associated with its fouling damage are about $1 billion per year.[2,100]

Another pest mollusk is the shipworm *Teredo navalis*, which was first introduced into San Francisco Bay. It has caused serious damage since the early 1990s. Currently, damages are estimated to be approximately $200 million per year.[101]

16.3 Crop, pasture, and forest losses and associated control costs

Many weeds, pest insects, and plant pathogens are biological invaders. These non-indigenous species cause several billion dollars worth of losses to crops, pastures, and forests annually in the United States. In addition, several billion dollars are spent each year on pest control.

16.3.1 Weeds

In crop systems, including forage crops, an estimated 500 introduced plant species have become weed pests; some of these, such as Johnson grass (*Sorghum halepense*) and kudzu (*Pueraria lobata*), were actually introduced as crops and then became pests.[102] Most of these weeds were accidentally introduced with crop seeds, from ship-ballast soil, or from various imported plant materials, among which were yellow rocket (*Barbarea vulgaris*) and Canada thistle (*Cirsium arvense*).

In U.S. agriculture, weeds cause an overall reduction of 12% in crop yields. In economic terms, this reduction represents about $33 billion in lost crop production annually, based on the potential value of all U.S. crops of more than $267 billion per year.[1] Based on a survey showing that about 73% of the weed species are non-indigenous,[103] it follows that about $24 billion per year of the annual crop losses are due to introduced weeds. However, non-indigenous weeds are often more serious pests than native weeds; this estimate of $24 billion per year is therefore a conservative one. In addition to direct losses, approximately $4 billion per year of herbicides are applied to U.S. crops,[104] of which about $3 billion worth are used for control of non-indigenous weeds. Therefore, the total costs of introduced weeds to the U.S. economy is about $27 billion annually.[52,104.]

In pastures, 45% of weeds are non-indigenous species.[103] U.S. pastures provide about $10 billion in forage crops annually,[35] and the estimated losses per year due to weeds are approximately $2 billion.[105] Forage losses due to non-indigenous weeds are nearly $1 billion per year.

Some introduced weeds are toxic to cattle and wild ungulates, such as leafy spurge (*Euphoria esula*).[106] In addition, several non-indigenous thistles have reduced native forage plant species in pastures, rangelands, and forests, thus reducing cattle grazing.[107] According to former Interior Secretary Bruce Babbitt,[108] ranchers spend about $5 billion each year to control invasive non-indigenous weeds in pastures and rangelands. Nevertheless, these weeds continue to spread.

Control of weed species in lawns and gardens, and on golf courses is a significant proportion of the total management costs of about $36 billion per year.[1] In fact, Templeton et al.[109] estimated that each year about $1.3 billion of the $36 billion is spent just on residential weed, insect, and disease pest control each year. Because a large proportion of these weeds, such as dandelions (*Taraxacum officinale*), are exotics, we estimate that $500 million is spent on residential exotic weed control and an additional $1 billion is invested in non-indigenous weed control on golf courses.

Weed trees also have an economic impact; from $3 million to $6 million per year alone is being spent in efforts to control the melaleuca tree in Florida.[110]

16.3.2 Vertebrate pests

Horses (*Equus caballus*) and burros (*Equus asinus*), deliberately released in the western United States, have attained wild populations of approximately 50,000 animals.[111] These animals graze heavily on native vegetation, allowing non-indigenous annuals to displace native perennials.[112] Burros inhabiting the northwestern United States also diminish the primary food sources of native bighorn sheep and seed-eating birds, thereby reducing the abundance of these native animals.[23] In general, the large populations of introduced wild horses and burros cost the nation an estimated $5 million per year in forage losses.[113]

Feral pigs (*Sus scrofa*), native to Eurasia and North Africa, have been introduced into some U.S. parks for hunting, including parks in the California coastal prairie and in the Hawaiian Islands, and they have substantially changed the vegetation in these parks.[114] In Hawaii, more than 80% of the soil is bare in regions inhabited by pigs.[23] This disturbance allows annual plants to invade the overturned soil and intensifies soil erosion. Pig control per park in Hawaii (approximately 1500 pigs per park)[115] costs about $150,000 a year. Assuming that the three parks in Hawaii all have similar pig control problems, the total comes to $450,000 a year.[116]

Feral pigs have also become a serious problem in Florida, where their population has risen to more than 500,000[32]; similarly, in Texas their number ranges from 1 million to 1.5 million.[117] In Florida, Texas, and elsewhere, pigs damage grain, peanut, soybean, cotton, hay, and various vegetable crops, as well as the environment.[118] Pigs also transmit and are reservoirs for serious human and livestock diseases, including brucellosis, pseudobrucellosis, and trichinosis.[119]

Nationwide, there are an estimated 4 million feral pigs. Based on environmental and crop damages of about $200 per pig annually (one pig can cause up to $1000 worth of damage to crops in one night)[120] amounts to about $800 million per year. This estimate is conservative, because pigs cause significant environmental damage and diseases that cannot be easily translated into dollar values.

Other animals that threaten crop production include birds. European starlings (*Sturnus vulgaris*) are serious pests and are estimated to occur at densities of more than 1 per ha in agricultural regions.[121] Starlings are capable of destroying as much as $2000 worth of cherries per ha.[122] In grain fields, starlings consume about $6 worth of grain per ha.[122] Conservatively assuming $5 per ha for all damages to many crops in the United States, the total loss due to starlings would be approximately $800 million a year. In addition, these aggressive birds have displaced numerous native birds.[60] Starlings have also been implicated in the transmission of 25 diseases, including parrot fever and other diseases of humans.[60,63]

16.3.3 Insect and mite pests

Approximately 500 non-indigenous insect and mite species are pests in crops in the United States. Hawaii has 5246 identified native insect species, and an additional 2582 introduced insect species.[10,20,123] Introduced insects account for 98% of the crop pest insects in the state.[124] In addition to Florida's 11,500 native insect species, 949 introduced species have, mostly accidentally, invaded the state (42 species were intentionally introduced for biological control).[125] In California, the 600 introduced insect species are responsible for 67% of all crop losses.[13]

Each year, pest insects destroy about 13% of potential crop production in the United States, representing a value of about $33 billion.[1] Considering that about 40% of the pests were introduced,[103] we estimate that introduced pests cause about $13 billion in crop losses each year. In addition, about $1.2 billion worth of pesticides are applied for all insect control each year.[104] The portion applied against introduced pest insects is approximately $500 million per year. Therefore, the total annual cost for introduced non-indigenous insect pests is approximately $14.4 billion. In addition, based on the analysis of management costs of lawns, gardens, and golf courses, we estimate the control costs of pest insects and mites in lawns and gardens and on golf courses to be at least $1.5 billion a year.

About 360 non-indigenous insect species have become established in American forests as well,[126] and approximately 30% of these are now serious pests. Insects cause the loss of approximately 9% of forest products, amounting to a cost of $7 billion per year.[1,127]

Because 30% of the pests are non-indigenous, annual losses attributed to non-indigenous species are about $2.1 billion per year.

The gypsy moth (*Lymantria dispar*), intentionally introduced into Massachusetts in the 1800s for possible silk production, has developed into a major pest of U.S. forest and ornamental trees, especially oaks.[22] The U.S. Forest Service currently spends about $11 million annually on gypsy moth control.[22]

16.3.4 Plant pathogens

There are an estimated 50,000 parasitic and non-parasitic diseases of plants in the United States, most of which are caused by fungae species.[128] In addition, more than 1300 species of viruses are plant pests in the United States.[128] Many of these microbes are non-native and were introduced inadvertently with seeds and other parts of host plants, and have since become major crop pests in the United States.[103] Including the introduced plant pathogens plus other soil microbes, we estimate conservatively that more than 20,000 species of microbes have invaded the United States.

U.S. crop losses to all plant pathogens total approximately $33 billion per year.[1,104] Approximately 65%,[103] or an estimated $21 billion per year of losses, are attributable to non-indigenous plant pathogens. In addition, $720 million is spent annually for fungicides,[104] with approximately $500 million a year for the control of non-indigenous plant pathogens. This brings the costs of damage and control of non-indigenous plant pathogens to about $21.5 billion per year. In addition, based on the earlier discussion of pests in lawns and gardens and on golf courses, we estimate the control costs of plant pathogens for lawns, gardens, and golf courses to be at least $2 billion per year.

In forests, more than 20 non-indigenous species of plant pathogens attack woody plants.[126] Two of the most serious plant pathogens are the chestnut blight fungus (*Cryphonectria parasitica*) and Dutch elm disease (*Ophiostoma ulmi*). Before the accidental introduction of chestnut blight, approximately 25% of eastern U.S. deciduous forest consisted of American chestnut trees.[129] Now chestnut trees have all but disappeared. Removal of elm trees devastated by *O. ulmi* costs about $100 million a year.[22]

In addition, plant pathogens of forest plants cause the loss of approximately 9%, or $7 billion, of forest products each year.[1,127] The proportion of introduced plant pathogens in forests is similar to that of introduced insects (about 30%); thus approximately $2.1 billion in forest products are lost each year to non-indigenous plant pathogens in the United States.

16.4 Livestock pests

Exotic microbes (e.g., calf diarrhea rotavirus) and parasites (e.g., face flies, *Musca autumnalis*) were introduced along with livestock brought into the United States.[130,131] In addition to the hundreds of pest microbes and parasites that have already been introduced, more than 60 microbes and parasites could invade and become serious pests to U.S. livestock.[132] A conservative estimate of the losses to U.S. livestock from exotic microbes and parasites was reported to be approximately $3 billion per year in 1980.[130,131] Current livestock losses to pests are an estimated $9 billion per year.[133]

16.5 Human diseases

The non-indigenous diseases now having the greatest impact on humans are acquired immune deficiency syndrome (AIDS), syphilis, and influenza.[112,134] In 1993 there were

103,533 cases of AIDS and 37,267 deaths from the disease.[56] The total U.S. health care cost for the treatment of AIDS averages about $6 billion per year.[135]

New influenza strains originating in the Far East spread quickly to the United States. Influenza causes 540 deaths in the United States each year.[1] Costs of hospitalizations for a single outbreak of influenza, such as type A, can exceed $300 million per year.[136]

In addition, each year there are approximately 53,000 cases of syphilis in the United States; it costs $18.4 million each year just to treat newborn children infected with syphilis.[137]

In total, AIDS and influenza take the lives of more than 40,000 people each year in the United States, and treatment costs for these diseases total approximately $6.5 billion per year. The costs of treating other exotic diseases pushes this total much higher. An increasing threat of exotic diseases exists because of rapid transportation, encroachment of civilization into new ecosystems, and increasing environmental degradation.

16.6 The non-indigenous species threat

With more than 50,000 non-indigenous species in the United States, the fraction that are harmful does not have to be large to inflict significant damage to natural and managed ecosystems and cause public health problems. A suite of ecological factors may cause non-indigenous species to become abundant and persistent. These include the lack of natural enemies (e.g., purple loosestrife and imported fire ant); the development of new associations between alien parasite and host (e.g., the AIDS virus in humans and the gypsy moth in U.S. oaks); effective predators in a new ecosystem (e.g., the brown tree snake and feral cats); artificial or disturbed habitats that provide favorable invasive ecosystems for the aliens (e.g., weeds in crop and lawn habitats); and invasion by some highly adaptable and successful species (e.g., the water hyacinth and the zebra mussel).

Our study reveals that economic damages associated with non-indigenous species and their control amount to approximately $137 billion a year. The Office of Technology Assessment (OTA)[2] reported average costs of $1.1 billion a year ($97 billion over 85 years) for 79 species. The reason for our higher estimate is that we included more than 10 times the number of species in our assessment and found higher costs reported in the literature than the OTA[2] for some of the same species. For example, for the zebra mussel, the OTA reported damages and control costs of slightly more that $300,000 per year; we used an estimate of $5 billion per year.[99]

Although we reported total economic damages and associated control costs to be $137 billion a year, precise economic costs associated with some of the most ecologically damaging exotic species are not available. The brown tree snake, for example, has been responsible for the extinctions of dozens of bird and lizard species on Guam. Yet for this snake, only minimal cost data are known. In other cases, such as the zebra mussel and feral pigs, only combined damage and control cost data are available. The damage and control costs are considered low when compared with the extensive environmental damages these species cause. If we had been able to assign monetary values to species extinctions and losses in biodiversity, ecosystem services, and aesthetics, the costs of destructive non-indigenous species would undoubtedly be several times higher than $137 billion a year. Yet even this understated economic loss indicates that non-indigenous species are exacting a significant toll.

We recognize that nearly all of the United States' crop and livestock species are non-indigenous and have proven essential to the viability of the country's agriculture and economy. Non-indigenous crops such as corn and wheat are certainly vital to U.S. food production, but many other non-indigenous species, such as zebra mussel and exotic weeds, have enormously negative impacts.

The true challenge lies not in determining the precise costs of the impacts of exotic species, but in preventing further damage to natural and managed ecosystems. Sound prevention policies must take into account the means through which non-indigenous species gain access to and become established in the United States. Since the modes of invasion vary widely, a variety of preventive strategies will be needed. For example, public education, sanitation, and effective screening and searches at airports, seaports, and other ports of entry will help reduce opportunities for biological invaders to become established in the United States.

Fortunately, the problem is gaining the attention of policymakers. On Feb. 2, 1999, President Clinton issued an executive order allocating $28 million to combat alien species invasions and creating the interagency National Invasive Species Council, which was asked to produce a plan within 18 months for mobilizing the federal government to defend again non-indigenous species invasions. In addition, the Federal Interagency Committee for the Management of Noxious and Exotic Weeds (FICMNEW) has been formed, in part to help combat non-indigenous plant species invasions.[138] The objective of FICMNEW is education, the formation of partnerships among concerned groups, and stimulation of research on the biological invader problem. Former Interior Secretary Babbitt[139] also established the Invasive Weed Awareness Coalition to combat the invasion and spread of non-native plants, such as knapweed (*Centaurea* spp.) and St. John's wort (*Hypericum perforatum*).

Certainly, these policies and practices will help prevent accidental and intentional introduction of further potentially harmful exotic species, but we do have a long way to go before the resources devoted to the problem are in proportion to the risks. We hope that this chapter's environmental and economic assessment will advance the argument that investments made now to prevent future introductions will be returned many times over in the preservation of natural ecosystems, reductions in losses to agriculture and forestry, and the avoidance of threats to public health.

Acknowledgments

We thank the following people for reading an earlier draft of this chapter and for their many helpful suggestions: D. Bear, Council on Environmental Quality, Executive Office of the President, Washington, DC; J.W. Beardsley, University of Hawaii; A.J. Benson, U.S. Geological Survey, Gainesville, FL; B. Blossey, Cornell University; C.R. Bomar, University of Wisconsin, Stout; F.T. Campbell, Western Ancient Forest Campaign, Springfield, VA; R. Chasen, Editor, BioScience; P. Cloues, Geologic Resources Division, Natural Resource Program Center, Lakewood, CO; W.R. Courtenay, Florida Atlantic University; R.H. Cowie, Bishop Museum, Honolulu, HI; D. Decker, Cornell University; R.V. Dowell, California Department of Food and Agriculture; T. Dudley, University of California, Berkeley; H. Fraleigh, Colorado State University; H. Frank, University of Florida; T. Fritts, U.S. Geological Survey, Washington, DC; E. Groshoz, University of New Hampshire; J. Jenkins, Forest Service, USDA, Radnor, PA; J.N. Layne, Archbold Biological Station, Lake Placid, FL; J. Lockwood, University of Tennessee; J.D. Madsen, U.S. Army Crops of Engineers, Vicksburg, MS; R.A. Malecki, N.Y. Cooperative Fish and Wildlife Research Unit, Ithaca, NY; E.L. Mills, Cornell University; S.F. Nates, University of Southwestern Louisiana; H.S. Neufeld, Appalachian State University; P.J. O'Connor, Colorado State University; B.E. Olson, Montana State University; E.F. Pauley, Coastal Carolina University; M. Pimentel, Cornell University; S. Pimm, University of Tennessee; W.J. Poly, Southern Illinois University; G. Roberts, University of Hawaii; G.S. Rodrigues, Empresa Brasilerira de Pesquisa Agropecuaria, Brazil; M. Sagoff, Institute for Philosophy and Public Policy, University of Maryland, College Park; B. Salter, Maryland Department of Natural Resources; D.L.

Scarnecchia, University of Idaho; D. Simberloff, University of Tennessee; J.N. Stuart, University of New Mexico; S.B. Vinson, Texas A&M University; L.A. Wainger, University of Maryland; J.K. Wetterer, Columbia University; and C.E. Williams, Clarion University of Pennsylvania.

References

1. USBC. *Statistical Abstract of the United States 1996,* 200th ed., U.S. Bureau of the Census, U.S. Government Printing Office, Washington, D.C., 1998.
2. OTA. *Harmful Non-Indigenous Species in the United States*, Office of Technology Assessment, Washington, D.C., United States Congress, 1993.
3. Raven, P.H. and Johnson, G.B. *Biology, 3rd edition*, St. Louis, MO, Mosby Year Book, 1992.
4. Nature Conservancy. America's Least Wanted: *Alien Species Invasions of U.S. Ecosystems*, The Nature Conservancy, Arlington, VA, 1996.
5. Wilcove, D.S., et al. Quantifying threats to imperiled species in the United States, *BioScience*, 48, 1998, 607.
6. Armstrong, S., Rare plants protect Cape's water supplies, *New Scientist*, *February 11,* 1995, p. 8.
7. Bryan, R.T. Alien species and emerging infectious diseases: past lessons and future applications, in *Proceedings of the Norway/UN Conference on Alien Species, July 1–5*, Sandlund, G.T., Schel, P.J., and Viken, A. (Eds.), Norwegian Institute for Nature Research, Trondheim, Norway, 1996, 74.
8. Morse, L.E., Kartesz, J.T., and Kutner, L.S. Native vascular plants, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 205, 1995.
9. Morin, N. Vascular plants of the United States, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 200.
10. Frank, J.H. and McCoy, E.D. Introduction to insect behavioral ecology: the good, the bad and the beautiful: non-indigenous species in Florida, *The Florida Entomologist*, 78, 1, 1995.
11. Frank, J.H., et al. Immigration and introduction of insects, in Simberloff, D., Schmitz, D.C., and Brown, T.C. (Eds.), *Strangers in Paradise,* Island Press, Washington, D.C., 1997, 75.
12. Simberloff, D., Schmitz, D.C., and Brown, T.C. *Strangers in Paradise*, Island Press, Washington, D.C., 1997.
13. Dowell, R.V. and Krass, C.J. Exotic pests pose growing problem for California, *California Agriculture*, 46, 1992, 6.
14. Babbitt, B. Statement by Secretary of the Interior on invasive alien species, *Proceedings, National Weed Symposium*, *BLM Weed Page*, April 8–10, 1998.
15. Malecki, R.A., et al. Biological control of purple loosestrife, *BioScience*, 43, 1993, 680.
16. Thompson, D.G., Stuckey, R.L., and Thompson, E.B. *Spread, Impact, and Control of Purple Loosestrife (Lythrum salicaria) in North American Wetland*, U.S. Fish and Wildlife Service, Fish and Wildlife Research, Washington, D.C., 2, 1987.
17. Gaudet, C.L. and Keddy, P.A. Predicting competitive ability from plant traits: a comparative approach, *Nature*, 334, 243, 1998.
18. ATTRA. *Purple Loosestrife: Public Enemy #1 on Federal Lands*, Washington, D.C., ATTRA Interior Helper Internet: http//refuges.fws.gov/NWRSFiles/HabitatMgmt/PestMgmt/LoosestrifeProblem.html.,1997.
19. Hiebert, R.D. and Stubbendieck, J. *Handbook for Ranking Exotic Plants for Management and Control,* U.S. Department of Interior, National Park Service, Denver, CO, 1993.
20. Eldredge, L.G. and Miller, S.E. Numbers of Hawaiian species: supplement 2, Including a review of freshwater invertebrates, *Bishop Museum Occasional Papers no*. 48, 1997, 3.
21. Vitousek, P.M. Diversity and biological invasions of Oceanic Islands, in *Biodiversity,* Wilson, E.O. and Peter, F.M. (Eds.), Washington, D.C., National Academy of Sciences, 1988, 181.

22. Campbell, F.T. Killer pigs, vines, and fungi: alien species threaten native ecosystems, *Endangered Species Technical Bulletin,* 19, 1994, 3.
23. Kurdila, J. The introduction of exotic species into the United States: there goes the neighborhood, *Environmental Affairs*, 16, 1995, 95.
24. Vitousek, P.M., et al. Biological invasions as global environmental change, *American Scientist,* 84, 1996, 468.
25. Vitousek, P.M., et al. Introduced species: a significant component of human-caused global change, *New Zealand Journal of Ecology*, 21, 1, 1997.
26. Whisenant, S.G. Changing Fire Frequencies on Idaho's Snake River Plain: Ecological and Management Implications. *The Station.* General Technical Report INT – U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, 276, 1990.
27. Campbell, F.T. *"Worst" Invasive Plant Species in the Conterminous United States*, Report, Western Ancient Forest Campaign, Springfield, VA, 1998.
28. Miller, J.H. Exotic plants in southern forests: their nature and control, *Proceedings, Southern Weed Science Society*, 48, 1995, 120.
29. Randall, J.M. Weed control for the preservation of biological diversity, *Weed Technology*, 10, 1996, 370.
30. Mills, E.L., et al. Biological Invasions in the Hudson River Basin, *New York State Museum Circular No. 57,* The University of the State of New York, State Education Department, 1997.
31. Center, T.D., Frank, J.H., and Dray, F.A. Biological control, in *Strangers in Paradise*, Simberloff, D., Schmitz, D.C., and Brown, T.C. (Eds.), Island Press, Washington, D.C., 1997, 245.
32. Layne, J. N. Nonindigenous mammals, in *Strangers in Paradise*, Simberloff, D., Schmitz, D.C., and Brown, T.C. (Eds.), Island Press, Washington, D.C., 157, 1997.
33. Pimentel, D. Unpublished data, 1951.
34. Smith, R. Producers need not pay startling "rodent tax" losses, *Feedstuffs*, 56, 1984, 13.
35. USDA. *Agricultural Statistics,* U.S. Department of Agriculture, Washington, D.C., 1998.
36. Wachtel, S.P. and McNeely, J.A. Oh rats, *International Wildlife,* 15, 1985, 20.
37. Chopra, G. Poultry farms, in *Rodents in Indian Agriculture*, Prakash, I. and Ghosh, P.K. (Eds.), Scientific Publishers, Jodhpur, India, 1992, 309.
38. Ahmed, E., Hussain, I., and Brooks, J.E. Losses of stored foods due to rats at grain markets in Pakistan, *International Biodeterioration & Biodegradation*, 36,1995, 125.
39. Richards, C.G. J. The pest status of rodents in the United Kingdom, in *Mammals as Pests,* Putman, R.J. (Ed.), Chapman and Hall, London, 1989, 21.
40. Amarasekare, P. 1993. Potential impact of mammalian nest predators on endemic forest birds of western Mauna Kea, Hawaii, *Conservation Biology,* 7, 1993, 316.
41. Pimentel, D. The control of the mongoose in Puerto Rico, *American Journal of Tropical Medicine and Hygiene*, 41, 1955, 147.
42. Vilella, F.J. and Zwank, P.J. Ecology of the small Indian mongoose in a coastal dry forest of Puerto Rico where sympatric with the Puerto Rican nightjar, *Caribbean Journal of Science,* 29, 1993, 24.
43. Henderson, R.W. 1992. Consequences of predator introductions and habitat destruction on amphibians and reptiles in the post-Columbus West Indies. *Caribbean Journal of Science*, 28, 1992, 1.
44. Everard, C.O.R. and Everard, J.D. Mongoose rabies in the Caribbean, Annals of the New York Academy of Sciences, 653, 1992, 356.
45. Pimentel, D. Unpublished data.
46. Foote, D. Personal communication, 1998.
47. Nassar, R. and Mosier, J. Projections of pet population from census demographic data, *Journal of the American Veterinary Medical Association*, 198, 1991, 1157.
48. Luoma, J.R. Catfight, *Audubon*, 99, 1997, 90.
49. Fitzgerald, B.M. Diet of domestic cats and their impact on prey populations, in *The Domestic Cat: the Biology of Its Behavior,* Turner, D.C. and Bateson, P. (Eds.), Cambridge University Press, Cambridge, 1990, 123.
50. Dunn, E.H. and Tessaglia, D.L. Predation of birds at feeders in winter, *Journal of Field Ornithology,* 65, 1994, 8.

51. McKay, G.M. Feral cats in Australia: origin and impacts. *Unwanted Aliens? Australia's Introduced Animals,* Nature Conservation Council of NSW. The Rocks, NSW, Australia, 1996.
52. Pimentel, D. and Greiner, A. Environmental and soci-economic costs of pesticide use, in *Techniques for Reducing Pesticide Use: Economic and Environmental Benefits*, Pimentel, D. (Ed.), John Wiley & Sons, Chichester, UK 1997, 51.
53. Carter, C.N. Pet population control: another decade without solutions? *Journal of American Veterinary Medicine Association,* 197, 1990, 192.
54. Sacks, J.J., Kresnow, M., and Houston, B. Dog bites: how serious a problem? *Injury Prevention,* 2, 1996, 52.
55. Colburn, D. Dogs take a big bite out of health care costs, *The Washington Post*, February 2, 1999, z5.
56. CDC. Summary of Notifiable Diseases, United States, 1995, Mortality and Morbidity Weekly Report of the Communicable Disease Center, 44, 1996, 1.
57. Temple, S.A. Exotic birds, a growing problem with no easy solution. *The Auk*, 109, 1992, 395.
58. Moulton, M.P. and Pimm, S.L. The introduced Hawaiian avifauna: biogeographic evidence for competition, *The American Naturalist,* 121, 1983, 669.
59. Pimm, S.L. *The Balance of Nature?* The University of Chicago Press, Chicago, 1991.
60. Laycock, G. *The Alien Animals,* Natural History Press, New York, 1966.
61. Roots C. *Animal Invaders*, Universe Books, New York, 1976.
62. Long, J. L. Introduced Birds of the World: The Worldwide History, Distribution, and Influence of Birds Introduced to New Environments, Universe Books, New York, 1981.
63. Weber, W.J. *Health Hazards from Pigeons, Starlings and English Sparrows: Diseases and Parasites Associated with Pigeons, Starlings, and English Sparrows which Affect Domestic Animals*, Thomson Publications, Fresno, CA, 1979.
64. Robbins, C.S. Non-native birds, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems,* LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 437.
65. Smith, R.H. Rodents and birds as invaders of stored-grain ecosystems, *Books in Soils, Plants, and the Environment: Stored-Grain Ecosystems*, in Jayas, D.S., White, N.D.G., and Muir, W.E. (Eds.), Marcel Dekker, New York, 1992, 289.
66. Haag-Wackernagel, D. Regulation of the street pigeon in Basel, *Wildlife Society Bulletin,* 23, 1995, 256.
67. Johnston, R.F. and Janiga, M. *Feral Pigeons*, Oxford University Press, 1995.
68. McCoid, M.J. and Kleberg, C. Non-native reptiles and amphibians, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 433.
69. Lafferty, K.D. and Page, C.J. Predation of the endangered tidewater goby, *Eucyclogobius newberryi,* by the introduced African clawed frog, *Xenopus laevis*, with notes on the frog's parasites, *Copeia*, 3, 592, 1997.
70. Fritts, T.H. and Rodda, G.H. Invasions of the brown tree snake, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 454.
71. Rodda, G.H., Fritts, T.H., and Chiszar.D. The disappearance of Guam's wildlife. *BioScience*, 47, 1997, 565.
72. Teodosio, R. Tree snake brings Guam blackouts, *Pacific Magazine*, 12, 1987, 42.
73. Coulehan, K. Powerless again. About your partners in business: snales and GPA. *Guam Business News*, January 1987, 13.
74. BTSCP. *Brown Tree Snake Control Plan,* Brown Tree Snake Control Committee, Aquatic Nuisance Species Task Force, Honolulu, HI, 1996.
75. Fritts, T. Personal communication, 1998.
76. Fritts, T. 1998 Federal budget, 1998.
77. Holt, A. Hawaii's reptilian nightmare, *World Conservation*, 4/97 – 1/98: 31, 1997.

78. Courtenay, W.R., Jennings, D.P., and Williams, J.D. Appendix 2. Exotic fishes of the United States and Canada, in *A List of Common and Scientific Names of Fishes from the United States and Canada*, Robins C.R. (Ed.), Special Publication 20. American Fisheries Society, Bethesda, MD, 1991.
79. Courtenay, W.R. Biological pollution through fish introductions, in McKnight, B.N. (Ed.), *Biological Pollution: The Control and Impact of Invasive Exotic Species,* Indiana Academy of Science, Indianapolis, 1993, 35.
80. Courtenay, W.R. Nonindigenous fishes, in *Strangers in Paradise,* Simberloff, D., Schmitz, D.C., Brown, T.C. (Eds.), Island Press, Washington, D.C., 1997, 109.
81. Dill, W.A. and Cordone, A.J. History and Status of Introduced Fishes in California, 1871–1996, *Fish Bulletin* 178, State of California, The Resources Agency, Department of Fish and Game, 1997.
82. Maciolek, J.A. Exotic fishes in Hawaii and other islands of Oceania, in *Distribution, Biology, and Management of Exotic Fishes,* Courtenay, W.R. and Stauffer, J.R. (Eds.), Johns Hopkins University Press, Baltimore, 1984, 131.
83. Wilcove, D.S. and Bean, M.J. *The Big Kill: Declining Biodiversity in America's Lakes and Rivers,* Environmental Defense Fund, Washington, D.C., 1994.
84. Taylor, J.N., Courtenay, W.R., and McCann, J.A. Known impacts of exotic fishes in the continental United States in *Distribution, Biology, and Management of Exotic Fishes,* Jayas, D.S., White, N.D.G., and Muir, W.E. (Eds.), Johns Hopkins University Press, Baltimore, 1984, 322.
85. Bjergo, C., Boydstun, C., Crosby, M., and Kokkanakis, S., Sayers, R. Non-native aquatic species in the United States and coastal water, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 428.
86. Hendrix, P.F. 1995, *Earthworm Ecology and Biogeography*, Lewis Publishers, Boca Raton.
87. Jenkins, J.C. *Measuring and Modeling Northeaster Forest Response to Environmental Stresses*. Ph.D. Dissertation Submitted to the University of New Hampshire, Durham, NH, 1998.
88. Alsop, F.J. and Laughlin, T.F. Changes in the spruce-fir avifauna of Mt. Guyot, TN, 1967–1985, *Journal of the Tennessee Academy of Science*, 66, 1991, 207.
89. Neufeld, H.S. Personal communication, 1998.
90. Vinson, S.B. Impact of the invasion of *Solenopsis invicta* (Buren) on native food webs, in *Exotic Ants: Biology, Impact, and Control of Introduced Species*, Williams, D.F. (Ed.), Boulder, CO, Westview Press, 1994, 241.
91. Allen, C.R., Lutz, R.S., and Demarais, S. Red imported fire ant impacts on northern bobwhite populations, *Ecological Applications*, 5, 1995, 632.
92. Vinson, S.B. The economic impact of the imported fire ant infestation on the State of Texas, Report, Texas A&M University, College Station, TX, 1992.
93. TAES. *Texas Imported Fire Ant Research & Management Plan,* Report. Texas Agricultural Extension Service, Texas A&M University, College Station TX, 1998.
94. Corn, M.L., et al. *Harmful Non-Native Species: Issues for Congress*, Congressional Research Service, Library of Congress, Washington, D.C., 1999.
95. Lafferty, K.D, and Kuris, A.M. Biological control of marine pests, *Ecology*, 77, 1996.
96. Benson, A.J. and Boydstun, C.P. Invasion of the zebra mussel into the United States, in Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 445.
97. Keniry T, and Marsden,J.E. Zebra mussels in Southwestern Lake Michigan, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems,* LaRoe E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 445.
98. Griffiths, D.W., et al. Distribution and dispersal of the zebra mussel (*Dreissena polymorpha*) in the Great Lakes Region, *Canadian Journal of Fishery and Aquatic Science*, 48, 1381, 1991.

99. Khalanski, M. Industrial and ecological consequences of the introduction of new species in continental aquatic ecosystems: the zebra mussel and other invasive species, *Bulletin Francais de la Peche et de la Pisciculture*, 0, 1997, 385.
100. Isom, B.G. ASTM (American Society for Testing and Materials) Special Technical Publication, 894. Rationale for Sampling and Interpretation of Ecological Data in the Assessment of Freshwater Ecosystems, American Society for Testing and Materials, Philadelphia, PA, 1986.
101. Cohen, A.N. and Carlton, J.T. *Nonindigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta*, United States Fish and Wildlife Service, Washington, D.C., 1995.
102. Pimentel, D., et al. Benefits and risks of genetic engineering in agriculture, *BioScience*, 39, 1989, 606.
103. Pimentel, D. Habitat factors in new pest invasions, *Evolution of Insect Pests — Patterns of Variation*, in Kim, K.C. and McPheron, B.A. (Eds.), John Wiley & Sons, New York, 1993, 165.
104. Pimentel, D. Techniques for Reducing Pesticides: Environmental and Economic Benefits, John Wiley & Sons, Chichester, UK, 1997.
105. Pimentel, D. *Handbook on Pest Management in Agriculture*.Volumes 1, 2, and 3. Boca Raton, FL, CRC Press, 1991.
106. Trammel, M.A. and Butler, J.L. Effects of exotic plants on native ungulate use of habitat, *Journal of Wildlife Management*, 59, 808, 1995.
107. Dewey, S.A. Weedy thistles of the western USA, Westview Special Studies in Agricultural Science and Policy Noxious Range Weeds National Noxious Range Weed Conference on a Forum for Continuing Cooperation, Westview Press, Boulder, CO, 1991.
108. Templeton S.R., Zilberman, and D., Yoo, S.J. An economic perspective on outdoor residential pesticide use in *Environmental Science & Technology*, 1998.
109. Richardson, Curtis J. Personal communication, 1998.
110. Pogacnik, T. Wild horses and burros on Public lands, in t*o the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe E.T. (Ed.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 456.
111. Rosentreter, R. Displacement of rare plants by exotic grasses in Monsen, S.B. and Kitchen, S.G. (Eds.), *Proceedings — Ecology and Management of Annual Rangeland*s. USDA, Forest Service, Rocky Mountain Research Station, Washington, D.C., 1994, 170.
112. Pimentel, D., Lach, L., Zuniga, R., and Morrison, D. Environmental and economic costs associated with introduced non-native species in the United States. Manuscript, 1999.
113. Kotanen, P.M. Responses of vegetation to a changing regime of disturbance: effects of feral pigs in a California coastal prairie, *Ecography*, 18, 190, 1995.
114. Stone, C.P., Cuddihy, L.W., and Tunison, T. Response of Hawaiian ecosystems to removal of pigs and goats, in *Alien Plant Invasions on Native Ecosystems in Hawaii: Management and Research*, University of Hawaii Cooperative National Park Studies Unit, Honolulu, 1992, 666.
115. Zuniga, P.C.R. Personal communication, 1999.
116. Bach, J. P. Personal communication, 1999.
117. Rollins, D. *Statewide Attitude Survey on Feral Hogs in Texas*, Texas Agricultural Extension Service, Texas, 1998.
118. Davis, D.S. *Feral Hogs and Disease: Implications for Humans and Livestock*, Department of Veterinary Pathology, Texas A&M University, College Station, TX, 1998.
119. Bach, J.P. Personal communication, 1999.
120. Moore, N.W. How many wild birds should farmland support? in *Bird Problems in Agriculture*, Wright, E.N., Inglis, I.R., and Feare, C.J. (Eds.), Croydon, UK, The British Crop Protection Council, 1980, 2.
121. Feare, C.J. The economics of starling damage, in *Bird Problems in Agriculture*, Wright, E.N., Inglis, I.R., and Feare, C.J. (Eds.), Croydon, UK: The British Crop Protection Council, 1980, 39.
122. Howarth, F.G. Hawaiian terrestrial arthropods: an overview, *Bishop Museum Occasional Papers*, 30, 1990, 4.

123. Beardsley, J.W. Introduction of arthropod pests into the Hawaiian Islands, *Micronesia Supplement,* 3, 1991,1.
124. Frank, J.H. and McCoy, E.D. Precinctive insect species in Florida, *The Florida Entomologist*, 78, 21, 1995.
125. Liebold, A.M., et al. Invasion by exotic forest pests: a threat to forest ecosystems, *Forest Science*, 41, 1995, 1.
126. Hall, J.P. and Moody, B. *Forest Depletions Caused by Insects and Diseases in Canada 1982–1987*. Forest Insect and Disease Survey Information Report ST-X-8, Ottawa, Canada: Forest Insect and Disease Survey, Canadian Forest Service, Natural Resources Canada, 1994.
127. USDA. *Index of Plant Diseases in the United States,* Crop Research Division, ARS, U.S. Department of Agriculture, Washington, D.C., 1960.
128. Campbell, F.T. and Schlarbaum, S.E. *Fading Forests: North American Trees and the Threat of Exotic Pests,* Natural Resources Defense Council, New York, 1994.
129. Drummond, R.O., et al. Estimated losses of livestock to pests, in *Handbook of Pest Management in Agriculture,* Pimentel, D. (Ed.), CRC Press, Boca Raton, FL, 1981, 111.
130. Morgan, N.O. Potential impact of alien arthropod pests and vectors of animal diseases on the U.S. livestock industry, in *Handbook of Pest Management in Agriculture*, Pimentel D. (Ed.), Boca Raton, FL: CRC Press, 1981, 129.
131. USAHA. *Foreign Animal Diseases: Their Prevention, Diagnosis and Control,* Committee on Foreign Animal Diseases of the United States Animal Health Association, Richmond, VA, 1984.
132. Hart, K. Personal communication, 1999.
133. Newton-John, H. Exotic human diseases, *in Pests and Parasites as Migrants,* Gibbs A.J., Meischke H.R.C. (Eds.), Sydney, Cambridge University Press, 1985, 23.
134. USPHS. *For a Healthy Nation: Returns on Investments in Public Health*, U.S. Department of Health and Human Services, Public Health Service, Washington, D.C., 1994.
135. Chapman, L.E., et al. Influenza — United States, 1989–90 and 1990–91 seasons, *Mortality and Morbidity Weekly Report Surveillance Summaries*, 41,1992, 35.
136. Bateman D.A., et al. The hospital cost of congenital syphilis, *Journal of Pediatrics*, 130, 1997, 752.
137. FIWC. *Pulling Together: National Strategy for Invasive Plant Management.* Wednesday March 10, 1999. hhtp://bluegoose.arw.r9.fws.gov/ficmnewfiles/NatlweedStrategytoc.html.
138. Babbitt, B. 1999. *Weed Coalition Announces National Strategy to Combat the Spread of Non-Native Invasive Plants,* Wednesday, March 10, 1999, U.S. Department of the Interior, Washington, D.C.

chapter fifteen

Alien invertebrate animals in South Africa

Lori Lach, M.D. Picker, J.F. Colville, M.H. Allsopp, and C.L. Griffiths

Contents

15.1 Introduction

Though the history of plant and vertebrate introductions to South Africa is fairly well known,[1] invertebrate introductions have received comparatively little attention. Insects and other invertebrates have been introduced both intentionally and accidentally. In terrestrial systems, about 56 insects have been introduced for biological control of weeds, and at least 54% have caused some documented damage to the target weed.[2] The impacts of these introduced insects on non-target organisms, however, are not well studied.

0-8493-0836-4/02/$0.00+$1.50

Alien invasive invertebrates that cause large-scale negative impacts on conservation areas and on the welfare of humans and animals either have yet to invade South Africa, are escaping notice, or have not been investigated. For example, Braack et al.[3] reviewed the status of 21 exotic arthropods recorded in Kruger National Park and noted that none had caused serious consequences. De Moor[4] failed to find any alien aquatic insects that had established in South Africa. There are exceptions, however. The Argentine ant has invaded fynbos areas and threatens protea conservation; the varroa mite poses a serious threat to commercial and wild honeybees; and the freshwater snail *Lymnaea columella* is facilitating the spread of livestock disease.

Because they are more directly tied to economic output, introduced invertebrates that establish in crops have received more attention. Of the 40 top crop pests in the country, 42% are non-native.[5] Many introduced crop pests in South Africa, including the red scale (*Aonidiella aurantii*), Mediterranean fruit fly (*Ceratitus capitata*), and codling moth (*Cydia pomonella*), are well-known pests from elsewhere in the world.[6] Pimentel et al.[7] estimate that crop losses due to introduced arthropods in South Africa amount to $1 billion per year (U.S. dollars).

Here we review the history, distribution, and impacts of six invertebrate invaders in South Africa. These case studies were selected to illustrate the range of impacts, natural history traits, and habitats of the alien invaders.

15.2 Argentine ant (Linepithema humile)

Of the 16 pest ant species in South Africa, two, *Technomyrmex albipes* and *Linepithema humile*, are non-native, and the latter is by far the more damaging.[8] *Linepithema humile*, also known as the Argentine ant, a native of Argentina and Brazil, is thought to have arrived in South Africa around 1898[9] in shipments of cattle fodder from South America.[10] Globally, its introduced range tends to occur between 30° and 36°N latitude (e.g., Bermuda, Mediterranean Europe, southern and western United States, Chile, Western Australia, New Zealand) or in the higher altitudes of tropical locations, such as the Hawaiian Islands.[11] In South Africa it has been reported throughout much of the Cape Peninsula and the southern coast of the Western and Eastern Cape Provinces as far east as East London, and as far north as Clanwilliam,[9,10,12–15] as well as in Johannesburg and Pretoria households and other buildings.[8,10,13,16] A 1984 survey of Western Cape fynbos, the heath-like shrub vegetation indigenous to mountains and lowland areas of southwest Africa, found *L. humile* in 30 of 175 sites surveyed. The ants were more likely to be found in lower-altitude areas and those accessible by paved roads; the invader was never found in sites that could be reached only on foot.[14]

15.2.1 Environmental impacts

Argentine ants are noted for their ability to displace native ants throughout their introduced range.[17–22] In South Africa they have been shown to eliminate other dominant ants, including *Anoplolepis custodiens, A. steingroeveri,* and *Pheidole capensis*.[23,24] Evidence from other sites they have invaded shows that their highly aggressive nature, chemical defense, high abundance, and superior exploitative and interference competition have also played roles in their elimination of native ants.[25,26] Some South African ant species, such as *Tetramorium quadrispinosum, Ocymyrmex cilliei, Meranoplus perengueyi, T. simillimum,* and *Oligomyrmex* spp., are able to coexist with *L. humile* because of interspecific differences in temperature-dependent foraging patterns, chemical defenses, or small body size and stealthy behavior.[24,27,28]

Displacement of native ants can have cascading effects on other community members, including plants that rely on ants for seed dispersal, known as myrmecochores. Native ants, such as *A. custodiens* and *P. capensis*, are attracted to a myrmecochorous seed by its eliaosome, an oily, fleshy attachment. By transporting the seeds to their underground nests, these ants protect them from fire and from predation by rodents. Argentine ants displace both of these species and disrupt this mutualism by eating the eliaosome and leaving the seed exposed.[29] More than 170 fynbos plant species rely on ants for dispersal of their seeds, and it is thought that the Argentine ant invasion may pose a serious threat to their long-term survival.[29] Myrmecophilous lycaenid butterfly larvae may also be at risk. South Africa is home to 407 species of lycaenid butterflies. Of the 156 for which some life history information has been published, 94 (60%) are obligate myrmecophiles, and another 28 (18%) have facultative relationships with ants.[30] Some of these, including the mountain copper (*Aloeides thyra*), the foxtrot copper (*Phasis thero*), and the Boland thestor (*Thestor protumnus*), occur exclusively in the Western Cape Province and have relationships with ants that are displaced by Argentine ants.[31,32] Research is needed to determine the extent of overlap between *L. humile* and lycaenids that rely on ants. Observations suggest that *L. humile* would not fulfill the tending roles of the ants they displace.[33]

Argentine ants also may be affecting the fynbos through impacts on pollinators and pollination. Visser et al.[34] found that arthropod abundance and richness in insect-pollinated *Protea nitida* flowerheads were significantly lower when Argentine ants were present in high numbers than when they were absent. Recently, an exclusion experiment has shown that Argentine ants significantly reduce seed set in *P. nitida*.[35] Moreover, visitation of the Argentine ants to *P. nitida* flowerheads is facilitated by the presence of honeydew-producing membracids that occur on branches below the flowerheads.[35] Native ants recruit to flowerheads in lower numbers, and their visitation is not affected by membracids.

Research on the impacts of Argentine ants on other ground-dwelling insects in South Africa has yet to be done. Reports from other areas where Argentines have invaded have produced mixed results; some show a lower abundance of native arthropods and/or a higher abundance of scavenging or armored insects in invaded versus uninvaded areas,[21,36,37] while Holway[38] found no differences in non-ant arthropod abundance and richness between invaded and uninvaded sites. Argentine ants are known to prey on a variety of insects, including Coleoptera, Diptera, Hymenoptera, Lepidoptera, and Neuroptera.[39–43]

15.2.2 Agricultural and household impacts

Most documented impacts of *L. humile* on South African crop production involve its relationships with Homoptera. Argentine ants are highly attracted to the honeydew produced by mealybugs, scale, and aphids. In exchange for the honeydew, they protect the Homoptera from their natural enemies, sometimes to the extent of building shelters for them.[9] The tending ability results in high numbers of mealybugs in vineyards,[15] and of various coccids, pseudococcids, and aphids in citrus and guava.[9,44] Red scale (*Aonidiella aurantii*) in citrus, though not a honeydew producer, benefits indirectly from the presence of Argentine ants.[45] The Argentine ant's limited distribution in South Africa keeps it from being the most important ant pest on a national level, but it is an economically significant species in Western Cape agriculture.[44] Although there has not been an economic assessment of the damage it causes via its interaction with Homoptera, it is well known that attempts to control the Homoptera without controlling the ants that tend them are pointless.[44]

Linepithema humile may also be having an impact on crop pollination. Potgieter[46] and Durr[9] both contend that the presence of Argentine ants on various orchard blossoms deters pollinators. Honeybees (*Apis mellifera*) are important pollinators, and Buys[47] observed that bees that visit flowers with ants are more "nervous" and may spend less time visiting. Moreover, *L. humile* is known to attack and destroy beehives in South Africa[47] and elsewhere it has invaded,[48] prompting the need for beekeepers to invest in control measures. Also, because it can forage at night when bees do not, *L. humile* can effectively outcompete honeybees for nectar; Buys[49] found that *L. humile* took 42% of the nectar from *Eucalyptus sideroxylon* flowers before honeybees began foraging in the morning.

Argentine ants are also known to invade households to forage for food and water.[8,9,46] In Cape Town and surrounding areas, it may be in the process of being displaced by the native ant *Lepisiota capensis*.[50]

15.3 Varroa mite (Varroa destructor)

The ectoparasitic varroa mite *Varroa destructor*,[51] formerly *Varroa jacobsoni*, is the most serious pest of honeybees to arise in the 20th century. Relatively harmless on its natural host, the Eastern honeybee *Apis cerana*, the varroa mite has recently crossed onto the Western honeybee *Apis mellifera* and spread from its Asian origins throughout most of the world.[52] On the commercially important *Apis mellifera*, the varroa mite is no longer a relatively benign pest, resulting in the death of the honeybee colony in almost all cases.[53]

Varroa destructor is reddish-brown in color and flat and oval in shape, measuring 1.1 mm long and 1.5 mm wide. It is found between the abdominal segments or body regions of adult bees, particularly adjacent to the wax glands, or in the honeybee brood. Fertilized female mites invade a cell containing a bee larva, and once the cell is capped, feed on the haemolymph of the larva and lay eggs in the cell. The first egg laid typically develops into a male, and all subsequent eggs into female mites.[54] The young mites develop in 6 to 8 days, the male offspring mates with his sisters in the cell and then dies, and the mother mite and her mated daughters leave the cell when the young bee emerges.[55] Drone brood is more attractive to the mites than worker brood.[56] Adult mites live off the blood of adult bees and have a lifespan of 2 to 3 months; a female may produce two, or at most three, broods.

The varroa mite was absent from sub-Saharan Africa until it was found in South Africa in August 1997.[57] In late 1997 it was found only on the Cape peninsula, but it has subsequently spread over most of the country and is currently found in all provinces of South Africa.[58] It can be reliably predicted that the mite will be present in all honeybee colonies in South Africa within 3 to 5 years, and thereafter will spread through the honeybee colonies of neighboring countries. The mite spreads between honeybee colonies through the drifting of foraging bees and drones, and the robbing of hives by foraging bees. Migrating honeybee swarms, and especially the commercial migration of bees by beekeepers, result in the rapid wider dispersal of the mite.

15.3.1 Environmental and agricultural impacts

Large numbers of mites cause brood death, while lesser infestations result in abnormalities in the brood, typically compacted size and vestigial wings. The combination of adult mortality, brood mortality, and brood abnormalities weakens the colony and eventually causes its death. A role for viruses has been indicated in the final mortality of the colony.[59] In regions of the world where the varroa mite is well established, such as Europe and the United States, wild honeybee populations have all but disappeared as a result of varroa mortality.[53]

There are good reasons to believe that the varroa mite will not be as virulent to African honeybees as has been the case in Europe and the United States. Africanized honeybees in Brazil have shown considerable tolerance to varroa,[60] as have bees in North Africa.[61] This tolerance of African honeybees to varroa is presumed to be due to the shorter post-capping developmental period found in African honeybees, which limits the time available for reproduction of the mite, resulting in large numbers of infertile mites[55,62] and preventing mite numbers from reaching lethal proportions. African honeybees also exhibit active defense against the mites.[63] Environmental conditions also seem to influence the virulence of the varroa mite, with the mite being more dangerous under cooler, temperate conditions than under tropical conditions.[55]

However, and in contrast to the Americas, perhaps the greatest threat of varroa in Africa is to the wild honeybee populations that pollinate as much as 40 to 50% of the country's indigenous flowering plants,[64] although very little specific data on the actual contribution of honeybees in this regard are available. Should South Africa and the rest of Africa suffer the losses of wild bees witnessed in other parts of the world,[65–67] there may be very significant implications for floral conservation and biodiversity in Africa. Preliminary results indicate that the mite is already present in honeybee colonies in many conservation regions of South Africa,[68] and it is certain to spread throughout the country. As wild honeybee colonies are obviously beyond the reach of chemical treatment, little can be done to salvage this situation should varroa mites prove to be extremely destructive in Africa. The possible ecological consequences of such an event are alarming. Should the honeybee population be lost due to varroa, the prospect of ecological damage, including the possible loss of plant species due to lack of adequate pollination, is not inconceivable, with possible consequent loss of animal species.

The varroa mite can be readily controlled in commercial honeybee colonies by application of commercial varroacides, normally pyrethroids. Two such varroacides have been registered for use in South Africa. Varroa mite populations in commercial colonies can also be successfully controlled by improved colony management. Wild honeybee colonies and honeybee colonies used by most small-scale beekeepers in Africa cannot, however, be treated with commercial varroacides. No natural enemies to varroa mites have thus far been identified, and no successful biological control agents have thus far been developed.[69]

15.3.2 Economic impacts

Early indications are that the varroa mite is proving to be extremely damaging to honeybee colonies in South Africa, and significant colony mortality has already occurred.[68] The value of commercial honeybees in South Africa has recently been calculated at approximately ZAR 3.2 billion (South African rand) per year, the vast majority of this attributable to the commercial pollination of a multitude of crop plants. This crop production, and the approximately 300,000 jobs it supports, are directly threatened by the potential impact of varroa mites. It has been shown that varroa-infested colonies are less effective commercial pollinators than uninfected colonies.[68]

The direct agricultural impact of the varroa mite will be to make beekeeping more difficult and more expensive, to reduce the numbers of wild honeybees available to beekeepers, and to reduce the availability and efficiency of honeybee colonies available for commercial crop pollination. These costs will inevitably be passed on to consumers. Commercial beekeeping and commercial crop production will suffer losses but will survive with the judicious use of pesticides. In addition, the varroa mite is certain to have a disastrous impact on small-scale rural beekeeping in Africa, and on beekeeping development programs.

15.4 Spotted stem borer (*Chilo partellus partellus*)

The history of the appearance and spread of the spotted stem borer *Chilo partellus* in Africa is summarized by Kfir.[70,71] Native to Asia and the Indian subcontinent,[72] this alien was first noticed in 1932 in Malawi, in Tanzania in 1952, and in South Africa in 1958.[73] It soon became dominant in the more humid coastal regions of KwaZulu-Natal and the southern and western parts of Mpumalanga. More recently, it has invaded the higher-altitude grain-growing areas of the Highveld, where it is gradually displacing *Busseola fusca*, a native stem borer.[74]

15.4.1 Agricultural impacts

Chilo partellus and the native pest species *B. fusca* are the two most important pests of maize and grain sorghum in South Africa.[75] Adults occur in early summer (September and October), and again in late summer (February and March). These overlapping generations (with a mean of 2.5 generations during the summer period) ensure that all stages of the life cycle are present at all times during the summer. First instar larvae disperse on silken threads, and then bore into and damage culms and tillers of maize and sorghum. Mature larvae overwinter in the dry cereal stalks and emerge once temperatures rise. Damage to sorghum is maximal if the larvae infest the piping stage of the grain (during the first 40 days of growth); planting after the first adult peak reduces infestation rates somewhat.[6]

Crop losses of maize and sorghum to the combined action of the cereal stem borers *C. partellus* and *B. fusca* range between 10 and 100%.[74] In a mixed population, where *C. partellus* levels were moderate to severe and *B. fusca* levels were low, experimental yield losses of untreated plots (as compared to sprayed control plots) were 3.29 tons/ha or 21% for maize and 4.4 tons/ha or 58% for grain sorghum.[76] Where sorghum is attacked by both species, *C. partellus* has been found to be the more injurious of the two.[77] Since the 1970s, *C. partellus* has increased its proportion in mixed infestations from 10 to 90%.[74]

15.4.2 Economic impacts

Despite the fact that the mean annual value of maize and sorghum, ZAR 8 billion, far exceeds the value of any other agricultural product in South Africa,[78] very little has been published on the yield-loss or economic-loss estimates resulting from damage by Lepidoptera to crops. Moran,[79] using a formula that incorporated various ecological, agricultural, and economic parameters associated with crop pests, ranked *C. partellus* 44th out of the 101 most important crop pests, of which just over half were native species. A more recent ranking, which incorporates research efforts dedicated to each species as an indirect estimate of its economic worth, lists *C. partellus* as the seventh most important lepidopteran pest in South Africa.[80]

Along with yield losses, control costs can be high. For grain sorghum, pesticidal control of *B. fusca* and *C. partellus* accounted for 67% and 39% of the total cost of pest control for the 1987–88 and 1988–89 seasons, respectively.[81] In an epidemic year, as much as $7 million might be spent on insecticides for stem borers, but in an average year the figure is about $2.5 million.[82]

Insecticides have little effect on the pest, though, as they are only effective against the first instar larvae (overlapping generations and reinfection aggravate the problem), and they are also not always cost-effective.[70] In light of the limited efficacy of pesticide regulation of stem borers, and the lack of financial resources on the part of many farming communities and their reluctance to use pesticides on subsistence crops, interest has focused on regulation by habitat management (tillage control and interplanting effects), and the use of parasitoids. The agricultural practice of slashing the stubble with subsequent plowing and cutting up the remains has been found to reduce *C. partellus* populations on maize by 89% and on sorghum by 94%.[83] Blocks of napier grass have been shown to reduce the infestation of nearby maize by acting as a trap crop.[84]

None of the 13 parasitoids introduced for biological control of stem borers has become established. Nine different indigenous parasitoids and hyperparasitoids have been recorded from eggs and caterpillars of *C. partellus*, but they are apparently incapable of stopping the annual population increase of the moth, and they will not attack aestivating larvae.[71] These parasitoids do not, even in combination with predators (for example, the army ant *Dorylus helvolus*) and pathogens, suppress population densities below economic damage levels (taken as 10% of plants showing visible *C. partellus* damage).[70]

15.5 Red scale (*Aonidiella aurantii*)

The red scale *(Aonidiella aurantii)* probably originated in central Asia and India, but today it is a cosmopolitan pest of citrus. It was probably introduced into South Africa at least a century ago and is currently found wherever citrus is grown.[85]

The adult female lives under a circular, dull yellow or reddish-brown scale. The young (crawlers) escape from beneath the body of the mother and move toward the brighter, terminal branches of the host tree, where they infest fruit, leaves, twigs, and even branches.[6] They soon become sessile and molt, a period when the scale is free from the body of the insect. In the warmer areas of South Africa, such as Mpumalanga, the period between molts is shorter, and three broods per year are possible.

15.5.1 Agricultural impacts

Globally, more money has probably been spent on control and research of red scale than on any other agricultural pest species.[6] In South Africa, red scale is found on a variety of fruit and ornamental trees, but heavy infestations are generally encountered only on citrus and roses. Losses to trees can range in severity: light infestations that result in rejection of fruit, mostly for export; moderate infestations that result in larger levels of fruit rejection, and may damage trees through leaf and fruit drop; and heavy infestations that result in unsalable fruit and in defoliation, dieback, and occasionally the deaths of trees.[6]

Damage to citrus crops from red scale is tied to the history of pesticide use. With the advent of organophosphate pesticides in 1948, pests of citrus were regulated solely by spraying. By 1974, red scale had developed organophosphate resistance, and the mounting costs of insecticides were severely eroding profit margins. The application of miscible oils (1 to 1.4%) to infected trees in place of organophosphates resulted in resurgences of parasitoids (especially the native *Aphytis africanus* and other imported species of *Aphytis*) and predators of red scale. This resulted in a reduction in scale infestations, and record-breaking harvests.[6] The current system of integrated pest management effectively regulates red scale, but requires constant inspection for growth of scale populations, accurate timing of chemical application, and development of the appropriate set of microclimatic conditions that favor the biocontrol agents (for example, control of ant populations in orchards by excluding their access to trees,[86]) as well as releasing mass-reared biocontrol agents.[87]

15.5.2 Economic impacts

Economic damages associated with red scale peaked with the pest's development of resistance to organophosphates and the absence of natural enemies. The visual requirements for export fruit are strict; even small infestations of red scale are not tolerated. Infested fruit are culled, resulting in major economic losses. For the period 1966–67, damage from red scale accounted for 12.6% of the 24.6% of fruit crop that was not acceptable for export.[88] At that time, nearly ZAR 1 million were spent on the material costs of chemical control at a single citrus estate.[89] The switch from organophosphates to miscible oils resulted in a sixfold reduction in the cost of spraying, and more than doubled the amount of exported fruit.

15.6 Snail pests (*Physa acuta* and *Lymnaea columella*)

To date, of the seven known snails introduced to South Africa, two have become invasive.[90] The two, *Physa acuta* and *Lymnaea columella*, are currently increasing their distributions across southern Africa.[90–92] Both species are likely North American natives that were introduced via the aquarium trade.[93]

Lymnaea columella was first recorded in the western Cape Province in the early 1940s[91] and has since then invaded every major river and catchment system in South Africa.[92–94] Initial introductions were via all major ports, after which the snail was transported to the Gauteng region, possibly in aquarium plants; once in this region, it entered the Vaal River system and from there spread farther along the Orange River.[91] Updated distribution patterns indicate a widespread occurrence throughout South African rivers and catchment systems, with higher concentrations around large urban centers.[92,93] Several natural history factors likely have contributed to *L. columella's* invasiveness. It tolerates a wide range of water temperatures and habitat, and it can be found in moist mud and vegetation, rivers, streams, lakes, and even artificial water sources such as drinking troughs. Moreover, it is an oviparous hermaphrodite capable of self-fertilization.[91] It is a moderate-sized, spirally coiled dextral snail.[93]

Physa acuta was first introduced in the early 1950s, but it has likely been introduced repeatedly, considering its concentrated distribution patterns around the major ports and cities.[90] Since the first introductions, this snail has spread and become established in most South African rivers and catchments.[90,92,93] *Physa acuta* exhibits a number of attributes of a good invader species. These include high recolonization rates of disturbed habitats, high dispersal rates, high reproductive efficiency (year-round breeding, early reproductive maturity, and short incubation periods and mean generation times),[95] rapid doubling time, and low susceptibility to attack by indigenous predators.[90,96] It is a translucent, medium-sized sinistral snail with variable shell shape.[93]

15.6.1 Environmental and economic impacts

Studies that adequately document the ecological impacts of either of these species on the indigenous invertebrate fauna are lacking. Both species are thought to be sufficiently abundant to have caused impacts on indigenous fauna, but further research is needed.[93] In its submerged habitat, *L. columella* is found in association with several indigenous snail species, including *Bulinus africanus, Biomphalaria pfeifferi,* and *L. natalensis. Physa acuta* has been observed preying on an adult *B. africanus*.[97]

Lymnaea columella poses a serious threat to the health of livestock and wild ungulates. It is a known intermediate host of the liver flukes *Fasciola hepatica* and *F. gigantica*, which infect both domestic stock and wild ungulates, but rarely humans,[93] and it has been blamed for the observed increase in fascioliasis recorded in Zimbabwe and South Africa.[94] The increase may be attributed to the presence of the snail in sites typically uninhabited by indigenous hosts of the disease. For example, the presence of *L. columella* in drinking troughs and freshwater sources may increase risk of transmission to livestock. Since *L.*

columella can survive out of water, it may increase the transmission of fascioliasis by moving onto pastures, allowing the parasite, once emerged, to encyst on grass stalks that may subsequently be eaten by cattle, the parasite's final host.

The economic impacts of increased fascioliasis in South Africa are considered to be high, as fascioliasis can cause considerable financial loss to farmers.[93] The extent of this financial loss is unknown, since no quantitative assessment of the economic loss to livestock farmers due to fascioliasis has been undertaken.[93] *Physa acuta* may act as an intermediate host of *Angiostrongylus cantonensis*, a rat lung nematode that is common in Southeast Asia and can cause eosinophilic meningitis in humans.[93] Although a large number of individuals have been screened, no patent human trematode infections of *P. acuta* have been found in South Africa.[90]

15.7 Mediterranean mussel (Mytilus galloprovincialis)

The most widespread and successful marine alien in the region, the Mediterranean mussel *Mytilus galloprovincialis*, was first recorded in South Africa by Grant et al.,[98] by which time it had already established extensive populations along the west coast between Cape Point and Luderitz, in Namibia. The exact date and site of introduction are unknown, but circumstantial evidence suggests that it was recent, probably in the late 1970s, and was mediated by man.[99,100] By the early 1990s *M. galloprovincialis* had spread as far east as Port Alfred, in the southeastern Cape, and was the dominant intertidal organism along the entire west coast, with an estimated biomass of some 194 tons wet mass per km of rocky coast.[101] Unpublished data suggest that standing stocks on the west coast have subsequently increased substantially, and that the species continues to spread eastward at a rate of about 10 to 20 km/year.[102,103]

15.7.1 Ecological impacts

Relative to indigenous mussel species, *M. galloprovincialis* has a rapid growth rate, high fecundity, and enhanced tolerance to desiccation.[101] Its spread has therefore greatly increased the overall biomass and vertical extent of mussel beds along the entire west and, to a lesser extent, south coasts of South Africa. This has had three potential implications for the wider intertidal community: competition with other organisms for rock space, provision of habitat for other species, and an increase in the amount of food available to predators.

Competition for primary rock space. Before the arrival of *M. galloprovincialis*, the dominant space-occupying invertebrates in the mid to low intertidal of the Cape west coast were the slow-growing indigenous mussel *Aulacomya ater* and limpets, notably *Scutellastra* (formerly *Patella*) *granularis* and *S. argenvillei*. Much of the shoreline was open space, kept clear by the intense grazing activities of these and other large limpets. Because *M. galloprovincialis* grows much faster and extends higher into the intertidal than *A. ater*,[101,104] it is now the dominant space-occupying intertidal species at most sites. The net results have been a massive increase in both mussel cover and biomass, and a movement of both the upper limit and "center of gravity" of the mussel beds upshore. This has been accompanied by a decline in the overall biomass of *A. ater*, although paradoxically this species now occurs higher up the shore than before, because it can now find protection within the dense mats of *M. galloprovincialis*.

Besides outcompeting indigenous mussel species, *M. galloprovincialis* competes successfully with adult limpets for primary rock space.[99,100] However, the shells of large mussels also offer a favored settlement and recruitment site for juvenile limpets. Thus, as mussels encroach, adult limpets initially become spatially constrained and then eventually eliminated. However, at the same time, the densities of smaller limpets on the mussels increase enormously. The net result is an initial increase in limpet biomass (while the adults on the rock and the juveniles on the mussels are both present), followed by a decline when the adults are finally eliminated. Effects on limpet fecundity differ between species. In the midshore, small *S. granularis* living on mussel shells attain sexual maturity, and the overall mass of gametes released by the dense population of small individuals in areas of 100% mussel cover actually exceeds that of the few large animals previously occurring on the equivalent area of bare rock.[100] Although the larger, low-shore *S. argenvillei* also recruits onto mussel shells, it is unable to attain sexual maturity in this confined habitat. As a result, reproductive output cannot be maintained in areas invaded by mussels.[102]

Provision of habitat for mussel infauna. Mussel beds are structurally complex habitats that provide refuge to a diverse community of associated organisms. Griffiths et al.[100] showed that the infaunal communities colonizing *M. galloprovincialis* and *A. ater* beds were similar in both species richness and composition (69 and 68 species, respectively, with 70% shared). However, because *M. galloprovincialis* grow to a larger size, attain a greater biomass, and develop thicker, more structurally complex beds, they support a much denser invertebrate fauna than *A. ater* beds (76,600 versus 34,000 individuals per m^2). They also tend to provide refuge for larger infaunal individuals.

Enhanced availability of food for predators. Mussels form an important component in the diets of a wide range of predatory species, including aquatic forms such as fish, rock lobsters, starfish, predatory whelks, and octopuses, as well as terrestrial ones, including shorebirds and humans.[105] Since the introduction of *M. galloprovincialis* has led to a massive increase in mussel standing stock, this might be predicted to result in enhanced food availability to natural predators. The benefits to terrestrial species may be particularly marked, since they are now able to gain access to mussel stocks in the upper shore, where none had occurred previously.

This question has been addressed by Hockey and van Erkom Schurink[99] and by Griffiths et al.,[100] both of whom compared the contents of oystercatcher middens in Saldanha Bay before and after the *M. galloprovincialis* invasion. The figures given in these papers show a dramatic switch in oystercatcher diet away from limpets (31 to 19%) and *A. ater* (36 to 3%) in favor of *M. galloprovincialis*, which comprised more than 66% of the oystercatcher's diet in 1987–91. The hypothesis that the *M. galloprovincialis* invasion has increased the overall availability of food to oystercatchers is further supported by the fact that the proportion of pairs successfully raising two chicks, rather than the usual one, more than doubled over this same period, from 12 to 27%.[99]

15.7.2 Economic impacts

As indicated above, the *Mytilus galloprovincialis* invasion has had profound effects on both the physical appearance and the ecological processes occurring within rocky shores over much of the South African coastline. This has not, however, resulted in direct negative economic implications. The only costs (so far undetermined) may be additional fouling of ships' hulls, seawater intake pipes, and other marine structures. However, since *M. galloprovincialis* is essentially an intertidal species, these costs are not likely to be excessive. One area in which mechanical removal is being contemplated is within the West Coast National Park, where mussel beds are becoming established on mudflats favored as a feeding ground for migratory waders. On the positive side, *M. galloprovincialis* has formed the basis of a substantial mariculture industry, based almost entirely within Saldanha Bay. The production if this fishery in 1997 was 2145 tons, with a wholesale value of approximately \$3.5 million.[106] This benefit is likely to greatly outweigh the costs of removal of the species from areas in which it is currently problematic.

15.8 Conclusion

The selected case studies demonstrate the range of realized and potential impacts that invasive alien invertebrates may have on conservation, agriculture, and the welfare of humans and livestock. As shown by the Argentine ant, varroa mite, and snail examples, human activities such as improving access to wilderness sites, commercial dispersion of honeybee hives, and artificially connecting waterways can facilitate the distribution of invasive invertebrates. As illustrated by the red scale example, ignorance about life history and natural enemies when implementing control options can serve to exacerbate negative impacts.

New introductions and rapid spread of alien species already here will continue to affect environmental and economic interests in the country. *Liriomyza huidobrensis,* a leaf miner, was first observed on South African potato crops in 1999; yields in invaded areas have dropped from 40 tons/ha to 15 tons/ha.[107] The fish parasite *Bothriocephalus acheilognathi*, which poses a serious economic threat to the aquaculture industry, has spread into four major catchments within 20 years of its introduction.[91] Future introductions of *Aedes* spp. mosquitoes, nematodes, and mites are not unlikely.[108]

When comparing estimates of quantifiable economic losses from invasive alien invertebrates among several countries, South Africa does not rank among the highest.[7] Nonetheless, as a country that is attempting to stabilize its economy, it can ill afford additional assaults on its food production. From a conservation perspective, invasive alien invertebrates such as the Argentine ant and the varroa mite, to the Cape Floral Kingdom, should be viewed not only as a local threat to South Africa's biological heritage, but also on a global scale as a threat to the planet's rarest plant biome.

Not all species introductions, even those that become invasive, result in negative impacts. As shown with *M. galloprovincialis*, in some cases an accidental introduction can have a positive economic impact. Finding the policy balance that allows for potentially beneficial organisms and prevents introduction of those that will likely prove damaging, often in the absence of complete scientific data, provides a challenge for decision-makers in South Africa and elsewhere.

References

1. Huntley, B.J. South Africa's experience regarding alien species: impacts and controls, in I*nvasive Species and Biodiversity Management, Sandlund,* O.T, Schei, P.J., and Viken A. (Eds.), Kluwer Academic Publishers, Dordrecht, The Netherlands, 1999, 363.
2. Hoffmann, J. Introduction. Biological control of weeds in South Africa, *Agriculture, Ecosystems and Environment* 37, 1, 1991.
3. Braack, L.E.O. et al. Exotic arthropods in the Kruger National Park, South Africa: modes of entry and population status, *African Entomology* 3, 39, 1995.
4. De Moor, F.C. Factors influencing the establishment of aquatic insect invaders, *Transactions of the Royal Society of South Africa,* 48, 141, 1992.
5. Dennill, G.B. and Moran, V.C. On insect-plant associations in agriculture and the selection of agents for weed biocontrol, *Annals of Applied Biology* 114, 157, 1989.
6. Annecke, D.P. and Moran, V.C. *Insects and Mites of Cultivated Plants in South Africa*, Butterworths, Durban, 1982.
7. Pimentel, D. et al. Economic and environmental threat of alien plant, animal and microbe invasions, *Agriculture, Ecosystems and the Environment*, 84, 1, 2001.
8. Prins, A.J., Robertson, H.G., and Prins, A. Pest ants in urban and agricultural areas of southern Africa, in *Applied Myrmecology: A World Perspective*, Vander Meer R.K. and Dendeno, A. (Eds.), Westview Press, Oxford, 1990, 25.
9. Durr H.J.R. The Argentine ant, *Iridomyrmex humilis* Mayr, *Farming in South Africa,* 27, 381, 1952.

10. Skaife, S.H. The Argentine ant, *Transactions of the Royal Society of South Africa* 34, 355, 1955.
11. Fluker, S.S. and Beardsley, J.W. Sympatric associations of three ants: I*ridomyrmex humilis, Pheidole megacephala* and *Anoplolepis longipes* in Hawaii, *Annals of the Entomological Society of America*, 63, 1290, 1970.
12. Skaife, S.H. The distribution of the Argentine ant *Iridomyrmex humilis* Mayr, *Annals of the Cape Provincial Museum,* 2, 297, 1962.
13. Prins, A.J. Hymenoptera, in *Biogeography and Ecology of Southern Africa*, Werger, M.J.A. (Ed.), Junk, Den Haag, 1978, 823.
14. De Kock, A.E. and Giliomee, J.H. A survey of the Argentine ant, *Iridomyrmex humilis* (Mayr) (Hymenoptera: Formicidae) in South African fynbos, *Journal of the Entomological Society of Southern Africa,* 52, 157, 1989.
15. Addison, P. and Samways, M.J. A survey of ants (Hymenoptera: Formicidae) that forage in vineyards in the Western Cape Province, South Africa, *African Entomology*, 8, 251, 2000.
16. Jackson, M.E. Occurrence of the Argentine ant on the Witwatersrand, *South African Journal of Science*, 76, 31, 1980.
17. Erickson, J. M. The displacement of native ant species by the introduced Argentine ant *Iridomyrmex humilis* Mayr, *Psyche*, 257, 1971.
18. Ward, P.S. Distribution of the introduced Argentine ant (*Iridomyrmex humilis*) in natural habitats of the Lower Sacramento Valley and its effects on the indigenous ant fauna, *Hilgardia,* 55, 1, 1987.
19. Holway, D.A. Distribution of the Argentine ant (*Linepithema humile*) in Northern California, *Conservation Biology*, 9, 1634, 1995.
20. Cammell, M.E., Way, M.J., and Paiva, M.R. Diversity and structure of ant communities associated with oak, pine, eucalyptus and arable habitats in Portugal, *Insectes Sociaux,* 43, 37, 1996.
21. Human K.G., and Gordon, D.M. Effects of Argentine ants on invertebrate biodiversity in Northern California, *Conservation Biology*, 11, 1242, 1997.
22. Suarez, A.V., Bolger, D.T., and Case, T.J. Effects of fragmentation and invasion on native ant communities in coastal Southern California, *Ecology*, 79, 2041, 1998.
23. Donnelly, D. and Giliomee, J.H. Community structure of epigaeic ants (Hymenoptera: Formicidae) in fynbos vegetation in the Jonkershoek Valley, *Journal of the Entomological Society of Southern Africa*, 48, 247, 1985.
24. De Kock, A.E. et al. The influence of fire, vegetation age and Argentine ants (I*ridomyrmex humilis*) on ant communities in Swartboskloof, in *Fire in South African Mountain Fynbos: Ecosystem, Community and Species Response at Swartboskloof,* van Wilgen, B.W., Richardson, D.M., Kruger, F.J., and van Hensbergen, H.J. (Eds.), Springer-Verlag, Berlin. 1992, 203.
25. Holway, D.A. Competitive mechanisms underlying the displacement of native ants by the invasive Argentine ant, *Ecolog*y, 80, 238, 1999.
26. Holway, D.A. et al. The ecological causes and consequences of ant invasions, *Annual Review of Ecology and Systematics*, in press.
27. De Kock, A. Interactions between the introduced Argentine ant, Iridomyrmex humilis Mayr, and two indigenous fynbos ant species, *Journal of the Entomological Society of Southern Africa*, 53, 107, 1990.
28. Witt, A.B.R. and Giliomee, J.H. Soil-surface temperatures at which six species of ants (Hymenoptera: Formicidae) are active, *African Entomology*, 7, 161, 1999.
29. Bond, W. and Slingsby, P. Collapse of an ant-plant mutualism: the Argentine ant (*Iridomyrmex humilis*) and myrmecochorous Proteaceae, *Ecology*, 65, 1031, 1984.
30. Pierce, N.E. et al. The ecology and evolution of ant association in the Lycaenidae (Lepidoptera), *Annual Review of Entomology,* in press.
31. Clark, G.C. and Dickson, C.G.C. *Life Histories of the South African Lycaenid Butterflies*, Purnell & Sons, Cape Town, 1971.
32. Migdoll, I. *Field Guide to the Butterflies of Southern Africa*, Struik Publishes, Cape Town, 1994.
33. Heath, A. personal communication, 2001.

34. Visser D, Wright, M.G., and Giliomee, J.H. The effect of the Argentine ant, *Linepithema humile* (Mayr) (Hymenoptera: Formicidae), on flower-visiting insects of *Protea nitida* Mill, (Proteaceae), *African Entomology*, 4, 285, 1996.
35. Lach, L. unpublished data, 2001.
36. Cole, F.R. et al. Effects of the Argentine ant on arthropod fauna of Hawaiian high-elevation shrubland, *Ecology,* 73, 1313, 1992.
37. Bolger, D.T. et al. Arthropods in urban habitat fragments in southern California: area, age, and edge effects, *Ecological Applications,* 10, 1230, 2000.
38. Holway, D.A. Effect of Argentine ant invasions on ground-dwelling arthropods in northern California riparian woodlands, *Oecologia*, 116, 252, 1998.
39. Wong, T.T.Y. et al. Predation of the Mediterranean fruit fly (Diptera: Tephritidae) by the Argentine ant (Hymenoptera: Formicidae) in Hawaii, *Journal of Economic Entomology*, 77, 1454, 1984.
40. Dreistadt, S.H., Hagen, K.D., and Dahlstein, D.L. Predation by *Iridomyrmex humilis* (Hym: Formicidae) on eggs of *Chrysopa carnea* (Neu: Chyrsopidae) released for inundative control of *Illinoia lirodendri* (Hom: Aphididae) infesting *Lirodendron tulipifera*, *Entomophaga*, 31, 397, 1986.
41. Bernays, E.A. Host range in phytophagous insects: the potential role of geralist predators, *Evolutionary Ecology*, 3, 299, 1989.
42. Way, M.J., Cammell, M.E., and Paiva, M.R. Studies on egg predation by ants (Hymenoptera: Formicidae) especially on the eucalyptus borer *Phoracantha semipunctata* (Coleoptera: Cerambycidae) in Portugal, *Bulletin of Entomological Research*, 82, 425, 1992.
43. Heimpel, G.E., Rosenheim, J.A., and Mangel, M. Predation on adult *Aphytis* parasitoids in the field, *Oecologia*, 110, 346, 1997.
44. Samways, M.J. Ant assemblage structure and ecological management in citrus and subtropical fruit orchards in Southern Africa, in *Applied Myrmecology: A World Perspective*, Vander Meer R.K. and Dendeno, A. (Eds.), Westview Press, Oxford, 1990, 570.
45. Samways, M.J., Magda, N. and Prins, A.J. Ants (Hymenoptera: Formicidae) foraging in citrus trees and attending honeydew-producing Homoptera, *Phytophylactica*, 14, 155, 1982.
46. Potgieter, J.T. The Argentine ant, *Farming in South Africa,* 12, 160, 137.
47. Buys, B. Relationships between Argentine ants and honeybees in South Africa, in *Applied Myrmecology: A World Perspective*, Vander Meer R.K. and Dendeno, A. (Eds.), Westview Press, Oxford, 1990, 519.
48. De Jong, David. Insects: Hymenoptera (ants, wasps, and bees), in *Honey Bee Pests, Predators, and Diseases, 2nd edition*, Morse, R.A. and Nowogrodski, R. (Eds.), Cornell University Press, Ithaca, NY, 1990, 136.
49. Buys, B. Competition for nectar between Argentine ants (*Iridomyrmex humilis*) and honeybees (*Apis mellifera*) on black ironbark (*Eucalyptus sideroxylon*), *South African Journal of Zoology*, 22, 173, 1987.
50. Robertson, H. personal communication, 2000.
51. Anderson, D.L. Variation in the parasitic bee mite *Varroa jacobsoni* Oud, *Apidologie*, 31, 281, 2000.
52. Matheson, A. First documented findings of *Varroa jacobsoni* outside its presumed natural range, *Apiacta*, 30,1, 1995.
53. Page, R.E. Varroa mite impacts Africanized bee spread and beekeeping, *California Agriculture,* 52, 9, 1998.
54. Martin, S. Life and death of varroa, in *Varroa! Fight the Mite*, Munn, P. and Jones, R. (Eds.), International Bee Research Association, 1997, 3.
55. Ritter, W. and De Jong, D. Reproduction of *Varroa jacobsoni* Oud in Europe, the Middle East and tropical South America, *Zeitschrift Angew Entomologie*, 98, 55, 1984.
56. Boot, W.J. et al. Invasion of *Varroa jacobsoni* into drone brood cells of the honeybee *Apis mellifera*, *Apidologie*, 26, 109, 1995.
57. Allsopp, M.H., Govan, V. and Davison, S. Bee health report: varroa in South Africa, *Bee World*, 78, 171, 1997.

58. Allsopp, M.H. Varroa in South Africa — the early years, in *Africanized Honeybees and Bee Mites*, Flottum K. (Ed.), A.I. Root & Co, Medina, 2000.
59. Bowen-Walker, P.L., Martin, S.J., and Gunn, A. The transmission of deformed wing virus between honeybees (*Apis mellifera* L.) by the ectoparasitic mite *Varroa jacobsoni* Oud, *Journal of Invertebrate Pathology*, 60, 1, 1998.
60. Camazine, S. Differential reproduction of the mite *Varroa jacobsoni* (Mesostigmata: Varroidae) on Africanized and European honeybees (Hymenoptera: Apidae), *Annals of the Entomological Society of America*, 79, 801, 1986.
61. Ritter, W., et al. Development of tolerance to *Varroa jacobsoni* in bee colonies in Tunisia, *Proceedings of the International Symposium on Recent Research in Bee Pathology*, Ritter, W. (Ed.), Bucharest, Apimondia, 54, 1990.
62. Rosenkranz, P. and Engels, W. Infertility of *Varroa jacobsoni* females after invasion into *Apis mellifera* worker brood as a tolerance factor against varroatosis, *Apidologie*, 25, 402, 1994.
63. Moretto, G., et al. The effects of climate and bee race on *Varroa jacobsoni* Oud infestations in Brazil, *Apidologie*, 22, 197, 1991.
64. Richardson, D. personal communication, 1999.
65. Kraus, B. and Page, R.E. Jr. Population growth of *Varroa jacobsoni* Oud in Mediterranean climates in California, *Apidologie*, 26, 149, 1995.
66. Finley, J., Camazine, S., and Frazier, M. The epidemic of honey bee colony losses during the 1995-1997 season, *American Bee Journal*, 137, 446, 1997.
67. Loper, G. Over-winter losses of feral honey bee colonies in southern Arizona, 1992-1997, *American Bee Journal*, 137, 446, 1997.
68. Allsopp, M.H. Varroa in Africa — a serious threat, *XXXVII Apimondia Congress*, 28th October – 1st November, Durban, 2001.
69. Chandler, D. et al. Prospective biological control agents of *Varroa destructor* n. sp., an important pest of the European honeybee, *Apis mellifera*. *Biocontrol Science and Technology*, 11, 429, 2001.
70. Kfir, R. Natural control of the cereal stemborers *Busseola fusca* and *Chilo partellus partellus* in South Africa, *Insect Science and its Application*, 17, 61, 1997.
71. Kfir, R. Maize and grain sorghum: southern Africa, in *African Cereal Stem-borers: Economic Importance, Taxonomy and Natural Enemies and Control.* Polaszek, A. (Ed.), Wallingford, Oxon, 1998, 29.
72. Mohyuddin, A.I. and Greathead, D.J. An annotated list of the parasites of graminaceous stem-borers in East Africa, with a discussion of their potential in biological control, *Entomophage*, 15, 141, 1970.
73. Van Hamburg, H. The grain-sorghum stem-borer *Chilo partellus partellus* (Swinhoe) (Lepidoptera: Pyralidae): seasonal changes in adult populations in grain sorghum in the Transvaal, *Journal of the Entomological Society of Southern Africa*, 42, 1, 1979.
74. Van Rensburg, C.D. J. and Bate, R. 1986. Preliminary studies on the relative abundance and distribution of the stem-borers *Busseola fusca* and *Chilo partellus partellus,* Technical Communication, Department of Agriculture and Water Supply, Republic of South Africa 212: 49, 1986.
75. Skoroszewski, R.W. and Van Hamburg, H. The release of *Apantales flavipes* (Cameron) (Hymenoptera; Braconidae) against stalk-borers of maize and grain-sorghum in South Africa, *Journal of the Entomological Society of Southern Africa*, 50, 249, 1987.
76. Van Den Berg, J. and Van Rensburg, J.B.J. Infestation and injury levels of stem borers in relation to yield potential of grain sorghum, *South African Journal of Plant and Soil,* 8, 127, 1991.
77. Van Den Berg, J., Van Rensburg, J.B.J., and Pringle, K.L. Comparative injuriousness of *Busseola fusca* (Lepidoptera: Noctuidae) and *Chilo partellus partellus* (Lepidoptera: Pyralidae) on grain sorghum, *Bulletin of Entomological Research*, 81, 137, 1991.
78. Anonymous. RSA — Statistics in brief, Central Statistics, Pretoria, 1997.
79. Moran, V.C. The phytophagous insects and mites of cultivated plants in South Africa: patterns and pest status, *Journal of Applied Ecology,* 20, 439, 1983.
80. Bell, J. C. and McGeogh, M.A. An evaluation of the pest status and research conducted on phytophagous Lepidoptera on cultivated plants in South Africa, *African Entomology,* 4, 161, 1996.

81. Anonymous. South Africa. Pesticide use, *Agricultural Information Services Ltd.,* 9 Bovinton Rd, London, 1989.
82. Van Den Berg, J. and Nur, A.F. Chemical control, in *African Cereal Stem-Borers: Economic Importance, Taxonomy and Natural Enemies and Control*, Polaszek, A. (Ed.), Wallingford, Oxon, 319, 1998.
83. Kfir, R. 1990. Prospects for cultural control of the stalk borers, *Chilo partellus partellus* (Swinhoe) and *Busseola fusca* (Fuller), in summer grain crops in South Africa, *Journal of the Entomology Society of Southern Africa,* 53, 41, 1990.
84. Van Den Berg, J. and Rebe, M. Infestation pattern of stem-borers in maize surrounded by graminaceous trap crops, *Proceedings of the Thirteenth Entomological Congress of the Entomological Society of Southern Africa*, Pietermaritzburg, 2001.
85. Skaife, S.H. *African Insect Life*, Revised by Ledger, J. Struik Publishers, Cape Town, 1979.
86. Steyn, J.J. The effect of the cosmopolitan brown house ant (*Pheidole megacephala* F.) on citrus red scale (*Aonidiella aurantii* Mask.) at Letaba, *Journal of the Entomological Society of Southern Africa,* 17, 252, 1954.
87. Hattingh, V. The status of IPM on citrus in South Africa, *Proceedings of the 12th Entomological Conference of the Entomological Society of Southern Africa*, Potchefstroom, 1999.
88. Bedford, E.C.G. Citrus pests in the Republic of South Africa, *Citrus and Subtropical Fruit Journal,* 544, 5, 1979.
89. Giliomee, J. Management of red scale *Aonidiella aurantii* (Mask.) on citrus in South Africa, *Citrus and Subtropical Fruit Journal*, 566,6, 1981.
90. Appleton, C.C. and Brackenbury, T.D. Introduced freshwater gastropods in Africa with special reference to *Physa acuta*, *Proceedings of the Workshop on Medical Malacology in Africa*, Harare, Zimbabwe, 43, 1997.
91. Van Eede, J.A. and Brown, D.S. Colonization of fresh waters in the Republic of South Africa by *Lymnaea columella* Say (Mollusca Gastropoda), *Nature*, 210, 1172, 1966.
92. De Moor, I.J. and Bruton, M.N. Alien and translocated animals in southern Africa (excluding Zimbabwe and Mozambique)-revised checklist and analysis of distribution on a catchment basis, *Annals of the Cape Provincial Museums, Natural History,* 19, 305, 1996.
93. De Moor, I.J. and Bruton, M.N. Atlas of alien and translocated indigenous aquatic animals in southern Africa, *South African Scientific Programmes Report no. 144,* Council for Scientific and Industrial Research, Pretoria, 1988, 310.
94. Ashton, P.J., Appleton, C.C. and Jackson, P.B.N. Ecological impacts and economic consequences of alien invasive organisms in southern African aquatic ecosystems, in *The Ecology and Management of Biological Invasions in Southern Africa*, Macdonald, I.A.W., Kruger, F.J., and Ferrar, A.A. (Eds.), Oxford University Press, Cape Town, 1986, 247.
95. Brackenbury, T.D. and Appleton, C.C. Effects of controlled temperatures on gametogenesis in the gastropod *Physa acuta* (Physidae) and *Bulinus tropicus* (Planorbidae), *Journal of Molluscan Studies,* 57, 461, 1991.
96. Brackenbury, T.D. and Appleton, C.C. Recolonization of the Umsimdunsi River, Natal, South Africa, by the invasive gastropod, *Physa acuta* (Basommatophora, Physidae), *Journal of Medical and Applied Malacology,* 5, 39, 1993.
97. Anonymous. *African Snail News,* 3, 1997.
98. Grant, W.S., Cherry, M.I., and Lombard, A.T. A cryptic species of *Mytilus* (Mollusca: Bivalvia) on the west coast of South Africa, *South African Journal of Marine Science,* 2, 149, 1994.
99. Hockey, P.A.R. and van Erkom Schurink, C. The invasive biology of the mussel *Mytilus galloprovincialis* on the southern African coast, *Transactions of the Royal Society of South Africa,* 48, 124, 1992.
100. Griffiths, C.L., et al. Marine invasive aliens on South African shores: implications for community structure and trophic functioning, *South African Journal of Marine Science,* 12, 713, 1992.
101. Van Erkom Schurink, C., and Griffiths, C.L. Marine mussels in South Africa — their distribution patterns, standing stocks, exploitation and culture, *Journal of Shellfisheries Research,* 9, 75, 1990.
102. Steffani, N. personal communication.
103. McQuaid, C.D. personal communication.

104. Van Erkom Schurink, C., and Griffiths, C.L. Factors affecting the relative rates of growth in four South African mussel species, *Aquaculture,* 109, 257, 1993.
105. Griffiths, C.L. and Hockey, P.A.R. A model describing the interactive roles of predation, competition and tidal elevation in structuring mussel populations, *South African Journal of Marine Science,* 5, 547, 1987.
106. Stutterford, M. *Fishing Industry Handbook*, South Africa, Namibia and Mozambique, Exbury Publications, Cape Town, 2000.
107. Visser, D. A devastating new pest, *Liriomyza huidobrensis*, on potatoes in South Africa, *Proceedings of the Thirteenth Entomological Congress of the Entomological Society of Southern Africa,* Pietermaritzburg, 2001.
108. Cornel, A.J. and Hartwig, E.K. Importation of used tyre casings: a possible health hazard, *South African Journal of Science*, 85, 487, 1989.

chapter eighteen

World exotic diseases

Yoonji Kim

Contents

18.1 HIV/AIDS

18.1.1 Origins

Human immunodeficiency virus (HIV), which causes acquired immune deficiency syndrome (AIDS), appears to have evolved from the African continent. Both the HIV-1 and HIV-2 subtypes were introduced to humans through a cross-species infection from primate pathogens. Although the nature of the viral transmission remains unclear, a cross-species infection might have occurred when African primates were hunted for food. HIV-1, the

0-8493-0836-4/02/$0.00+$1.50

main subtype responsible for the AIDS pandemic, originated in and spread from chimpanzees in central Africa. One particular study strongly points to a particular African chimpanzee, *Pan troglodytes troglodytes*, as the natural host and reservoir for HIV-1.[1] The study suggested that other chimpanzee subspecies could have infected humans with HIV-1 as well. The origin for the HIV-2 subtype points to *Cercocebus atys*, the sooty mangaby monkey in West Africa.[2] According to research presented at the 13th International AIDS Conference in Durban, South Africa, HIV-1 was possibly transmitted to humans as early as 1675.[3] But it was not until the 1930s that HIV established itself as an epidemic strain. The molecular differences in HIV-1 and HIV-2 strongly suggest that AIDS has been in existence for some time, but its pandemic effects were triggered by practices such as promiscuous behavior and the sharing of drug-injecting equipment.

18.1.2 Surveillance and response

HIV/AIDS has become a dominant invasive disease, reaching pandemic proportions. In 2000, 5.3 million people worldwide were newly infected. Of that total, 2.2 million individuals were women, and 600,000 were children under 15 years of age.[4] By the end of 2000 there were a total of 36.1 million men, women, and children living with either HIV or AIDS. Women accounted for 16.4 million cases, while the number of children infected was about 1.4 million.

Since the beginning of the pandemic, a total of 21.8 million people have died from AIDS. In the year 2000, 2.5 million adults and 500,000 children died from the disease.[5] Every region has been severely affected by the HIV/AIDS epidemic. The number of adults and children living with HIV/AIDS and the number of AIDS deaths in Brazil, the United Kingdom, South Africa, the United States, India, Australia, and New Zealand can be seen in Table 18.1. However, the greatest impact of this infectious and invasive disease has been on the sub-Saharan region of Africa. Nearly 70% of the world's AIDS cases are reported from Africa, and it is now the continent's leading cause of death in humans.[6] In sub-Saharan Africa, 70% of the adults and 80% of the children are infected with HIV. Of the world's 21.8 million AIDS deaths, 75% have been from Africa.[4]

The WHO estimates that an annual $7 billion is needed to fight AIDS in low and middle-income countries.[7,8] Of the total $7 billion, African countries will need at least $1.5 billion to alleviate HIV infections.[9] Monetary funds needed to continue AIDS prevention and management in low- and middle-income nations are projected to rise from $3.2 billion in 2002 to $4.7 billion in 2003, $6.8 billion in 2004, and $9.2 billion in 2005.[10] By 2005, $4.8 billion will be needed for prevention programs, while $4.4 billion will go toward management and care. Out of the $9.2 billion, sub-Saharan Africa will receive about 50%, and of that half, 66% will be spent on care and management. Other regions' projections are $2.3 billion in South and Southeast Asia, $1.4 billion in Latin America and the Caribbean, $890 million in East Asia and the Pacific, $470 million in Central Asia and Europe, and $210 million in North Africa and the Middle East.

More money is needed for antiretroviral combination drug therapies to slow down the progression of the virus. The affordability of these drugs is a major concern. On a worldwide scale, antiretroviral therapies are taken by only 1% of the patients, and drugs for treating opportunistic infections are unequally distributed as well.[11] Until last year, the annual cost of drugs for an HIV/AIDS patient ranged from $10,000 to $15,000.[7] Drug costs have now gone down to $600 or less per person per year. However, middle- and low-income countries in Africa, Latin America, and Asia still need financial support to purchase and distribute the drugs.

Table 18.1 The Number of Adults (15 to 49 years of age) and Children (0 to 14years of age) with HIV/AIDS and Number of AIDS Deaths in United States, Brazil, United Kingdom, South Africa, India, Australia, and New Zealand at the End of 1999

	Population[a]	Adults(15–49yrs)[b]	Children(0–14yrs)[b]	AIDS Deaths in 1999[b]
South Africa	43 million	4.1 million	95 000	250, 000
India	1 billion	3.5 million	160,000	310,000
United States	281 million	840,000	10,000	20,000
Brazil	170 million	530,000	9900	18,000
United Kingdom	60 million	30,000	500	450
Australia	19 million	14,000	140	100
New Zealand	3.8 million	1200	<100	<100

[a] Population Reference Bureau, 2000

[b] Joint United Nations Programme on HIV/AIDS. Report on the global HIV/AIDS epidemic June 2000. Geneva, Switzerland.

18.1.2.1 South Africa

South Africa has the largest number of people with HIV/AIDS, and the highest HIV infection rates in the world.[5] At present, 20% of South African adults have HIV.[12] This could rise as high as 30% by 2005 but it is expected to range between 22 and 27% by 2010. It is projected that 6 million South Africans will contract HIV/AIDS by 2005.[13] Young women and men have the greatest risk of infection. Sixty percent of HIV-infected adults contracted the virus before the age of 25. Young women are most vulnerable to HIV/AIDS because of their gender. According to the South African Department of Health, women between 20 and 30 years of age have the highest exposure rates. In the past few years, the HIV infection rate for women under the age of 20 years sharply increased as compared to other age groups.[13] In South Africa, the low status of women makes them susceptible to sexual abuse, mistreatment, and economic dependency. Consequently, practices such as multiple sex partners, unprotected sex, lack of sex education programs, discrimination, illiteracy, migrant labor, and overall poverty continue to worsen South Africa's HIV/AIDS epidemic.[14] Within the next two decades, 6 million to 10 million individuals will die from AIDS.[14] More than 50% of the people who die from AIDS will succumb to it before they reach the age of 35. AIDS deaths in South Africa are expected to rise to 250,000 by 2002 and more than 1 million by 2008.[13] By 2005, there will be 1 million South African "AIDS orphans" whose parents or guardians have died from the disease.

South African businesses are also experiencing the impact of HIV/AIDS. The number of employee deaths related to AIDS could account for 40 to 50% of any given company's workforce.[14] This places a heavy burden on many companies because they must deal with the changes in work productivity, performance, lost skills, absenteeism, training, and replacement costs.[14] The HIV/AIDS epidemic has put enormous pressures on South Africa. In order to alleviate the grave situation, the South African government has organized various HIV/AIDS prevention programs and management strategies. Since 1994 more resources have gone into responding to HIV/AIDS than other diseases.[15] Costs surrounding the epidemic ranged from $21.5 million to $32 million in 1991 to $1.2 billion to $2.9 billion in 2000.[16] South Africa has started the National HIV/AIDS Programme, a 5-year plan geared toward safer sex, condom distribution, HIV testing, facility and treatment improvements, research and surveillance, human rights, education, and other support networks. Education and prevention are aimed at young people, because 45% of the South African population is under 20 years of age.[14] In a collaborative strategy, the government has also formed the South African AIDS Vaccine Initiative, a research and development program.[17]

18.1.2.2 India

The HIV/AIDS epidemic is a very diverse problem in India. While some states have low HIV infection rates, other states are plagued with growing HIV populations. Because of India's enormous and expansive population, the national average of HIV infection is only 0.4%.[18] This is close to the U.S. national HIV average of 0.3%. But India's low rate does not represent the devastating repercussions of HIV/AIDS in the country. Eighty-nine percent of all HIV/AIDS individuals are between the ages of 15 and 49.[6] India's National AIDS Control Organisation (NACO) found 81% of AIDS patients had contracted the virus through heterosexual contact, 5% through blood transfusions and blood products, and 5% from drug-injecting equipment. AIDS cases through perinatal transmission were at 0.72%, while all forms of infection were at 7.6%. Men with AIDS surpassed the number of women with the virus, at 78.6% and 21.4%, respectively.

In India, poverty, illiteracy, migrant labor, cultural norms, and socio-economic conditions contribute to the spread of AIDS. Unfortunately, in India, sex is a very lucrative business. With India's sex-industry profits soaring to as much as $8.7 billion, cities like Mumbai (Bombay) now have 20 times the number of prostitutes as in New York City.[18] In the mid-1990s, more than 25% of all sex workers in India were HIV-positive. By 1997, more than 70% of the prostitutes were living with AIDS.

The majority of sex clients are men. This is a growing health threat, because women and children are unknowingly being exposed to contracting HIV/AIDS from their husbands and boyfriends. Only 40% of Indian women are aware of AIDS, and even fewer are educated as to preventive measures.[19] A 3-year study in 1993–96 discovered that more than 10% of Indian women living in Pune (near Mumbai) were HIV-positive. Ninety percent of the infected women reported being sexually active with only their husbands.[18]

India's growing drug culture is also adding to HIV infections. In the state of Manipur, HIV infection among intravenous drug users and their partners increased from 0% in 1988 to 70% in 1992. Also in 1999, 2.2% of pregnant women in Manipur tested positive for HIV. Data are limited, but the Indian government calculates there are roughly 1 million heroin users in the country.

In 1992, India's Ministry of Health and Family Welfare launched NACO, which implements AIDS prevention, education, surveillance, and research. State governments have also introduced similar programs in order to reduce high-risk behaviors associated with sex and drug use. In the state of Tamil Nadu, HIV/AIDS programs resulted in greater condom use and a decrease in men having casual sex.[5] Currently, India spends 1% of its GDP on health care, but this is still inadequate.[18] With the country's per capita income being roughly $444 a month, antiretroviral drug therapies would cost more than the average person's annual income. In March 2001, India's Ministry of Health and the International AIDS Vaccine Initiative decided to concentrate on developing an AIDS vaccine.[20]

18.1.2.3 United States

In the United States, about 40,000 people become HIV-positive each year. AIDS incidence in the United States has dropped from 1997 through 1999.[21,22] The majority of HIV/AIDS cases are men who have sex with men (MSM) or are injecting drug users (IDU). Fifty-six percent of AIDS cases are MSM and 22% are IDU.[21,22] Seventy percent of all new HIV infections are men. Of the 70% of HIV cases, 60% were acquired through homosexual contact, 15% from heterosexual contact, and 25% from injecting drug use. Recent outbreaks of gonorrhea and syphilis among gay and bisexual men suggest that the AIDS risk may be rising as well. Injecting drug users, their partners and children represent at least 36% of new AIDS cases recorded in 1999.[21,22] Of the male IDU, 52% were heterosexual and 13%

homosexual. Women IDU account for 21% of the HIV cases, while 13% of infected women have heterosexual IDU partners.

The Center for Disease Control and Prevention (CDC) reports that adolescents, women, and minorities are increasingly at risk for HIV and AIDS. Young people under 25 years of age account for half of all new HIV infections. Women represent 30% of those HIV infections. In 1999, 40% of women with AIDS were exposed through heterosexual contact, while 27% were IDU, and 11% had sexual contact with an injecting drug user. HIV prevalence is high within African-American and Hispanic populations. Out of the total number of AIDS cases recorded in 1999, 42% of men and 63% of women were African-Americans.[21,22] In that same year, 19% of AIDS patients were Hispanics. Among HIV infections, 18% were Hispanic women and 20% were Hispanic men.

The United States spends about $20 billion on AIDS each year.[10] The Department of Health and Human Services invests around $8.5 billion on HIV/AIDS. Ten percent, or $820 million, is spent on prevention, while about 65%, or $5.5 billion, is allocated for treatment. The National Institutes of Health (NIH) spends $2 billion of its total HIV/AIDS budget on research. Resources invested in HIV prevention have reduced the rate of new HIV infections from 150,000 a year in the mid-1980s to 40,000 a year today. Lifetime treatments for 200,000 HIV-infected individuals would cost the nation $31 billion. National organizations, such as the National Center for HIV, STD and TB Prevention (NCHSTP) and the CDC, spend more than $800 million each year toward prevention, education, support, surveillance, and research.[21,22] Some of the CDC's goals for 2005 are to halve the number of the "highest risk" people exposed to HIV/AIDS, reducing the yearly number of individuals infected from 40,000 to 20,000 people, to increase the accessibility to HIV/AIDS treatment and care, and to strengthen HIV prevention programs.

18.1.2.4 Brazil

The Ministry of Health in Brazil reports that 74.7% of men, 25.3% of women, and 3.5% of children under 13 years of age are HIV-infected. The country's HIV/AIDS problem continues to stem from sexual transmission and injecting-drug use. The percentage of HIV infections through homosexual contact remains at 8.9%. HIV infections from heterosexual transmission rose from 25.3% to 43.5%.[23] HIV in women has become a major concern in Brazil. Between 1994 and 1998, AIDS in men grew at 8%, while the rate for women increased dramatically by 71%. With this upward trend, HIV/AIDS will be the foremost killer of Brazilian women between 15 and 49 years of age. Brazil's IDU-linked HIV infections have declined in recent years. The decline is the result of successful drug prevention programs and campaigns. Nearly 20.5% of new AIDS diagnoses are drug-related.[23] At its peak infections from IDU accounted for 25.5% of the new cases. Seventy-five percent of IDU-HIV individuals were between 12 and 19 years of age. In some Brazilian cities, the HIV prevalence among 12 to 19-year-olds was 50%. Among heterosexually infected individuals, 19% of men and 35% of women had an injecting drug partner.[24]

In 1994, Brazil implemented risk-reduction campaigns as part of its AIDS program. The most successful has been the needle exchange program, which has significantly minimized HIV cases acquired through contaminated needles and syringes. Last year, 1 million syringes were exchanged, and 30,000 drug users benefited from the program. Twenty-three percent of IDUs who participated in the program reported that they sought drug-treatment assistance and practiced safer sex.[23] The risk of HIV transmission through sexual contact has prompted health programs to encourage condom use. In 1986, only 5% of individuals used condoms during their first sexual intercourse. However, in 1999, condom use was at 50%. From 1993 to 1999, condom sales in Brazil rose from 70 million a year to 320 million a year.[5]

The country's social health insurance invests $300 million each year in HIV and AIDS activities.[10] The government's universal drug policy distributes free antiretroviral drug therapies to all AIDS patients. This has cut the number of AIDS deaths by 50% and has decreased opportunistic infections by at least 60%.[50] According to the Brazilian Ministry of Health, domestic drug therapies saved the country $442 million in future care and support.[25] Despite the legal and economic challenges made by pharmaceutical companies, Brazil's production of generic drugs can reduce drug costs by as much as 80%. Brazil spends $339 million annually on drug costs. This is expected to rise to $462 million due to the increasing number of patients and the growing need for drug therapies.[5]

18.1.2.5 Australia and the United Kingdom

Sexual transmission through unprotected homosexual contact is the prevalent mode of transmission for HIV/AIDS infections in both the United Kingdom and Australia. In the United Kingdom more than 60% of diagnosed HIV patients and 70% of AIDS cases are among homosexual or bisexual men.[26] The rate of infection is increasing especially among young men. In 1999, 61% of HIV diagnoses were homosexual and bisexual men.[27] Similarly in Australia, 90% of the AIDS patients are MSM.[28] Australia's HIV/AIDS epidemic is mainly isolated to men who practice unsafe sexual intercourse with other men.[16] In 1995, homosexual transmission accounted for more than 80% of diagnosed HIV cases.

Heterosexual contact is a growing concern in the United Kingdom. Twenty percent of HIV cases and 16% of AIDS cases were heterosexually transmitted.[26] In 1997, heterosexually transmitted HIV infections represented about 25% of all new cases. In 1999 and 2000, the United Kingdom's Public Health Laboratory Service recorded rising numbers in HIV transmission between men and women.[29] Annual cases grew from 100 in 1986 to almost 800 by the mid-1990s.[26] Fifty-eight percent of heterosexual women are HIV-infected.[122] A large number of HIV infections have been attributable to people who live, work, or spend time abroad, particularly in sub-Saharan Africa.[29] More women than men are infected while abroad or from a partner who was infected abroad.

In Australia, HIV transmission through heterosexual contact is still low in comparison to homosexual transmission. However, within the past two decades the percentage of heterosexual HIV infections has grown. Ten cases were recorded in 1985, 30 in 1988, and 52 in 1994.[28] Australia's indigenous populations of Aborigines and Torres Strait Islanders are experiencing the HIV epidemic as well, and the rate of infection is higher within these two communities than in other groups. Between 1992 and 1994, non-indigenous populations in Australia represented 17% of total HIV cases, while 50% were Aboriginal and Torres Strait Islander populations. Within the Aboriginal and Torres Strait Islander groups, 55% of the infections were by homosexual contact and 24% were from heterosexual contact.[28]

Australia has a low rate of HIV infection within the IDU community. In 1994, IDU-associated AIDS cases made up 2.5% of cumulative infections.[30] In contrast to Australia, the United Kingdom has experienced alarming rates of HIV among IDU. Reports estimate that in 1996, 18% of injecting drug individuals in London and southeast England shared syringes and needles. In 1999, 32% of IDUs shared needles and syringes.[27] From 1998 to 1999, the sharing of needles and syringes went up from 35% to 41%.[29]

The United Kingdom has devised strategies to mitigate the HIV/AIDS epidemic. Needle exchange programs have been adopted in most urban areas since the late 1980s.[31] In July 2001, the Department of Health introduced its first national sexual health strategy, intended to promote good sexual health and to prevent sexual diseases.

Australia's National HIV/AIDS Strategy is an integrated system that includes the Commonwealth of Australia and its states and territories.[28] AIDS has been a top priority since the start of the epidemic.[30] One of Australia's goals for 1999 was to reduce high HIV infection rates in young homosexual men and to prevent further HIV spread in Aboriginal and Torres Strait Islander communities. In 1987, Australia adopted a needle exchange program. In 1994, when IDU-associated AIDS cases stood at 2.5%, it was calculated that 10 million syringes and needles were exchanged at more than 4000 outlets in Australia.[30]

18.1.2.6 New Zealand

New Zealand has one of the lowest AIDS rates in the world. In recent years, AIDS diagnoses have dropped from 78 cases in 1991 to 44 cases in 1994.[32] HIV and AIDS in New Zealand primarily affect sexually active individuals. Infections have mainly been reported in men with homosexual contact, but the number of individuals infected through heterosexual contact has been increasing.[33] New Zealand has adopted various programs and endorsed many activities such as a needle and syringe exchange program, national media campaigns, HIV testing, and blood-supply screenings.[32]

18.2 Tuberculosis

18.2.1 Origins

The origin of tuberculosis (TB) remains uncertain. Humans have been afflicted with the disease since antiquity. There are references to TB in ancient writings from Babylon, Egypt, China, India, and Greece.[33,34] Three-thousand-year-old mummies have been found to have traces of *Mycobacterium tuberculosis*.[36] The earliest date places TB back to the Neolithic period.[37] One study calculates that *M. tuberculosis* evolved 20,400 to 15,300 years ago.[38] Evidence found in Europe, Africa, and the Americas indicates tuberculosis was even prevalent in prehistoric populations.[39] Traces of tuberculosis have been recovered in human remains from Italy and Germany, while in the Americas, anthropologists excavated an AD 700 Peruvian mummy with tubercular bacilli.[36,87] In 1994, *M. tuberculosis* was discovered in another Peruvian mummy.[40]

Despite the literary references in China and India, substantial early evidence of TB is yet to be found. Most experts believe migrating populations arrived in the Americas around 20,000 to 25,000 years ago.* However, others either estimate populations immigrated as early as 30,000 to 35,000 years ago or as recently as 11,000 to 12,000 years ago.[41–43] Tuberculosis may have been endemic in some of the migrating groups, but the disease did not become epidemic in the Americas until the arrival of the Europeans.[39]

18.2.2 Surveillance and response

There are three specific strains within the *Mycobacterium* genus: *Mycobacterium bovis*, *Mycobacterium avium*, and *Mycobacterium tuberculosis*. *Mycobacterium tuberculosis* is the strain that infects humans. More than a third of the world's population is infected by the TB bacillus.[44] Within their lifetimes, 5 to 10% of these individuals will become infectious. Nearly 2 billion people are infected with *M. tuberculosis*, and 1% of the world population is infected each year. One person is infected each second, and 8 million each year. Tuberculosis kills one person every 10 seconds, 5000 people a day, and 2 million individuals each year. TB is the leading killer of women, with 605,000 deaths annually.[45] The World Health Organization predicts that from 2000 to 2020, an additional 1 billion people will be infected, 200 million will develop symptoms, and 35 million will die. There are 1.5 million new TB cases each year in sub-Saharan Africa, approximately 3 million cases in Southeast Asia, and over a quarter of a million cases in Eastern Europe.

* Populations that migrated to the Americans were probably from Central Asia.

The HIV/AIDS epidemic and the emergence of multi-drug-resistant TB (MDR-TB) strains have contributed to the rising number of TB infections. Inconsistent treatment with TB drugs has led to the multi-drug-resistant strains. More than 50 million people are now infected with MDR-TB.[6] Tuberculosis is also the foremost killer among people with HIV/AIDS. Approximately 15 million people are co-infected with HIV and TB; two thirds of these cases are from sub-Saharan Africa.[46] Twenty percent of the individuals who contract AIDS will die from tuberculosis.[6] From 1997 to 1999, the number of worldwide TB cases rose from 8.0 million to 8.4 million each year.[47]

In 1993, the WHO declared TB a global health threat. As a result, the Directly Observed Treatments, Short Course (DOTS) program was formed. The strategy behind the program was to facilitate cooperation between governments, public health officials, scientists, and national TB control programs. DOTS is based on surveillance systems, laboratory-based notifications, standard TB treatments, and stable drug supplies.[47] The percentage of the world's population with access to DOTS increased from 43% in 1998 to 45% in 1999.[44] Under DOTS, the average treatment success rate is 78% of confirmed cases and 82% in countries with higher TB incidence rates. Roughly 900,000 people are treated under DOTS services.[48]

Tuberculosis is a preventable disease. A six- to eight-course treatment costs as little as $10 to $15.[49] The WHO's goals for 2005 are to detect 70% of all new TB infections and to successfully treat 85% of all notified cases. In order to increase detection and treatment, a total of $1.5 billion more will be needed just to aid the 22 countries most affected by TB.[50] An annual $400 million will ensure that at least 70% of the 8 million newly infected people will be given quality care and treatment.

18.2.2.1 India

India accounts for one third of the world's TB cases and two thirds of the TB infections in Southeast Asia.[51] Out of the 20,000 people infected with the TB bacteria every day, 5000 will develop tuberculosis, and at least 1000 will die from the disease.[45] In one year, 2 million people will be infected and 450,000 will die from TB.[46] Compared to deaths from other tropical diseases, TB kills 14 times more people, it causes 21 times more deaths than malaria and 400 times more deaths than leprosy.[45] The Indian population between 15 and 54 years of age accounts for 85% of new TB infections.[51] One in five 15-year-olds are already infected.

Under the Revised National TB Control Programme (RNTCP), India's DOTS program is the second largest in the world. More than 25,000 people are placed into the program every month.[44] By the end of 2000, the DOTS program was accessible to at least 150 million Indians. This number is predicted to rise to 300 million patients by 2002. In 2001, the program was accessible to at least one third of the country. More than 500,000 people are currently in the program, preventing 100,000 deaths and 1 million infections so far.[45] TB costs India $3 billion every year.[51] The country spends $300 million a year on diagnosis and treatment alone. India's DOTS program is financed mainly through a $142 million low-interest loan from the World Bank.[46] India's primary goal for 2002 is to reach half the country, economically saving several hundred million dollars in future care and support.[45] HIV/AIDS and MDR-TB are the major priorities.

18.2.2.2 South Africa

South Africa has one of the highest TB incidences in the world with 341 infections per 100,000 people.[15] In one rural district, TB incidence rose from 154/100,000 in 1991 to 413/100,000 infections in 1995. In 1997, the Medical Research Council (MRC) of South Africa recorded 180,507 new infections, where 32.8% or 73,670 cases were HIV co-infections.[52] South Africa's National Tuberculosis Research Programme predicts that there will be 3.5 million new cases within the next 5 years. Without improvement, there will also be 1.7 million deaths over the next 10 years.[53] The HIV/AIDS epidemic in South Africa has only worsened the situation. In 1997, TB and HIV co-infections represented 32% of all reported TB cases.[52] In the South African provinces of KwaZulu–Natal, Mpumalanga, and Gauteng, HIV/TB incidence is a major issue. South Africa's TB epidemic is also due to the MDR-TB strains. Poor health services and inadequate treatment have worsened this problem. In 1996, more than 2000 confirmed cases were MDR-TB.[15]

TB control is a national health priority in South Africa. The government's aim is to provide accessible and adequate health coverage, especially for the poor.[51] Adopted in 1996, the National TB Control Programme improved TB surveillance and treatment. Although the cure rates were at 54% in 1997, South Africa continues to enhance its system. Successful treatment rates are stable at 74%, but 12% of all cases are still not examined.[44] An annual $100 million is spent on TB in South Africa, and an extra $3 billion will be needed to fight the disease over the next 10 years.[52] The simultaneous fight against HIV/AIDS can possibly prevent 1.7 million TB infections and save $400 million. The emergence of MDR-TB and HIV/TB co-infections has prompted South Africa to provide more support and resources for treatment and research.

18.2.2.3 United Kingdom

Until recently tuberculosis has been of a low priority in the United Kingdom. In the 1980s TB incidence dropped with the lowest number of cases (5085) being reported in 1987.[54] However, from 1982 to 1993, the re-emergence of TB cases has increased the annual number of reported cases in England and Wales within the 5000 to 6000 range.[55] Between 1999 and 2000, the Public Health Laboratory Service recorded 10.6% more TB notifications, from 6144 to 6797 cases.[56] From 1993 to 2000, the TB incidence in England and Wales jumped from 11.7/100,000 to 12.9/100,000.[57] The groups most affected by TB are 25- to 64-year-old men and foreign-born individuals. Men accounted for 75% of the increase, and a national survey conducted in 1998 found that 56% of the TB cases in England and Wales were among ethnic populations, primarily from the Indian subcontinent and Africa.[58] HIV/TB co-infection cases contributed to 3% of the cases reported in 1998. MDR-TB incidence has remained at low levels, but it is still under close surveillance. There were 385 TB-related deaths in 1997 and 392 deaths in 1998.[57]

More resources have been invested in the United Kingdom's TB control. The disease is now a concern in England and Wales. As many as 27% of the clinically proven TB cases are unrecorded.[59] Currently, a national vaccination program provides immunization for 75% of the population.[60] Effective treatment for notified TB cases, the screening of high-risk individuals, and better surveillance and research are the aims for improving public health in England and Wales.[57] More research will go into HIV/TB co-infections and MDR-TB as well.

18.2.2.4 Brazil

Out of the 22 countries most burdened with TB, Brazil is ranked tenth. According to the Brazilian Ministry of Health, 35% of the population carries the TB germ, and 10% will develop the disease.[61] TB incidence in Brazil is 50/100,000, with an estimated 124,000 new infections each year.[51,62,63] TB attributable to HIV totals about 6000 cases per year.

TB causes 19,000 deaths annually in Brazil.[51] The Pan American Health Organization (PAHO) calculated that from 1979 through 1995, 100,000 died from TB in Brazil. Brazil's indigenous Yanomami Indians are also affected. Introduced during Brazil's gold rush in the 1980s, TB is now at an epidemic level among the Yanomami group, with a prevalence of 6.4%.[36] Out of the 625 Yanomami Indians tested, 40 had active TB. The Brazilian government and various organizations are providing anti-TB drugs and maintaining close surveillance.

Brazil spends $23 million on TB control every year. Through DOTS, Brazil hopes to increase the diagnosis rate to 92% and the successful treatment rate to 85%, while cutting TB incidence by 50% and mortality by 66%.[51] The DOTS strategy is expanding to cover HIV/TB and MDR-TB surveillance and research as well. The DOTS program is available to 7% of the country. Despite a current successful treatment rate of 91%, only 82 patients were registered.[44] To help meet the TB goals, Brazil has created programs with bonuses for facilities to treat and cure TB infections.[51] Already, 23 of Brazil's 27 states have adopted the bonus program. Bonuses ranged from $57 to $85.

18.2.2.5 Australia

Australia's incidence is around five to six cases per 100,000 people each year.[64] Australia has one of the lowest TB rates in the world. HIV/TB cases and MDR-TB are low as well.[65] Most TB notifications are from New South Wales and Victoria, but incidence is highest in the Northern Territory.[66] TB is a major health threat to the elderly, immigrants, and native tribes such as the Maoris and Aborigines.[64] Incidence of TB in Aborigines is 10/100,000 to 13/100,000.[66] The TB rate for all foreign-born populations is 15/100,000. Australia's main goal for 2010 is to eradicate TB to where incidence is less than 1 case per 100,000 people.[67] Australia is also helping to combat high TB infections in neighboring countries such as Papua New Guinea, the Philippines, Indonesia, and East Timor.[51]

18.2.2.6 New Zealand

New Zealand's TB incidence is on the rise. Notifications reached their highest number in two decades with 452 cases in 1999. This was a 23% jump from the 368 cases recorded in 1998.[68,69] The national incidence for 1999 was 12.4/100,000 compared to 10.1/100,000 from the previous year. Areas in central, southern, and northern Auckland recorded higher than normal TB incidence, resulting in eight deaths in 1998 and 10 in 1999.[68,69] Total deaths in New Zealand due to TB rose from 356 in 1989 to 482 in 1999.[70] As with Australia, TB is a concern with immigrants. New Zealand's TB control is dependent on prompt reporting and diagnosis, complete treatment, and thorough management of all cases.[68,69]

18.3 Malaria

18.3.1 Origins

Malaria is a parasitic disease caused by *Plasmodium protozoa*. The four *Plasmodium* species parasitic to humans are *P. falciparum*, *P. malariae*, *P. ovale*, and *P. vivax*. The *P. falciparum* strain is most deadly to humans and accounts for the majority of malaria cases. The *Plasmodium* strains appear to have diverged about 129 million years ago, predating the emergence of hominids.[71,72] *P. falciparum* is genetically closer to *P. reichenowi* (parasite affecting chimpanzees) than to other human and non-human *Plasmodium* parasites.[72] The two strains mostly likely separated 6 million to 10 million years ago, about the same time human and chimpanzee lineages diverged.

Southeast Asia has always been considered malaria's place of origin.[73] However, recent molecular studies point to an African origin as well. Evidence strongly supports the likelihood that *Plasmodium* malaria evolved in Africa and then branched out to areas in Southeast Asia and South America.[74] The molecular evidence examined shows a genetic common ground with the *Plasmodium* genus from Africa.

18.3.2 Surveillance and response

Climate changes such as El Niño, population movements caused by wars, drug-resistant strains, lack of sanitation, and environmental destruction are having pandemic effects on the re-emergence of malaria. *Anopheline* mosquitoes are the primary vector. Malaria is an international health problem that affects 40% of the world's population in more than 90 countries.[75] With 10 new cases being recorded every second, 300 million to 500 million new malaria cases develop each year.[6] This is five times the number of cases of AIDS, TB, measles, and leprosy combined.[76]

Africa is greatly burdened with malaria. Nine out of 10 infections in Africa are from the sub-Saharan region. Malaria causes 1.5 to 3.0 million deaths each year.[6] The disease kills one child every 30 seconds, 3000 children per day, and more than 700,000 every year.[6,76] It is responsible for one quarter of all childhood deaths in Africa. Women are also at risk. Pregnancy can quadruple the risk of infection and double a woman's risk of death.

Imported cases of malaria are on the rise. "Airport malaria" and "weekend malaria" are spread from international travelers. In 2000, more than 12,000 people were infected among European travelers.[6] Drug-resistant malaria strains are developing as well. Anti-malarial drugs are losing their worldwide effectiveness. For instance in Southeast Asia, four leading anti-malarial drugs have become useless against some malarial strains.[76]

Malaria is a preventable disease. With proper treatment and control, worldwide malaria can be halved by 2010.[76] Organizations such as Roll Back Malaria (RBM), Multilateral Initiative on Malaria (MIM), WHO's Global Malaria Control Strategy, the Task Force, and the Medicines for Malaria Venture (MMV) are working toward better malaria prevention, research, and control. Although no vaccines are available yet, there are more than a dozen in development and testing. Research has also gone into safer vector control treatments and strategies. Since the ban of widespread DDT use, there have been efforts to find alternatives. RBM has helped eight Latin American countries to raise $750,000 to look for DDT substitutes and better vector management strategies.[77]

The economic costs from malaria are high. In endemic countries, malaria can slow economic growth by as much as 1.3% per year.[78] It was calculated that if malaria had been eradicated 35 years ago, Africa's current GDP would be 32% higher today.[76] In agricultural communities, malaria-stricken families can only harvest 40% of their crops, drastically cutting their income. According to 1997 estimates, malaria in sub-Saharan Africa was responsible for $2 billion in direct and indirect costs.[75] The average cost for each African nation is at least $300,000 a year; this is about 60 cents per person in a country of 5 million people. In Thailand, patients spend nine times the average daily wage on malaria treatment.[76] Families can spend more than 25% of their income on care and support alone. In order to achieve the 2010 goal, $1 billion will be needed every year.[78]

18.3.2.1 India

In India, an estimated 2 million people are infected with malaria each year.[79] *P. vivax* is the predominant strain, followed by *P. falciparum*. Between 1.2 million and 1.5 million, or 60% of the new infections are caused by *P. vivax*.[80] Thirty-three percent of India's population lives in poverty and thus the greatest probability of infection. Malaria had been confined to rural areas until urbanization. At the time when India initiated its malaria eradication program in the 1950s, 800,000 people were dying each year from the disease, while 75 million more were infected. In the 1970s, infections rose again to an estimated 6.5 million per year. From 1995 to 1997 there were four large epidemics, with more than 2.5 million cases reported in 1997.[81]

Within the past two decades, India has spent a quarter of its health budget on malaria prevention and management. However, the country's National Malaria Eradication Programme will need more financial support due to drug-resistant strains and the spread of mosquitoes in urban areas, contributing to the overall rise in malaria incidence. *P. falciparum* is becoming more widespread than *P. vivax*, as well as more resistant to anti-malaria drugs. India's malaria budget was $40 million; this was up 60% from the previous year.[81] The National Malaria Eradication Programme has proposed a 5-year plan targeting *P. falciparum* malaria. The program will cost $200 million, which India will borrow from the World Bank. The plan's initiative is to focus on the 100 high-risk districts that contribute 80% of the cases, and the 210 million people residing in them.

18.3.2.2 Brazil

More than half of all malaria cases in Latin America are from Brazil. In 1999, Brazil reported 609,594 cases, while the sum of 20 other Latin American countries was 596,580.[82] About 99% of all cases in Brazil are from the Amazon Basin; 58% are caused by *P. vivax*, 41% by *P. falciparum*, and 1.0% by *P. malariae*.[83] The high incidence in the Amazon Basin is attributable to environmental degradation from heavy mining activities, initiated by the 1980s gold rush. As a result, malaria prevalence is high in mining communities, as well as among the Basin's indigenous population of Yanomami Indians.[84] Both of these migrating populations have contributed to the emergence of drug-resistant malaria.

From 1988 to 1996, malaria control in Brazil cost $616 million, of which $526 million was used on prevention and $90 million on treatment.[85] The immediate spending on malaria control in Brazil is believed to have prevented almost 2 million cases and more than 200,000 deaths. In terms of future spending, the country probably saved $42.7 million in treatment and care.[86] With the rise of drug-resistant strains, Brazil will continue to focus its programs on preventive treatment, particularly targeting high-risk groups in the Amazon Basin.

18.3.2.3 South Africa

Malaria in South Africa has been increasing since 1995. Large proportions of cases are from migrating populations from neighboring countries, including international travelers and those susceptible to changes in the climate.[87,88] *P. falciparum* is the dominant in South Africa. According to the South African Health Department, there were 27,035 cases and 163 deaths in 1996. The figures dramatically jumped to 51,535 cases and 402 deaths in 1999 and 61,523 cases and 420 deaths in 2000. The Northern Province, Mpumalanga, and KwaZulu–Natal lead in malaria incidence, but KwaZulu–Natal has the most malaria notifications.

Malaria has cost Africa $300 billion.[89] It is estimated that Africa's GDP would be $100 billion greater if malaria were eradicated. Malaria control in Africa could save between $126 million and $3 billion each year. South Africa has one of the best malaria programs in Africa. The country uses new surveillance techniques such as geographical mapping and modeling to pinpoint outbreaks.[90] Roll Back Malaria has granted funding to the Medical Research Council of South Africa in order to create malaria maps of Africa. Its purpose is to better utilize resources on a larger scale.

18.3.2.4 Australia, New Zealand, and the United Kingdom

Australia has been free of endemic malaria since 1981.[91] Malaria still exists, but it is not transmitted in the country.[92] All reported cases are imported malaria from countries such as Papua New Guinea, the Solomon Islands, and Vanuatu.[93] In the last decade there were 7381 cases and seven deaths from malaria in Australia. It is estimated that each year 1.45 million individuals come to Australia from regions with high-risk malaria. Of the 705 malaria notifications in 1998, *P. vivax* represented 46% of the cases, while *P. falciparum* represented 25%.[94] Australia's proximity to neighboring countries inflicted with malaria and the nation's suitable climate still make the country vulnerable to a re-emergence of the disease.[95]

Cases in New Zealand were also acquired overseas. Seventy-three cases were reported in 1998.[96] In the United Kingdom, there are more than 2000 imported malaria cases and nine deaths each year.[97] All three countries have strongly advised their travelers to exercise caution before, during, and after overseas travel.

18.4 Cholera

18.4.1 Origins

Cholera is believed to have originated from the Indian subcontinent. It has been endemic in Bangladesh and in the deltas of the Ganges and Brahmaputra Rivers since antiquity.[98] Ancient historical texts from India, China, and the Middle East support the existence of cholera long ago.[99] *Vibrio cholerae* thrives in ocean waters and coastal estuaries.[160] This has enabled cholera to spread rapidly around the world. Beginning in the 19th century, there have been seven cholera pandemics. The first (1817–23), second (1829–51), third (1852–59), fourth (1863–79), fifth (1881–96), and sixth (1899–1923) caused by the *V. cholerae* 01 strain originated from Bangladesh.[101] The seventh pandemic, which began in 1961 and continues to spread today, started in Sulawesi, Indonesia.[102,103] Caused by *V. cholerae* strain 01 biotype El Tor, the ongoing outbreak struck the rest of Asia by 1964, invaded Africa by 1970, and hit the Americas by the early 1990s. Before the invasion, both Africa and Latin America had been cholera-free for more than a century. In 1992, a new cholera strain, *V. cholerae* 0139, emerged from the Bay of Bengal.[101] By 1993, *V. cholerae* 0139 Bengal was reported from India, Bangladesh, Nepal, Burma, Thailand, Malaysia, Saudi Arabia, China, and Pakistan.[104] The new epidemic strain is still contained in Asia.

18.4.2 Surveillance and response

International trade and travel expands cholera's epidemic threat. In all cases, unsanitary conditions, poor hygiene, overcrowding, and climate conditions have only worsened the cholera outbreaks (Table 18.2). Cholera primarily affects low- to middle-income countries whose climate is suitable for endemic malaria. In 2000, there were 137,071 cholera cases and 4908 deaths reported in 56 countries.[105] Eighty-seven percent (118,932) of those cases were in Africa, particularly from the countries of Comoros, Mozambique, Somalia, South Africa, and the Democratic Republic of the Congo. Cholera is a major problem in the sub-Saharan region; in 1999, 52% of notified cases occurred in southern Africa.[106]

Cholera in Asia declined in 2000. In 1998 the number of cases were 24,212 and 172 deaths; in 1999, the figures increased to 39,417 and 344, but by 2000 only 11,246 cases were notified, with 232 deaths.[105,106] The incidence drop is due to a decline in notified cases from Afghanistan, Iraq, and Iran. However, Afghanistan, India, and China still report the highest number of infections. In the Americas, cholera incidence dropped from 8126 cases and 103 deaths in 1999 to 3101 cases and 40 deaths in 2000. Brazil, Ecuador, Guatemala, and Nicaragua showed the largest decline in cholera incidence.

In all countries stricken with cholera, the main course of action is a focus on sanitation and hygiene. In 1991, the WHO Global Task Force on Cholera Control was created to help implement the program. The program emphasized the importance of clean water, sewage control, and personal hygiene.[103]

Table 18.2 Annual Number of Reported Cholera Cases and Deaths, 1998–2000

	1998[a]		1999[b]		2000[c]	
	Cases	Deaths	Cases	Deaths	Cases	Deaths
Brazil	2571	27	3233	83	715	17
United Kingdom	18	0	—	—	33	0
South Africa	20	1	68	2	19667	68
India	7151	10	3839	6	3807	18
Australia	5	0	4	0	1	0
New Zealand	1	0	1	0	—	—
World Total	**293,121**	**10,586**	**254,310**	**9,175**	**137,071**	**4,908**

[a] World Health Organization. WER 1999 74 (257-264)

[b] World Health Organization. WER 2000 75 (249-256)

[c] World Health Organization. WER 2001 76 (233-240)

Three cholera vaccines are currently available, but they are too expensive for low- to middle-income countries. Cholera vaccines range from \$0.51 per dose to \$3.00 per dose.[103] Obviously, if a vaccination program were included along with current control strategies, such as oral rehydration treatments and antibiotics, more infections and deaths would be prevented. In a case study using a two-dose oral cholera vaccine in a population of 43,000 individuals, it was estimated that a vaccination program would cost from \$23,600 to \$187,000, depending on vaccine costs per dose. The same study revealed that vaccine effectiveness was as high as 95% after the first dosage.

18.4.2.1 South Africa

South Africa's cholera epidemic reached its highest figures in the past two decades, with an estimated 20,000 infections and 73 deaths.[107] A large percentage of the cases were in KwaZulu–Natal Province, which recorded almost 500 new infections per day.

The lack of clean running water and sanitation is responsible for the cholera jump in South Africa. About 80% of the population do not have access to clean water or functioning toilets.[108] In rural areas, at least 8 million people live without clean water and depend on untreated rivers and streams for water.[107] In KwaZulu–Natal alone, more than 1 million people do not have adequate sanitation, while for South Africa as a whole, the number rises to 18 million.[108] The country's cholera management has been impeded by shortages in health workers and medical facilities and a lack of community awareness. However, funds allocated by the South African government and the European Union will go toward water and sanitation management. For instance, the European Union has provided \$66.6 million to build toilets.[108]

18.4.2.2 India

Cholera outbreaks emerge seasonally in India. Outbreaks are highest in May and June in the northern region, while the southern region experiences its peak in December and January.[98] Because cholera was not considered a notifiable disease in previous years, information on India's cholera patterns and past surveillance is limited. India also faces problems with the underreporting of cases and deaths.[109] This obstacle prevents the government from accurately assessing the cholera situation and implementing effective control and treatment. The emergence of drug-resistant strains and the *V. cholerae* 0139 Bengal strain have added to the obstacles faced by India. Despite the present conditions, oral rehydration treatments and antibiotics have improved India's public health. In addition to better surveillance and sanitation, greater community awareness, better cholera education, and various other health programs have been instituted.[109]

18.4.2.3 Latin America

When cholera arrived in Latin America in 1991, the number of cases and deaths exploded. Brazil's cholera epidemic infected 150,000 people and killed 1700 individuals.[62] Beginning in 1991, cholera cases in Brazil jumped from 2103 in 1991 to 60,340 in 1993.[110] Although the cholera incidence dropped to about 1000 in 1995, infections are on the rise again. More than 2000 and 3000 cases were reported in Brazil in 1998 and 1999, respectively. Latin America has considered building large-scale water treatment and sanitation systems, at a cost of $200 million.[111] However, efforts have been focused on smaller-scale alternatives, such as chlorine disinfectant tablets for use in treating household water.

18.4.2.4 Europe and Oceania

In Europe and Oceania, nearly all of the cholera cases are imported. The 35 infections reported in Europe in 2000 were all acquired overseas.[105] In Oceania, some countries experienced new cases after a decade of being free of endemic cholera. Cholera in the United Kingdom, Australia, and New Zealand were all imported.

18.5 Influenza

18.5.1 Origins

The "Spanish" influenza killed 20 million to 40 million people worldwide from 1918 to 1920. This was the worst pandemic invasive disease in history. Type A (H1N1) influenza virus, which is similar to one found in pigs, caused the 1918 pandemic.[112] The three types of influenza viruses are A, B, and C, but only A (and subtypes of A) and B have greatly affected human populations.[113] Type B viruses primarily infect humans, whereas type A and subtype A viruses can also infect birds, pigs, horses, and other mammals.[114] Influenza viruses found in pigs and birds are closely related to each other.[115] In analyzing influenza outbreaks from 1976, 1986, and 1988, it was found that the strains either were from swine or swine-derived viruses from avian sources.[114] In the 1957 (H2N1), 1968 (H3N1), and 1997 (H5N1) flu pandemics, it is believed that those viral strains originated from birds.[116] Studies of those three pandemics concluded that the three A subtypes diverged from a Eurasian group of avian strains.[117] In comparison, the examination of the 1918 pandemic suggests that the A subtype evolved from either Eurasian or North American avian strains. It is estimated that the influenza virus crossed from aquatic birds to humans about 10,000 years ago.[118]

18.5.2 Surveillance and response

The World Health Organization has a network of 110 National Influenza Centers in 83 countries and four Collaborating Centres for Virus Reference and Research located in the United States, Australia, Japan, and the United Kingdom.[114] In addition, the WHO's plan for Global Management and Control of an Influenza Pandemic concentrates on extensive surveillance and vaccine development. Vaccination is the most effective and efficient means of influenza prevention. The WHO is responsible for recommending new vaccine combinations each year. Annually, an average of 200 million influenza vaccine dosages are produced and globally distributed.[119] Influenza vaccinations have helped to reduce direct and indirect costs of influenza. From 1971 to 1978, vaccines saved $250 million per year.[120] Medical costs to patients 65 years and older dropped by $6.6 million each year. Although the distribution of 115 million doses of influenza vaccines cost a total of $808 million, this resulted in 13 million years of additional healthy life. The estimated value of one more year of healthy life is $63 per year.

Between 10 and 15% of the world's population comes down with influenza each year.[120] The type A virus causes the majority of outbreaks and cases. Those who are at greatest risk include the elderly, pregnant women, individuals with compromised immune systems, and people exposed to high-risk groups.[62] The number of deaths from influenza depends on the age group, the viral strain, and other factors. Most deaths are of individuals from high-risk groups. The medical expenses of treating influenza can come to $4.6 billion each year, while direct and indirect costs related to lost work days, wages, etc., can add up to $12 billion.[121] During pandemics, the situation worsens. For example, in the 1957 and 1968 pandemics, economic losses and medical expenses added up to $32 billion.[114]

18.5.2.1 United Kingdom

In the 1989–90 influenza epidemic in the United Kingdom, 39% of the patients hospitalized with the disease died.[122] In 1989, a total of 26,000 people died from the flu and its complications.[120] In the 1994–95 flu season, 6 million vaccines were distributed, at a cost of more than £30 million.

18.5.2.2 Australia

In Australia, 4.2 million people are at risk for influenza.[120] Most individuals are affected with type A (H3N2) virus.[123] The number of hospitalizations ranges from 18,000 to 37,000 per year. The number of reported deaths is anywhere from 800 to 2700, but it is likely that many deaths are not reported.[120]

18.5.2.3 New Zealand

The flu infects about 2.7% of New Zealand's population.[124] The type A (H3N2) virus is the predominant strain circulating in New Zealand; however, the B strain has emerged as well. Influenza incidence ranges from 50/100,000 to 249/100,000. Rates are highest among infants under 1 year of age (775.9/100,000) and lowest among the elderly (206.9/100,000). From 1990 to 1998 there were 307 influenza-related deaths. Ninety-four percent of deaths were from the age group 65 years and older. However, death rates among the indigenous Maoris (1.6/100,000) were higher than the death rates among Europeans (0.9/100,000). To better help the elderly fight influenza, New Zealand's vaccination coverage grew from 39% in 1997 to 55% in 1999.

18.6 Hepatitis

18.6.1 Origins

Little is known about the origins and evolution of viral hepatitis. Hepatitis is one of the oldest diseases in humans. The main human hepatitis viruses are A, B, C, D, E, F, and G.[125] Studies of hepatitis B virus (HBV) have resulted in a range of theories. The HBV in humans has six genotypes.[126] While genotypes A and D are distributed worldwide, the B and C genotypes are prevalent in East and Southeast Asia, genotype E is predominant in West Africa, and the F genotype is only found in indigenous populations within Central and South America.[127] It is believed that these genotypes diverged from a common ancestor 2300 to 3100 years ago.[126] One theory proposed that HBV co-evolved with humans as they migrated from Africa about 100,000 years ago.[128] Another study suggested HBV originated from the co-speciation of gibbons, orangutans, and chimpanzees from woolly monkey species more than 10 million to 35 million years ago.[129] It is widely believed that HBV originated from primates and that the close contact between primates and humans allowed for a cross-species transmission. Despite the countless studies and numerous theories, the evolutionary nature of HBV as well as that of other hepatitis viruses remains uncertain.

18.6.2 Hepatitis A

Hepatitis A virus (HAV) was identified in 1973. The virus is spread through feces, semen, saliva, or blood of infected people. The main mode of transmission is the fecal-oral route.[130] However, HAV can be contracted through anal-oral sexual contact and blood transfusions as well. In 1990, 28,000 cases were reported from North America, 162,000 from Central and South America, 278,000 from Europe, 251,000 from Africa and the Middle East, 676,000 from Asia, and 5000 from Oceania.[125] In total, there were 1,399,000 cases recorded in 1990. At present, 1.4 million HAV infections are reported each year. The incidence of HAV is 20% higher in men than in women. Worldwide, treatment for HAV costs $1.5 billion to $3 billion annually.[131] Two doses are required for a complete vaccination; vaccines cost an average of $40 per dose, plus a $15 fee for administration.[132] HAV hospital care for one person could cost $6900.[133] Medical expenses and financial losses due to HAV are high. In the United States, $500 million a year goes toward medical treatment and compensation for lost work.

18.6.2.1 South Africa

HAV endemicity is high in South Africa and very common among the country's black population;[134] by 20 years of age, nearly all black South Africans have HAV antibodies. In contrast, HAV endemicity is relatively low in the country's white population; by 20 years of age, 30 to 40% had HAV antibodies.

HAV infections are steadily increasing in South Africa. In 1990, the HAV incidence for whites was 15.8/100,000; for Asians, 5.7/100,000; for blacks, 0.5/100,000.[135] Growing HAV incidence in South Africa has prompted the government to improve sanitation and hygiene in the country. South Africa has also made efforts to improve the socio-economic conditions faced by poorer populations.

18.6.2.2 Australia

HAV is a notifiable disease in Australia.[136] According to the National Notifiable Disease Surveillance System (NNDSS), 1500 to 3000 cases were reported between 1991 and 1999. Incidence in the states and territories ranged from 1.9/100,000 to 51.7/100,000. More outbreaks have occurred in homosexual communities.[137] New HAV emergence was also found in low socio-economic groups.[138] However, the largest outbreak was from oyster consumption.[139] Australia's National Health and Medical Research Council reinforces vaccination and hygiene as key preventive measures, in addition to quality surveillance.

18.6.3 Hepatitis B

While 2 billion people worldwide have HBV, more than 350 million of them suffer from chronic infections.[140] People with chronic hepatitis become more susceptible to other diseases, such as cirrhosis and liver cancer. Eight to ten percent of HBV cases will develop into chronic hepatitis. In regions in sub-Saharan Africa, Asia, and the Pacific, most individuals contract HBV during childhood. Ninety percent of infants are infected within 1 year of age. Chronic hepatitis will affect 30 to 50% of the children with HBV. The virus is spread through perinatal and child-to-child transmissions, sexual contact, transfusions, and injecting drug use (IDU). The HBV vaccine has dramatically decreased the number of HBV infections and chronic hepatitis cases. Since 1982, more than 1 billion HBV vaccines were globally distributed. In 1991, the WHO recommended that HBV vaccines were to be included in a country's immunization program. WHO's plan is to immunize as many people as possible. The Global Alliance for Vaccines and Immunization (GAVI) was created in order to help fund national vaccination programs in low-income countries.

18.6.3.1 India

In 1995 HBV affected 36 million people in India, a number that represents 9% of the worldwide HBV cases in that year.[141] HBV is responsible for 70% of chronic hepatitis and 80% of cirrhosis cases.[142] Blood transfusions are the main route of infection among adults. In Delhi, a study reported that of the 7% of blood recipients acquired infected hepatitis by transfusions, 20% had the hepatitis B virus.[143] Professional blood donors represent a major HBV risk in India. Professional blood donors will continue to be a threat until national vaccination programs and other control measures are taken.

18.6.3.2 United Kingdom

The United Kingdom has a low HBV prevalence. Most individuals contract the virus through sexual contact or IDU.[144] Between 1985 and 1996, 9252 cases were reported. Most of the infections were observed in 15- to 24-year-olds and 25- to 44-year-olds. More than 66% of the cases were men who had sex with men (MSM) or were IDUs. Needle exchange programs designed toward HIV prevention have aided in HBV reduction as well. A universal vaccination program can reduce chronic infections for infants, children, adolescents, and adults.[145]

18.6.3.3 Brazil

Brazil's HBV incidence ranges from high in the western Amazon Basin to low in southern Brazil.[146] However, there are areas in southern Brazil with HBV prevalence. More men than women are infected with HBV. A report studying central Brazil showed that HBV incidence was higher in immigrants than in natives.

18.6.3.4 South Africa

Out of the 33 million black South Africans, 3.3 million suffered from chronic HBV in 1996.[147] The HBV incidence for black South Africans is 29/100,000. The infection rate is higher in rural areas than in cities. In the rural area of Transkei, HBV was present in 15.5% of the population, while it was present in only 7.4% of Durban's population.[148] In 1994, South Africa adopted a universal immunization program.[147] It provides vaccinations for newborns and infants up to 1 year old.

18.6.3.5 Australia and New Zealand

HBV incidence in Australia and New Zealand is low. Most cases of infection are found in indigenous groups, immigrants, IDU, and homosexual men.[149]

18.6.4 Hepatitis C

Three percent of the world population, or 170 million people, are infected with the hepatitis C virus (HCV).[150] Some 3 million to 4 million additional people will be infected each year.[151] In Africa, the Americas, Europe, and Southeast Asia, anti-HCV antibody prevalence is below 2.5%.[150] In the Western Pacific regions, prevalence is between 2.5 and 4.9%, while in the Middle East it can range from 1% to over 12%. Modes of HCV transmission include unscreened blood products, IDU, promiscuous sexual behavior, and non-sterilized instruments used either for rituals or for medical/dental procedures. The WHO reports that 80% of new cases will progress into chronic hepatitis, and 10 to 20% of these chronic infections will develop into cirrhosis. There is no HCV vaccine available yet. However, preventive measures recommended by the WHO, such as screening blood donors, providing health education programs about HCV transmission, and promoting community awareness, should be taken.

References

1. Gao, Feng., et al. Origin of HIV-1 in the chimpanzee *Pan troglodytes troglodytes. Nature*, 397, 436–441 (1999).
2. Hirsch, V.M., et al. An African primate lentivirus (SIVsm) closely related to HIV-2. *Nature,* 339, 389–392 (1989).
3. Vandamme, A.M., et al. Dating HIV and HCV epidemics with a new method to uncover clock-like molecular evolution: HIV-1 group M originated during the 1930s (Abstract). XIII International AIDS Conference in Durban, South Africa. 9–14 July 2000, www.iac2000.org (Retrieved 31 July 2001).
4. Joint United Nations Programme on HIV/AIDS, AIDS epidemic update: December 2000. Geneva, Switzerland.
5. Joint United Nations Programme on HIV/AIDS, Report on the global HIV/AIDS epidemic June 2000. Geneva, Switzerland.
6. National Institute of Allergy and Infectious Diseases (National Institute of Health and US Department of Health and Human Servies). NIAID Global Health Research Plan for HIV/AIDS, Malaria and Tuberculosis. www.niaid.nih.gov/default.htm (Retrieved on 10 June 2001).
7. World Health Organization. Affordable AIDS drugs are within reach. Note for the Press No.2: 14 February 2001. www.who.int/inf-pr-2001/en/note2001–02.html (Retrieved on 22 June 2001).
8. World Health Organization. WHO welcomes US announcement to contribute US$200 million to a global HIV/AIDS health fund. WHA 54/3:14 May 2001. www.who.int/inf-pr-2001/en/note2001-WHA3.html (Retrieved in 22 June 2001).
9. Smith, Davis N., AIDS watch. *Pharmaceutical Executive,* 21, 132 (2001).
10. Schwärtlander, B., et al. Resource needs for HIV/AIDS. *Science,* 292, 2434–2436 (2001).
11. Bloom, David E., et al. Something to be done: treating HIV/AIDS. *Science*, 288, 2171–2172 (2000).
12. Mutume, Gumisai, Pressure mounts for cheaper anti-AIDS treatment. *Population Reference Bureau.* February 2001. www.prb.org/regions/africa/pressure.html (Retrieved on 8 June 2001).
13. South African Department of Health, HIV/AIDS/STD Strategic Plan for South Africa 2000–2005. www.doh.gov.za/aids/index.html (Retrieved on 10 June 2001).
14. The impending catastrophe: a resource book on the emerging HIV/AIDS epidemic in South Africa. Abt Associates of South Africa, Inc. The Henry J. Kaiser Family Foundation. www.kff.org/content/2000/20000515a/ (Retrieved on 10 June 2001).
15. Karim, Salim S. Abdool., HIV and TB. *Lancet,* 349, 1542–1543 (1997).
16. Smith, Anthony M.A., et al. Australian and South African undergraduates' HIV-related knowledge, attitudes and behaviors. *Archives of Sexual Behavior* 27, 279–294 (1998).
17. Cohen, Jon., Ground zero: AIDS research in Africa. *Science,* 288, 2150–2153 (2000).
18. Hwang, Ann., AIDS has arrived in China and India. *World Watch*, 14, 12–20 (2001).
19. Population Reference Bureau., Only 4 out of 10 Indian Women have heard of AIDS. www.prb.org/press/HIV_AIDS/Indian_Women_AIDS.html (Retrieved on 8 June 2001).

20. Sharma, Dinesh, India to develop an AIDS vaccine. *Lancet*, 357, 1024 (2001).
21. Centers for Disease Control and Prevention. HIV prevention strategic plan through 2005, January 2001. www.cdc.gov/hiv/dhap.htm (Retrieved on 1 August 2001).
22. Center for Disease Control and Prevention. HIV/AIDS Surveillance Report, 2000, 12 (no.2) [5, 14, 27, 30] www.cdc.gov/hiv/stats/hasr1202.htm (Retrieved on 1 August 2001).
23. Sekles, Flavia, Brazil's needle-exchange programs reduce HIV risk for drug users. *Population Reference Bureau*, 2001 www.prb.org/regions/lac/Brazil_Needle.html. (Retrieved on 2 June 2001).
24. Ministry of Health Brazil. AIDS Drug Policy. www.aids.gov.br/assistencia/aids_drugs_policy.html (Retrieved on 20 June 2001).
25. Yamey, Gavin, U.S. trade action threatens Brazilian AIDS programme. BMJ, 322, 7283 (2001).
26. Public Health Laboratory Service. HIV/AIDS & STD Surveillance in the United Kingdom. www.phls.co.uk/facts/hiv/old/hivold.htm (Retrieved on 1 June 2001).
27. Unlinked Anonymous Surveys Steering Group. Prevalence of HIV and hepatitis infections in the United Kingdom 1999. London; Department of Health, Public Health Laboratory Service, Institute of Child Health (London), Scottish Centre for Infection and Environmental Health. 2000 www.phls.co.uk/facts/HIV/UAPMP (Retrieved on 1 June 2001).
28. Commonwealth Department of Health and Family Services. Partnerships in Practice: National HIV/AIDS Strategy 1996–1997 to 1998–1999(December 1996). Australian Department of Health and Aged Care. www.health.gov.au/hfs/pubhlth/publicat/hac.htm. (Retrieved on 2 June 2001).
29. Public Health Laboratory Service. PHLS CDSC HIV/AIDS CORE SLIDE SET. www.phls.co.uk/facts/yfever.htm (Retrieved on 1 June 2001).
30. Wodak, Alex and Lurie, Peter., A tale of 2 countries. *Journal of Drug Issues*, 27, 117–134 (1997).
31. Walford, Diana and Noah, Norman., International Editors. *Emerging Infectious Diseases*, 5 (online), London; Public Health Laboratory Service (1999). www.cdc.gov/ncidod/eid/vol5no2/walford.html (Retrieved on 20 June 2001).
32. Lichtenstein, Bronwen., Tradition and Experiment in New Zealand AIDS policy. *AIDS and Public Policy Journal*, 12, 79–88 (1997).
33. Ministry of Health., AIDS and New Zealand, 39 November 1998. www.moh.govt.nz (Retrieved on 9 June 2001).
34. Bloom, Barry R. and Murray, Christopher J.L., Tuberculosis: commentary on reemergent killer. *Science*, 257, 1055–1064 (1992).
35. Tynes, L Lee., Tuberculosis: The continuing story. *JAMA*, 270, 2616–2617 (1993).
36. Fackelman, Kathleen., Tuberculosis outbreak. *Science News*, 153, 73–75 (1998).
37. Manchester, Keith., Tuberculosis and leprosy: evidence for interaction of disease in *Human Paleopathology*, Aufderheide, Arthur C. and Ortner, Donald J. (Eds.), Smithsonian Institution Press, Washington, 1991, 23.
38. Kapur, V., Whittman, T.S., and Musser, J.M., Is *Mycobacterium tuberculosis* 15,000 years old? *J Infect Dis*, 170, 1348–1349 (1994).
39. Daniel, T.M., The origins and precolonial epidemiology of tuberculosis in the Americas: can we figure them out? *Int J Tuberc Lung Dis*, 4, 395–400 (2000).
40. Salo, Wilmar L.I., et al. Identification of Mycobacterium tuberculosis DNA in a pre-Columbian Peruvian mummy., *Proc. Natl. Acad. Sci.*, 91, 2091–2094 (1994).
41. Gibbons, A., The peopling of the Americas. *Science News*, 274, 31–33 (1996).
42. Hurtago de Mendoza, D. and Braginiski, R.Y., Y chromosomes point to native American Adam. *Science*, 283, 1439–1440 (1999).
43. Wiess, K.M., American origins. *Proc. Natl. Acad. Sci.*, 91, 833–835 (1994).
44. World Health Organization. Global Tuberculosis Control. WHO Report 2001. Geneva, Switzerland, WHO/CDS/TB/2001.287 (Retrieved 30 May 2001)
45. Revised National TB Control Programme (RNTCP). TB India. Central TB Control, Directorate General of Health Services, Ministry of Health and Family Welfare. www.tbcindia.org (Retrieved on 16 June 2001).
46. De Cock, Kevin M. and Dworkin, Mark S., HIV infection and TB. *World Health*, 51, 14 (1998).

47. World Health Organization. Tuberculosis. WHO Fact Sheet 2000 N°104. Geneva, Switzerland. www.who.int/gtb (Retrieved on 30 May 2001).
48. World Health Organization. HIV causing tuberculosis cases double in Africa (Press Release WHO/21:23 April 2001). Geneva, Switzerland. www.who.int/inf/pr-2001/en/pr2001.html (Retrieved on 30 May 2001).
49. World Health Organization. WHO and STOP TB launch new initiative to increase access to free TB drugs: The Global TB Drug Facility (Press Release WHO/13:20 March 2001). Geneva, Switzerland. www.who.int/inf-pr-2001/en/pr2001/-13.html (Retrieved on 30 May 2001).
50. World Health Organization. $1.5 billion need to meet TB control targets in highest burden countries (Press Release WHA 54/3: 16 May 2001). Geneva, Switzerland. www.who.int/gtb/press/index.htm (Retrieved on 30 May 2001).
51. STOP TB. Country Profiles. www.stootb.org (Retrieved on 13 June 2001).
52. Fourie, Bernard., TB in South Africa-The burden of tuberculosis in South Africa. MRC National Tuberculosis Research Programme, South Africa. www.sahealthinfo.org/Publications/body_tb/body_body_tb.htm (Retrieved on 4 June 2001).
53. Global Corps. South Africa. www.globalcorps.com/orgs/ngo/dotw/safrica.html (Retrieved on 4 June 2001).
54. Walford, Diana and Noah, Norman., *Emerging Infectious Diseases*, 5 (online), London; Public Health Laboratory Service (1999). www.cdc.gov/ncidod/eid/vol5no2/walford.html (Retrieved on 20 June 2001).
55. Hayward, A.C., and Watson, J.M., Tuberculosis in England and Wales, 1982 to 1993:Notifications exceeded predictions. *Comm Dis Rep CDR Review,* 5, 29–33 (1995).
56. Public Health Laboratory Service., Press Release 26 January 2001. www.phls.co.uk/new/pressreleases/01pr/010126.htm (Retrieved on 20 June 2001).
57. Public Health Laboratory Service., Tuberculosis Briefings. www.phls.co.uk/facts/tb/briefing.htm (Retrieved on 20 June 2001).
58. Joint Tuberculosis Committee of the British Thoracic Society (BTS). Control and prevention of tuberculosis in the United Kingdom: Code of Practice 2000. *Thorax,* 55, 887–901 (2000).
59. Sheldon, C.D., et al. Notification of tuberculosis:how many cases are never reported? *Thorax,* 47, 1015–1018 (1993).
60. Evans, Meirion R., Is tuberculosis taken seriously in the United Kingdom? *BMJ*, 311, 1483–1485 (1995).
61. Csillag, Claudio., Bonuses proposed for TB cures in Brazilian health centres. *Lancet*, 352, 1453 (1998).
62. Pan American Health Organization (PAHO). Brazil: Basic country profiles: summaries 1999. www.paho.org/SHA/prflbra.html (Retrieved on 30 July 2001).
63. Pan American Health Organization (PAHO). The use of socioeconomic factors in mapping tuberculosis risk areas in a city of northeastern Brazil. *Pan American Journal of Public Health (online),* 8¸December 2000. www.paho.org/English/DVI/ES.v8n6.html (Retrieved on 17 June 2001).
64. Queensland Government-Queensland Health. Tuberculosis in Australia and New Zealand into the 1990s. www.health.qld.gov.au/phs/shs/CONAUS19900202v1.htm (Retrieved on 13 June 2001).
65. Dawson, David., Tuberculosis in Australia: bacteriologically confirmed cases and drug resistance, 1996. WHO Collaborating Centre in Tuberculosis Bacteriology, Queensland Health Pathology Services, Brisbane. www.health.gov.au.pubhlth/cdi/cdi2209/cdi2209.htm (Retrieved on 13 June 2001).
66. Isaacs, David and Mellis, Craig M., Tuberculosis in children in Australia: strategies for control. MJA, 168, 121–124 (1998).
67. Mycobacterium Reference Laboratory. The Australian Tuberculosis Project. www.vidrl.org.au/mrl/atb.htm (Retrieved on 13 June 2001).
68. Ministry of Health. No room for complacency about TB in New Zealand says Ministry of Health. Media Release: 24 March 2001.www.moh.govt.nz (Retrieved on 13 June 2001).
69. Ministry of Health. World TB Day is a reminder that the disease still exists. Media Release: 26 March 2001. www.moh.govt.nz (Retrieved on 13 June 2001).

70. New Zealand Health Information Service (NZHIS). Health Statistics. www.nzhis.govt.nz/stats/diseases/tuberculosis.html (Retrieved on 13 June 2001).
71. Escalante, Ananias A. and Ayala, Francisco., Evolutionary origin of *Plasmodium* and other Apicomplexa based on rRNA genes. *Proc. Natl. Acad. Sci.*, 92, 5793–5797 (1995).
72. Escalante, Ananias A. and Ayala, Francisco., Phylogeny of the malarial genus *Plasmodium*, derived from rRna gene sequences. *Proc. Natl. Acad. Sci.*, 91, 11373–11377 (1994).
73. Escalante, Ananias A., et al. The evolution of primate malaria parasites based on the gene encoding cytochrome b from the linear mitochondrial genome. *Proc. Natl Acad. Sci.*, 95, 8124–8129 (1998).
74. Conway, David J., Origin of *Plasmodium falciparum* malaria is traced by mitochondrial DNA. *Molecular and Biochemical Parasitology*, 111, 163–171 (2000).
75. World Health Organization. Malaria. Fact Sheet No 94: 1998 October. www.who.int/inf-fs/en/fact094.html (Retrieved on 24 July 2001).
76. Roll Back Malaria (RBM). www.rbm.who.int (Retrieved on 25 July 2001).
77. Kapp, Clare., WHO wins reprieve for DDT against malaria. *Lancet*, 356, 2076 (2000).
78. Yamey, G., Global campaign to eradicate malaria. *BMJ*, 322, 1191–1192 (2001).
79. Dua, V.K., et al. Chloroquine resistant *Plasmodium vivax* malaria in India. *Tropical Medicine and International Health*, 1, 816–819 (1996).
80. Sharma, V.P., Re-emergence of malaria in India. *Indian Journal of Medical Research*, 103, 26–45 (1996).
81. Jayaraman, K.S., India plans $200 million attack on malaria. *Nature*, 386, 536 (1997).
82. Pan American Health Organization (PAHO). Situation of malaria programs in the Americas. www.paho.org/English/SHA/be_v22n1-malaria.htm (Retrieved on 22 July 2001).
83. Lucia, Ana., et al. High prevalence of asymptomatic malaria in gold mining areas in Brazil. *Clinical Infectious Diseases*, 12, 79–88 (1997).
84. Milliken, William., Malaria and antimalarial plants in Roraima, Brazil. *Tropical Doctor*, 27 (suppl), 20–25 (1997).
85. Akhavan, Dariush., et al. Cost-effective malaria control in Brazil-cost-effectiveness of a malaria control program in the Amazon Basin of Brazil, 1988–1996. *Social Science and Medicine*, 49, 1385–1399 (1999).
86. D' A Gusmao, Renato., Cost-effective control in Brazil. *World Health*, 51, 30 (1998).
87. South African Department of Health. Malaria: current situation in South Africa. www.doh.gov.za/pr/1996/pr0308b.html (Retrieved on 22 July 2001).
88. South African Department of Health. Statistics Notes. 2 (2000). www.doh.gov/za/search/index.html (Retrieved on 22 July 2001).
89. Anon. Malaria costs Africa $300 billion. *African Business*, 255, 28 (2000).
90. Medical Research Council of South Africa. MIM:A global movement against malaria. www.mrc.ac.za.mrcnews/jun99/mim.html (Retrieved on23 July 2001).
91. Jenkin-Grant, A., et al. Airport malaria in Cairns. *MJA*, 166, 307–308 (1997).
92. Walker, John., Malaria in the changing world: an Australian perspective. *International Journal of Parasitology*, 28, 947–953 (1998).
93. Robinson, Priscilla., Imported malaria treated in Melbourne, Australia: Epidemiology and clinical features in 246 patients. *J Travel Med*, 8, 76–81 (2001).
94. Australia's notifiable disease status, 1998. *Communicable Disease Intelligence*, 23 (1998). www.health.gov.au:80/hfs/pubhlth/cdi/cdi2311/cdi2311g.htm (Retrieved on 25 July 2001).
95. Bryan, Joan H., Malaria transmission and climate change in Australia. *MJA*, 164, 345–347 (1996).
96. Orchard, Valerie, et al. The communicable disease picture in New Zealand today. *Healthcare Review Online*, 4 (2000). www.enigma.co.nz.hcro_articles/0005/vol4no5_001.htm (Retrieved on 24 July 2001).
97. Bradley, D.J. and Bannister, B., Guidelines for malaria prevention in travelers from the United Kingdom for 2001. *Communicable Disease and Public Health*, 4, 84–101 (2001).
98. Deb, B.C., Epidemiology of cholera in India and its treatment and control. *Current Science*, 59, 702–707 (1990).

99. Sanyal, Suhas C., Cholera in its present day scenario. *Journal of the Indian Medical Association*, 98, 371–376 (2000).
100. World Resources Institute. Cholera returns. *World Resources* 1998–1999. www.wri.org/wri/wr-98–99/cholera.html (Retrieved on 26 July 2001).
101. Cowell, Rita R., Global climate and infectious diseases: the cholera paradigm. *Science*, 274, 2025–2031 (1996).
102. World Health Organization. Global surveillance of epidemic-prone infectious diseases. Geneva, Switzerland. WHO/CDS/CSR/2000.1. www.who.int (Retrieved on 30 May 2001).
103. World Health Organization. Potential use of oral cholera vaccines in emergency situtations:report of a WHO meeting. Geneva, Switzerland. WHO/CDS/CSR/EDC/99.4 www.int/gtb (Retrieved on 30 May 2001).
104. Shimada, T., et al. Outbreak of Vibrio cholerae non-01 in Bangladesh. *Lancet*, 341, 1347 (1993).
105. World Health Organization. Cholera, 2000. *Weekly Epidemiological Record*, 76, 233–240 (2001).
106. World Health Organization. Cholera, 1999. *Weekly Epidemiological Record*, 75, 249–256 (2000).
107. Morris, Kelly., Prevention fails to halt South Africa's well-treated cholera epidemic. Lancet, 357, 290 (2001).
108. Sidley, Pat., Cholera sweep throughout South Africa province. BMJ, 322, 71–71 (2001).
109. Sur, D., et al. Severe cholera outbreak following floods in a northern district of West Bengal. *Indian Journal of Medical Research*, 112, 178–182 (2000).
110. Pan American Health Organization (PAHO). Number of cholera cases in the Americas 1991–2000. www.paho.org/English/HCP/HCT/Cholera_cases_1991–2000.htm (Retrieved on 2 September 2001).
111. Tauxe, Robert., et al. Epidemic cholera in the new world-translating field epidemiology into new prevention strategies. *Emerging Infectious Diseases*, 1 (online), 141–146 (1995). www.cdc.gov/ncidod/eid/vol1no4/tauxe.htm (Retrieved on 1 August 2001).
112. Taubenberger, J.K., et al. Initial genetic characterization of the 1918 'Spanish' influenza virus. *Science*, 275, 1793–1796 (1997).
113. World Health Organization. Influenza A (H5N1): fact sheet no. 188. www.who.int/inf-fs/en/fact094.html (Retrieved on 13 July 2001).
114. World Health Organization. The WHO influenza programme: fact sheet no. 212. www.who.int/inf-fs/en/fact212.html (Retrieved on 13 July 2001).
115. Webster, R.G., et al. Evolution and ecology of influenza A viruses. *Microbiology*, 56, 159–179 (1992).
116. World Health Organization. WHO report on global surveillance of epidemic-prone infectious diseases. WHO/CDS/CSR/ISR/2000.1 Geneva, Switzerland (Retrieved on 13 July 2001).
117. Reid, Ann H., et al. Origin and evolution of the 1918 "Spanish" influenza virus hemagglutinon gene. *Proc. Natl. Acad.*, 96, 1651–1656 (1999).
118. Smith, Rosalind Stanwell., The search for permafrost and other victims of the 1918 influenza-commentary. *Health and Hygiene*, 22, 9–13 (2001).
119. World Health Organization. Content of 2002 "Southern Hemisphere" influenza vaccine decided. Press Release WHO/41 18 September 2001. www.who.int/inf-pr-2001/en/pr2001–41.html (Retrieved on 19 September 2001).
120. Frequently asked questions. International Influenza Education Panel. www.influenza-news.com (Retrieved on 5 August 2001).
121. National Foundation for Infectious Diseases (NFID). Facts about influenza for adults. www.nfid.org/factsheets/influadult.html (Retrieved on 5 August 2001).
122. Wiselka, M., Influenza:diagnosis, management, and prophylaxis. *BMJ*, 308, 1341–1345 (1994).
123. Thomson, Jennifer, Ling, Ming and Hampson, Alan., Annual report of the National Influenza Surveillance Scheme. *Comm. Dis. Intell.*, 24, 145–152 (2000).
124. Jennings, Lance., Influenza surveillance and immunisation in New Zealand, 1990–1999, *New Zealand Public Health Report*, 8, 9–16 (2001).
125. World Health Organization. Hepatitis A. WHO/CDS/CSR/EDC/2000.7. Geneva, Switzerland. www.who.int/emc (Retrieved on 5 July 2001).
126. Simmonds, P., The origin and evolution of hepatitis viruses in humans. *Journal of General Virology*, 82, 693–712 (2001).

127. Arauz-Ruiz, P., et al. Genotype F prevails in HBV infected patients of Hispanic origin in Central America and may carry the precore stop mutant. *Journal of Medical Virology,* 51, 305–312 (1997).
128. Norder H., Complete genomes, phylogenetic relatedness, and structural proteins of six strains of the hepatitis B virus, for of which represent two new genotypes. *Virology,* 198, 489–503 (1994).
129. MacDonald, D.M., et al. Detection of hepatitis B virus infection in wild-born chimpanzees (*Pan troglodytes verus*): phylogenetic relationships with human and other primate genotypes. *Journal of Virology,* 74, 4253–4257 (2000).
130. Lemon, S.M., Type A viral hepatitis: epidemiology, diagnosis, and prevention. *Clinical Chemistry,* 43, 1494–1499 (1997).
131. André, F.E., Approaches to a vaccine against hepatitis A: development and manufacture of an inactivated vaccine. *Journal of Infectious Diseases,* 171 (suppl 1), S33-S39 (1995).
132. Tormans, G., et al. Cost-effectiveness analysis of hepatitis A prevention in travelers. *Vaccine*, 10 (suppl 1), S88-S92 (1992).
133. Berge, J.J., et al. The cost of hepatitis A infections in American adolescents and adults in 1997. *Hepatology,* 31, 469–473 (2000).
134. Karim, A., et al. Sero-epidemiology of hepatitis A in black South African children. *SAMJ*, 83, 748–750 (1993).
135. Sathar, M.A., et al. Racial differences in the sero-prevalence of hepatitis A virus infection in Natal/KwaZulu, South Africa. *Journal of Medical Virology,* 44, 9–12 (1994).
136. Fleet, G.H., et al. Foodborne viral illnesses:status in Australia. *International Journal of Food Microbiology,* 59, 127–136 (2000).
137. Stokes, M.L., Outbreak of Hepatitis A among homosexual men in Sydney. *American Journal of Public Health,* 87, 2039–2041 (1997).
138. Dick, A., et al. Persisting hepatitis A infection in a central Queensland town. C*omm. Dis. Intell.,* 18, 83–89 (1994).
139. Amin, J., et al. Hepatitis A epidemiology in Australia: national seroprevalence and notifications. *Medical Journal of Australia*, 174, 338–341 (2001).
140. World Health Organization. Hepatitis B: Fact sheet WHO/204. www.int/inf-fs/en/fact204.html (Retrieved on 5 July 2001).
141. Irshad, M., et al. Prevalence of hepatitis B virus infection in healthy person in North India. *Natl. Med.J. India.,* 7, 210–212 (1994).
142. Tandon, B.N., et al. Epidemiology of hepatitis B virus infection in India. *Gut,* 38 (suppl 2), S56-S59 (1996).
143. Tandon, B.N., et al. Etiological spectrum of viral hepatitis and prevalence of markers of hepatitis A and B virus infection in North India. *Bulletin of WHO,* 62, 67–73 (1984).
144. Balogun, M.A., et al. Acute hepatitis B infection in England and Wales: 1985–96. *Epidemiological Infections,* 122, 125–131 (1999).
145. Fenn, P., et al. An economic evaluation of universal vaccination against hepatitis B virus. *Journal of Infection,* 32, 197–204 (1996).
146. Souto, F.J.D., Epidemiological survey of infection with hepatitis B virus in the savannah and wetlands (Pantanal) of central Brazil. *Annals of Tropical Medicine and Parasitology, 9*1, 411–416 (1997).
147. Kew, M.C., Progress towards the comprehensive control of hepatitis B in black South African children. *SAMJ,* 83, 748–750 (1993).
148. Vos, G.H., et al. Hepatitis B antigen and antibodies in rural and urban southern African blacks. *South African J. Med.,* 57, 868–870 (1984).
149. Kaldor, J.M., et al. The incidence of hepatitis B infection in Australia: an epidemiological review. *Medical Journal of Aust.,* 165, 322–326 (1996).
150. Anon. Global surveillance and control of hepatitis C. *Journal of Viral Hepatitis*, 6, 35–47 (1999).
151. World Health Organization. Hepatitis C: fact sheet no.164. www.who.int/inf-fs/en/fact164.html (Retrieved on 19 July 2001).

section nine

World Overview

chapter seventeen

Economic and environmental threats of alien plant, animal, and microbe invasions*

David Pimentel, S. McNair, J. Janecka, J. Wightman, C. Simmonds, C. O'Connell, E. Wong, L. Russel, J. Zern, T. Aquino, and T. Tsomondo

Contents

* This chapter appeared in *Agriculture, Ecosystems, and Environment*, 84, 1–20, 2000. It is reprinted here with permission of Elsevier Science.

17.1 Introduction

Quantifying the environmental damage and loss of biodiversity due to alien species invasions worldwide is complicated by the fact that only some 1.5 million of the estimated 10 million species on earth have been identified and described.[1] The total numbers of introduced species in the United States, the United Kingdom, Australia, South Africa, India, and Brazil range from about 2000 to 50,000 species (Table 17.1). Many native species are threatened by competition and predation from invaders, while many other species are endangered by hybridization with alien species or major ecosystem changes caused by these species. Nonetheless, a total of more than 120,000 species of plants, animals, and microbes are known to have invaded these six nations; these alien species provide a base to assess several environmental threats such invaders pose (Table 17.1).

Given the number of species that have invaded the six nations studied here, we estimated that 480,000 alien species have been introduced into the varied ecosystems on earth. Many such introduced species, such as corn, wheat, rice, plantation forests, domestic chicken, cattle, and others, are beneficial and now provide more than 98% of the world's food supply — a value of more than $5 trillion per year.[2]

All crop and livestock species originated somewhere; the chicken, for example, is from Southeast Asia. Other alien species are used for landscape restoration, biological pest control, sport, pets, and food processing. However, alien species are also known to cause major economic losses in agriculture, forestry, and several other segments of the world economy; they also compromise ecological integrity.[3,4]

In the recent past, the rates and risks associated with alien species introductions have increased enormously, because human population growth and human activities that alter the environment have escalated rapidly.[4] Currently, there are 6 billion humans on earth.[5] Large numbers of people are traveling faster and farther, and more goods and materials are being traded among nations.[2,6] These human activities are accelerating the spread of alien species of plants, animals, and microbes worldwide.

This study assesses the magnitude of some of the environmental and economic impacts caused by alien plant, animal, and microbe invasions in the United States, the United Kingdom, Australia, South Africa, India, and Brazil. Although these nations have some of the best national data in the world, the total number of invading species is still unknown, making the assessment incomplete and extremely difficult. The lack of some data means that our economic and environmental information is underestimated.

17.2 Methodology

The approach employed in this investigation was to assemble all the published data on available invasive species in the countries under review. The number of alien species for each major group was totaled; these data are found in Table 17.1. Published information on the environmental impacts of non-indigenous species was also assembled. In addition, published data were assembled on the economic impacts of invasive species on crops, pastures, forests, public and livestock health, and natural ecosystems. In the cases where the data were available, control-cost data were tabulated and included with the economic cost data in Tables 17.2, 17.3, and 17.4.

17.3 Alien species in the United States, the United Kingdom, Australia, South Africa, India, and Brazil

Alien species cause major environmental and economic problems worldwide. In the United States, about 400 of the 958 species on the U.S. Threatened or Endangered Species

Table 17.1 Species Number per Category in the United States, United Kingdom, Australia, South Africa, India, and Brazil

	United States		United Kingdom		Australia		South Africa		India		Brazil	
Category	Total spp. number	Number Alien spp.	Total spp. number	Number Alien spp.	Total spp. number	Number Alien spp.	Total spp. number	Number Alien spp.	Total spp. number	Number Alien spp.	Total spp. number	Number Alien spp.
Plants	42,000[a]	25,000[a]	27,515[f]	26,000f	20,0001	1,952[m]	24,000[q]	8,750[r]	45,000[y]	18,000[z]	55,000[p]	11,605[ii]
Mammals	346[b]	20[c]	54[g]	17[g]	296[l]	20[l]	247[p]	16[s]	316[p]	30[aa]	428[jj]	25[kk]
Birds, reptiles, and amphibians	650[b]	97[a]	542[h]	47[h]	850[n]	70[n]	725[t]	8[t]	1,221[bb]	4[cc]	1,635[i]i	3[ll]
Fishes	247[b]	53[a]	80[i]	48[i]	700[o]	20[o]	394[u]	24[u]	741[dd]	NA	985[mm]	NA
(Freshwater)	938[e]	138[a]	54[j]	12[j]	216[o,p]	29[o,p]	220[v]	20[w]	2,546[ee]	300[ff]	3,000[mm]	76[ii]
Arthropods	650,000[e]	4,500[a]	24,700[k]	1,000[k]	85,920[ii]	150[ii]	86,000[w,x]	NA	54,430[gg]	1,100[hh]	1,000,000[nn]	NA
Microbes	134,644[d]	20,000[e]	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

a= 4, b=69, c=74, d=148-153, e=estimate, f=67, g=154,155, h=88, i=156, j=158, k=39, l=34, m=21, n=89,159, o=99, p=69, q=160, r=161, s=123, t=162, u=101, v=163, w=82, x=122, y=164, aa=170, bb=173, cc=174, dd=175, ee=175, ff=176, gg=177, hh=178, ii=70, jj=165, kk=166, ll=167, mm=168, nn=169.

Table 17.2 Economic Losses to Introduced Pests in Crops, Pastures, and Forests in the United States, United Kingdom, Australia, South Africa, India, and Brazil (billion dollars per year)

Introduced pest	United States	United Kingdom	Australia	South Africa	India	Brazil	Total
Weeds							
Crops	27.9	1.4	1.8	1.5	37.8	17.0[d]	87.4
Pastures	6.0	—	0.6	—	0.92	—	7.52
Vertebrates							
Crops	1.0[a]	1.2[b]	0.2[c]	—	—	—	2.4
Arthropods							
Crops	15.9	0.96	0.94	1.0	16.8	8.5	44.1
Forests	2.1	—	—	—	—	—	2.1
Plant pathogens							
Crops	23.5	2.0	2.7	1.8	35.5	17.1	82.6
Forests	2.1	—	—	—	—	—	2.1
Total	**78.5**	**5.56**	**3.24**	**4.3**	**91.02**	**42.6**	**228.72**

a = Losses due to English starlings and English sparrows[4]

b = Calculated damage losses from the European rabbit (see text)

c = 34

d = Pasture losses included in crop losses

— = data not available

List are considered at risk primarily because of competition with and predation by non-indigenous species.[7] In the fynbos region of South Africa, 80% of the threatened species are endangered because of invading alien species.[8] Most plant and vertebrate animal introductions have been intentional, whereas most invertebrate and microbe introductions have been accidental.[9] More than 120,000 species of plants, animals, and microbes have invaded the six nations investigated here, and many of these species are causing a wide array of damage both to managed and natural ecosystems (Table 17.1). Based on data available, the percentage of the total invading species for each nation is United Kingdom 53%, India 19%, South Africa 7%, United States 6%, Australia 3%, and Brazil 1% (Table 17.1).

17.4 Crop, pasture, and forest losses

Introduced plant, animal, and microbe species cause \$55 billion to \$248 billion per year in losses to world agriculture.[10]

17.4.1 Weeds

In crop systems, including forage crops, many intentionally introduced plant species have become weed pests.[11] Most weeds are accidentally introduced with crop seeds, from ship-ballast soil, or from various imported plant materials.[4]

In U.S. agriculture, weeds cause a reduction of 12% in potential crop yields. In economic terms, this reduction represents about a \$33 billion loss in crop production annually, based on the crop potential value of all U.S. crops of more than \$267 billion per year.[2] Based on the estimate that about 73% of the weeds are non-indigenous,[12] it is likely that about \$27.9 billion of these crop losses are due to introduced weeds (Table 17.2).

In U.S. pastures, 45% of weeds are alien species.[12] U.S. pastures provide about \$10 billion in forage crops annually,[13] and the estimated loss due to weeds is \$2 billion.[14] Since

Table 17.3 Environmental Losses to Introduced Pests in the United States, United Kingdom, Australia, South Africa, India, and Brazil (billion dollars per year)

Introduced Pest	United States	United Kingdom	Australia	South Africa	India	Brazil	Total
Plants	0.148[a]	—	—	0.095[j]	—	—	0.178
Mammals							
Rats	19.000[b]	4.100[k]	1.200[k]	2.700[l]	25.000[l]	4.00[l]	56.400
Other	18.106[c]	1.200[m]	4.655[n]	—	—	—	23.961
Birds	1.100[d]	0.270[o]	—	—	—	—	1.370
Reptiles & Amph.	0.006[e]	—	—	—	—	—	0.006
Fishes	1.000[f]	—	—	—	—	—	1.000
Arthropods	2.137[g]	—	0.228[p]	—	—	—	2.365
Mollusks	1.305[h]	—	—	—	—	—	1.305
Livestock diseases	9.000[i]	—	0.249[i]	0.100[i]	—	—	9.349
Human Diseases	6.500[i]	1.000[i]	0.534[i]	0.118[i]	—	2.33[i]	10.467
Total	**58.299**	**6.570**	**6.866**	**3.013**	**25.000**	**6.733**	**106.481**

a = A total of $45 million/yr in purple loosestrife control plus $100 million/yr in aquatic weed control[4]

b = 4

c = Damages from cats at $17 billion/yr, pigs $800.5 million/yr, dogs $250 million/yr, mongooses $50 million/yr, and horses and burros $5 million/yr[4]

d = Damages from pigeons[4]

e = Damages from the brown tree snake[4]

f = Damages from invading fish species[4]

g = Damages from the imported fire ant estimated to be $1 billion/yr, gypsy moth $11 million/yr, varroa mite $82 million/yr, Formosan termite $1 billion/yr, and green crab $44 million/yr[4]

h = Control/damage costs from the zebra mussel, Asian clam, and shipworm[4]

i = See text

j = About $40 million/yr for aquatic weed control; $50 million/yr for Working for Water; and $5 million/yr preventing plant invasions

k = Estimated 4.6 rats per capita and $15 damages /rat/yr[4]

l = Estimated 2.7 rats per capita in India (similar estimates for South Africa and Brazil)[75] and $10 damages/rat/yr[4]

m = Estimated 1 cat/3 people and 1/3rd are feral and each feral cat kills 2 birds/yr, then damages are calculated to be $1.2 billion/yr[4]

n = Estimated 18 million feral cats in Australia[82] and these cats kill 144 million birds/yr, then damages are calculated to be $4.3 billion/yr.[4] In addition, feral pigs are causing $80 million/yr in damages[34] and mice approximately $75 million/yr[179]

o = Estimated 0.5 pigeons/person and damages calculated to be $9/pigeon/yr, then damages are approximately $270 million/yr[4]

p = Three exotic insect and mite species cause $228 million/yr in damages to the wool industry[133]

— = data not available

about 45% of the weeds are alien,[12] the approximate forage losses due to non-indigenous weeds are nearly $1 billion a year. According to former Interior Secretary Bruce Babbitt,[15] ranchers spend about $5 billion a year to control invasive alien weeds in pastures and rangelands, but these weeds continue to spread (Table 17.2).

Most of the United Kingdom's alien plants occur in few habitats. More than 80% of alien species are present in waste ground areas, urban sites, roadsides, and similar disturbed habitats.[16] An estimated 63% of the alien plant species grow in hedges and scrub areas.[17] Vegetation in rock walls and woodlands consists of about 40% alien species.

Table 17.4 The Number of Humans with AIDS or HIV Infections and Health Care Costs (x 1 million) in the United States, United Kingdom, Australia, South Africa, India, and Brazil

	United States	United Kingdom	Australia	South Africa	India	Brazil
AIDS cases	103,533[a]	2,000[b]	730[c]	90,000[f]	80.000[k]	16,200[d]
HIV cases	650,000[l]	30,000[m]	11,080[e]	3,600,000[f]	21,000,000[g]	550,000[n]
Treatment costs/yr	$6,000[h]	$51[b]	$33.7[i]	$100[j]	NA	$800[o]

a = 180, b = 181, c = 182, d = 143, e = 141, f = 183, g = 184] h = 185, i = 186, j = 187, k= 188, l= 189, m= 190, n= 191, o= 192.

However, U.K. plant communities, such as grazed mesic grasslands and native *Pinus sylvestris* woodlands, contain no alien plant species.[17]

U.K. croplands and gardens contain 43% alien weeds.[17] In U.K. agriculture, weeds cause a reduction of about 10% in crop yields, but in some crops losses can be as high as 32%.[18,19] In economic terms, about $3.2 billion in total potential crop production is lost annually because of weed infestations. Given that about 43% of the weeds are alien, it is likely that $1.4 billion of the crop losses are caused by alien weeds (Table 17.2).

In Australia, some 60% of the weeds in crops are alien, based on a survey of major weeds in cereal crops.[20] The introduced blackberry (*Rubus proceus*) from Asia alone is causing $77 million a year in damage to crop production.[21] A total of 463 exotic grasses and legumes were intentionally introduced for forage; however, only four species proved beneficial for pasture and did not end up as weeds.[22] Indirect and direct losses due to pasture weeds are estimated to be $970 million a year.[23] Weeds cause an estimated $4 billion a year in total damage to cropland and pastures combined.[21] Since 60% of these weeds are alien, they account for about $2.4 billion per year in losses to agriculture[24] (Table 17.2).

In South Africa, the reduction in crop production due to all weeds is 16.6%,[19] at a cost of about $2.2 billion per year of potential production. Assuming that 67% of the weeds in the crops are alien,[25] these species account for total crop losses of about $1.5 billion a year (Table 17.2). Two of the most serious alien weeds in South African pasture lands are the shrub (*Lantana camara*) and the cactus plant (*Opuntia ficus-indica*), both introduced from Central America.[26–28]

In India, weeds are estimated to cause a 30% loss in potential crop production,[28] worth about $90 billion a year in reduced crop yields. Assuming that 42% of the weeds in crop production are alien,[29,30] the total cost of alien weeds to India is about $37.8 billion per year (Table 17.2).

Lantana camara, a major weed shrub in India, was introduced from Australia as an ornamental plant. It has invaded the majority of Indian pasture lands (13.2 million ha), as well as other areas.[28] *Lantana* is toxic to cattle, and the cost of its control is $70 per ha.[28] Since about 4% of India's land area is pasture, the damage from *Lantana* is estimated to be $924 million per year (Table 17.2).

In Brazil, alien weed species now make up 75% of the weed species in crop production areas.[31] Alien weeds destroy an estimated 13.4% of the country's crop and pasture production,[19] causing about $17 billion a year in losses (Table 17.2). These and other invasive plants change key natural ecosystems, alter fire regimes, and reduce the resources available to native animals.

17.4.2 Vertebrate pests

The English or house sparrow (*Passer domesticus*) and the European starling (*Sturnus vulgaris*) were both introduced into the United States. Both birds have become agricultural pests, together causing an estimated $1 billion a year in crop damages[4] (Table 17.2).

The European rabbit (*Oryctolagus cuniculus*) is abundant in the United Kingdom and Australia. U.K. rabbit densities may reach 30 per ha.[32] The animals are reported to reduce wheat production from 5 to 8% and livestock forage production by about 20%.[33] Assuming a conservative 10 rabbits per ha on cropland, where each rabbit causes $11 of damage per year,[32] the total crop damage from European rabbits amounts to about $800 million a year. Rabbits also do $400 million a year in damage to pastures. Thus the total damage from the European rabbit in the United Kingdom comes to $1.2 billion a year (Table 17.2).

In Australia, European rabbits also damage forage, causing losses that range from $90 million to $100 million per year.[34] Approximately 15 rabbits consume the equivalent pasture forage needed by one sheep (*Ovis aries*).[34] The impact on sheep production per year is estimated to be $110 million, including reduced sheep production and the cost of rabbit control. Total costs of rabbit damage to various aspects of agriculture are estimated to be $200 million per year (Table 17.2), but this does not include the land degradation caused by these animals.

17.4.3 Insect and mite pests

Pest insects and mites destroy about 13% of potential crop production, representing a value of about $33 billion in U.S. crops.[35] Based on the fact that about 40% of these pests are alien species,[12] the alien pests probably cause about $15.9 billion in crop losses each year (Table 17.2).

Furthermore, about 360 alien insect and mite species have become established in U.S. forests.[36] Insects cause the loss of approximately 9% of forest products, amounting to $7 billion per year.[2,37] Because 30% of the pests are alien species, annual losses attributed to them are about $2.1 billion per year (Table 17.2).

An estimated 1500 species of arthropods in the United Kingdom cause economic damage; about 30% of these are alien species.[38,39] Each year, arthropods damage or destroy approximately $3.2 billion in crops in the United Kingdom, based on 10% crop losses.[19] With about 30% of the losses in crops due to alien arthropods, the loss attributed to them amounts to $960 million per year (Table 17.2).

An estimated 36% of the pest arthropods in Australia are alien species.[40] The gross potential crop production in Australia is estimated at $24 billion per year. Crop losses due to insects and mites in Australia are estimated to be 10.7% of potential production.[19] Based on total crop losses to arthropods of about $2.6 billion per year, alien pests account for crop losses of about $936 million per year (Table 17.2).

Insect and mite pests in South Africa cause losses of 16.7%[19] of potential crop production each year, a portion worth $2.3 billion a year. Approximately 45% of the insect and mite pests are alien species[41]; thus the economic crop losses caused by introduced arthropods in South Africa are estimated to be $1 billion a year (Table 17.2).

Several hundred arthropod species in India are crop pests. Approximately 30% of the insect and mite crop pest species are alien.[42,43] Arthropods as a group reduce potential crop production by 18.7% in India.[19] Based on total potential crop production in India, crop losses to alien arthropods total $16.8 billion a year (Table 17.2).

In Brazil, about 14.4% of potential crop production is destroyed by insects and mites.[19] Approximately 35% of these species are alien.[31] The crop loss caused by alien insects and mites is estimated to be $8.5 billion a year (Table 17.2).

17.4.4 Plant pathogens

U.S. crop losses due to plant pathogens total approximately $33 billion per year.[2,35] Since 65% of all plant pathogens are alien species,[12] an estimated $23.5 billion a year can be attributed to alien plant pathogens (Table 17.2).

In U.S. forests, more than 20 non-indigenous species of plant pathogens attack woody plants.[36] Approximately 9% — a total of $7 billion — of forest products are lost annually due to plant pathogens.[2,37] Assuming that the proportion of alien plant pathogens in forests is similar to that of introduced insects, or about 30%, then approximately $2.1 billion in forest products are lost each year to non-indigenous plant pathogens in the United States (Table 17.2).

In the United Kingdom, an estimated 74% of the plant pathogens are introduced species.[44] Most of these alien plant pathogens were brought into the United Kingdom with seeds and other crop parts needed for agriculture. The economic loss due to plant pathogens amounts to 8.3% of potential production, or about $2.7 billion a year.[19] If 74% of the losses are due to alien plant pathogens, then about $2 billion a year in damages are associated with alien plant pathogens attacking crops (Table 17.2).

Total potential crop production in Australia is approximately $22 billion a year, with about 15.2% of crop losses being attributed to plant pathogens.[19] The economic losses from all plant pathogens is about $3.3 billion a year. Because a large number of plant pathogens are introduced when crop seeds and other plant parts are brought in, an estimated 82% of all crop plant pathogens are alien species (based on plant pathogens in field crops).[45] With 82% of Australian plant pathogens being alien, about $3 billion worth of crops are lost each year due to alien plant pathogens (Table 17.2).

Based on an assessment of diseases of fruits and vegetables,[46,47] approximately 85% of the plant pathogens attacking crops in South Africa are considered to be introduced. Most of these pathogens came in with crop introductions. In total, plant pathogens in South Africa cause an estimated 15.6%,[19] or $2.1 billion a year, of loss of potential crop production. Since 85% of the pathogens are alien, about $1.8 billion per year in crop losses is due to alien species.

In India, plant pathogens reduce potential crop production by approximately 16%, for a total of $48 billion per year.[48] Approximately 30,000 species of plant pathogens attack Indian crops, including 23,000 species of fungi[49] and 650 species of plant viruses.[50] Approximately 74% of the major plant pathogens in India are considered alien species, based on the major plant pathogens in vegetable crops.[51] The estimated cost of alien plant pathogens to Indian crops amounts to about $35.5 billion per year (Table 17.2).

About 75% of the plant pathogens attacking Brazilian crops are considered alien species.[31] Most of these were introduced with crops. Overall, plant pathogens cause a crop loss of about 13.5% each year.[19] Estimated losses from alien plant pathogens total about $17.1 billion a year (Table 17.2).

17.5 Environmental damage and control costs due to alien species

Many of the approximately 120,000 species of plants, animals, and microbes that have invaded the United States, the United Kingdom, Australia, South Africa, India, and Brazil cause a wide array of environmental and economic damage to both managed and natural ecosystems (Table 17.1). In some cases, an alien species can cause extensive extinctions — the brown tree snake (*Boiga irregularis*), for example, caused the extinction of more than 75% of one group of species on the island of Guam.[4,52]

17.5.1 Plants

Most alien plants now established in the United States were introduced for food, fiber, or ornamental purposes. For example, of the approximately 25,000 alien plant species (mostly ornamentals) that have been brought into Florida for cultivation, more than 900 have escaped and become established in surrounding natural ecosystems.[53–55]

About 5000 alien plants have become established in U.S. natural ecosystems, displacing several native plant species.[56] This is particularly true of the alien weeds that are invading approximately 700,000 ha per year of the wildlife habitats in the United States.[15]

One of these pest weeds is the European purple loosestrife (*Lythrum salicaria*), which was introduced in the early 19th century as an ornamental plant.[57] It has been spreading at a rate of 115,000 ha per year and is changing the basic structure of most of the wetlands it has invaded.[58] Some $45 million is spent on control of purple loosestrife each year[59] (Table 17.3).

The presence of alien aquatic plants, such as hydrilla (*Hydrilla verticillata*), water hyacinth (*Eichhornia crassipes*; native to South America), and water lettuce (*Pistia stratiotes*), alter the habitats of fish and other aquatic species, choke waterways, alter nutrient cycles, and reduce the recreational value of rivers and lakes. Florida spends about $14.5 million each year to control hydrilla.[60] Despite this large expenditure, hydrilla infestations in two Florida lakes have prevented recreational use, causing an annual loss of $10 million.[60] In the United States, a total of $100 million is invested annually in the control of alien aquatic weed species[61] (Table 17.3).

Water hyacinth is also a major weed in South Africa, where it is reducing already scarce water resources.[62] More than $25 million per year is spent on controlling water hyacinth, and water lettuce is causing damage valued at $15 million per year.[63] In Cape Town, invading woody species are estimated to reduce the total water supply by 30%.[64] The economic investment of the program Working for Water totals $50 million a year.[65] In addition, more than $5 million a year is being spent to prevent future alien plant invasions in South Africa.[66]

Of the 27,515 plant species identified in the United Kingdom, only 1515 are considered to be native[67] (Table 17.1). More than 80% of the alien plant species in the United Kingdom are established in disturbed habitats.[16,67]

Many of the alien plant species introduced into Australia have become weeds and have invaded a wide range of environments. These invasive plants are reducing yields in crops and pastures and are changing the natural environment.[68]

Of the 55,000 known plant species in Brazil,[69] an estimated 21.1% (11,605) are alien species.[70] Introduced grass species are having significant negative impacts upon Brazil's ecosystems, because they displace native grasses and make the ecosystem more susceptible to fires than native grasses do.[71]

17.5.2 Mammals

The proportion of alien mammals that have been introduced in the six nations studied range from 6% in the United States to 31% in the United Kingdom (Table 17.1). Domestic mammal introductions include dogs (*Canis familiaris*), cats, cattle, horses (*Equus caballus*), sheep, pigs, and others. Other species intentionally or accidentally introduced include the house mouse (Mus musculus), the European rabbit, the brown rat (*Rattus norvegicus*), and the black rat (*Rattus rattus*).[52,72]

Feral pigs, native to Eurasia and North Africa, are a serious problem in many parts of the world, including the United States and Australia. The number of alien feral pigs in the United States is estimated to be 4 million[4]; in Australia, pigs range from 4 million to 20 million.[34] Feral pigs cause soil erosion; they damage agricultural crops, fences, native plants, and animals; and they pose a threat to livestock and humans. They spread various animal diseases, including tuberculosis, brucellosis, rabies, and foot-and-mouth disease.[72,73] Feral pigs cause an estimated $800.5 million per year in damages in the United States[4] and at least $80 million in Australia[34] (Table 17.3).

Many other small mammals have been introduced into most, if not all, nations in the world. These species include a number of rodents, such as the European black rat, the brown or Asiatic rat, the house mouse, the European rabbit, and the domestic cat and dog.

Some introduced rodents have become serious pests on farms, in industries, and in homes.[74] On farms, rats and mice are particularly abundant and destructive. The United States has an estimated 1.25 billion rats,[4] while India harbors approximately 2.5 billion.[75] In the United States, the best estimate suggests that an individual adult rat causes $15 of damage per year[4]; in India, the estimate is that each rat causes at least $10 per year in damage. In sum, rats cause $19 billion per year in damage in the United States, and about $25 billion per year in India (Table 17.3). Losses from rats based on the damage they cause in other nations are estimated in Table 17.3. Although no economic data are available, in India rats bite about 20,000 people per year, resulting in admittances to hospitals.[76] Also, rats are major vectors for and carriers of more than 38 human and livestock diseases in India.[77]

There are an estimated 63 million pet cats in the United States[78] and as many as 30 million feral cats.[79] Cats prey on native birds and on small native mammals, amphibians, and reptiles.[80] Assuming that eight birds are killed per feral cat each year,[81] some 240 million U.S. birds are killed per year by feral cats.[4] Each adult bird is valued at $30. This cost of a bird is based on the literature that reports that a bird-watcher spends $0.40 per bird observed, a hunter spends $216 per bird shot, and specialists spend $800 per bird reared for release. In addition, the EPA values each small, immature fish at $10; certainly an adult bird has a value three times that of a small, immature fish.[37] Pet cats were estimated to kill another 326 million birds.[4] Therefore the total damage to the U.S. bird population by feral and pet cats is approximately $17 billion per year. This cost does not include small mammals, amphibians, and reptiles that are killed by feral and pet cats.[80]

In Australia, feral cats are also a serious problem, killing native bird, mammal, marsupial, and amphibian populations. There are an estimated 3 million pet cats and 18 million feral cats in Australia.[82] The cats are believed to have eliminated 23 native Australian species of animals.[83,84] Assuming that each of the 18 million feral cats kills eight birds per year, and that the minimum value of a bird is $30,[4] then the total impact from cats is $4.3 billion per year (Table 17.2).

Pet cats and feral cats are also a serious problem in South Africa. For example, on Prince Edward Island, feral cats prey on native birds, causing significant problems with the burrowing petrels (*Procellaria* sp.).[63] An estimated $1.3 million was allocated for cat control over a 7-year period on Prince Edward Island alone, where each cat killed approximately 210 birds per year.[63]

Cats and dogs are also a serious problem in most other nations, but reliable data are not available to estimate their impacts.

17.5.3 Birds

Three of the most common bird-pest invaders worldwide are the common pigeon (*Columba livia*), the English sparrow, and the European starling. These three species, plus other invading bird species, cause a total of $2.4 billion per year in damage in the six nations investigated (Tables 17.2 and 17.3).

A total of 97 of the 1000 bird species in the United States are alien.[85] Of the 97 introduced U.S. bird species, only 5% are considered beneficial, while more than half (56%) are pests.[85] One example is the pigeon, which was intentionally introduced into the United States,[86,87] where it now causes extensive damage totaling $1.1 billion per year[4] (Table 17.3).

Of the 542 species of birds in the United Kingdom, 47 are alien.[88] Pigeons are a particularly serious problem, because they foul buildings, statues, cars, and sometimes pedestrians. On farms they consume grains,[89] causing production and other economic losses. They are also responsible for transmission of at least three poultry diseases, including Newcastle disease.[90–92] Pigeon damage in the United Kingdom is estimated to be at least $270 million per year (Table 17.3).

The number of alien bird species invading the other four nations studied are listed in Table 17.1, but there are no economic data available for the other countries.

17.5.4 Amphibians and reptiles

Although amphibians and reptiles introduced into the United States number only about 53, the negative ecological impacts from these few species have been enormous.[93,94] All of these species inhabit states where it seldom freezes; Florida is now host to 30 species, and Hawaii to 12.[93,94]

The brown tree snake was accidentally introduced to snake-free Guam immediately after World War II when military equipment was moved onto the island.[95] Soon the snake population reached densities of 100 per ha, dramatically reducing native bird, mammal, and lizard populations, as well as causing major problems for small farms and pet owners. Of the 13 species of native forest birds originally found on Guam, only three species still exist in the wild.[96] The snake often crawls up utility poles and has caused a total of 1500 power outages on the island. With about 86 outages per year, a conservative estimate of the cost of the power outages alone is $1 million a year.

In addition, the brown tree snake is slightly venomous, and it causes public health problems, especially when it bites children. At one Guam hospital, bitten infants required hospitalization and intensive care at a total cost of $25,000 per year.[97]

Then there's the cost of endangered species recovery efforts, environmental planning related to snake containment on Guam, and other programs directly stemming from the snake's invasion, which together exceed $1 million per year. In addition, up to $2 million per year is invested in research and control of this serious pest.[97] Hawaii's concern about the snake has prompted the federal government to invest $1.6 million per year in brown tree snake control.[98] Thus, the total cost for the brown tree snake is more than $5.6 million per year (Table 17.3).

There are about 700 species of reptiles and amphibians in Australia, although only two have been introduced.[99] One of these introduced species, the cane toad (*Bufo marinus*), from South America, was introduced as a biological control agent for insect pests in sugarcane.[99] However, the toad has become a pest itself, because it is poisonous to dogs, cats, and other mammals that attack it.[100]

In South Africa, alien species account for 13 of the 299 total reptile species and 11 of the 95 amphibian species.[101] One introduced turtle species is the red-eared slider (*Chrysemys scripta-elegans*) from North America. This turtle has become a major threat to the 12 indigenous terrapin species.[102]

17.5.5 Fish

A total of 138 non-indigenous fish species have been introduced into the United States.[103–105] Most of these introduced fish have been established in states with mild climates, such as Florida (50 species)[105] and California (56 species).[106] In Hawaii, 33 non-indigenous freshwater fish species have become established.[107] Forty-four native U.S. species of fish are threatened or endangered because of non-indigenous fish.[108] An additional 27 species of native U.S. fish are negatively affected by introductions.[108]

Although some native fish species are merely reduced in numbers by non-indigenous species, others are forced into extinction or become hybridized with the invaders. Some alien fish have provided benefits by improving the sport fishing industry. However, other alien fish species have hurt this industry. Sport fishing contributes $69 billion a year to the economy of the United States.[2,109] Based on that estimate, the economic losses from alien fishes is approximately $1 billion annually[2,4] (Table 17.3).

Most of the alien fish species in South Africa are regarded as pests.[110] In addition, seven alien parasitic diseases of fish have been introduced into native fish populations along with the alien fish species.[111] Although the introduction of some sport fish, such as rainbow trout (*Oncorhynchus mykiss*) and largemouth bass (*Micropterus salmoides*), both native to North America, may be somewhat beneficial to the sport-fishing industry, they are also known to have negative impacts on native fish. Introduced species such as carp (*Cyprimus carpio*), bass, and trout threaten about 60% of the endemic freshwater fishes in South Africa.[112] In total, alien fish are responsible for the reduction or local extinction of at least 11 species of fish in South Africa.[110]

17.5.6 Arthropods

Approximately 4600 arthropod species (2582 species in Hawaii and approximately 2000 in the continental United States) have been introduced into the United States. More than 95% of these introductions were accidental, when species entered with plants or in soil and water ballast from ships.

The introduced balsam woolly adelgid (*Adelges piceae*) inflicts severe damage in native balsam–fir forest ecosystems.[111] According to Alsop and Laughlin,[112] this aphid is destroying the old-growth spruce–fir forest in many regions. Over a 20-year period, *A. piceae* has spread throughout the southern Appalachians and has destroyed up to 95% of the Fraser firs (*Abies fraseri*).[113] Alsop and Laughlin[112] report the loss of two native bird species and the invasion of three other species as a result of adelgid-mediated forest death.

Other introduced insect species have become pests of U.S. livestock and wildlife. For example, the red imported fire ant (*Solenopsis invicta*) from South America kills poultry chicks, lizards, snakes, and ground-nesting birds.[114] A 34% decrease in swallow (*Hirundinidae* spp.) nesting success, as well as a decline in the northern bobwhite quail (*Colinus virginianus*) populations, was caused by these ants.[115] The estimated damage to livestock, wildlife, and public health caused by fire ants in Texas is about $300 million a year. An additional $200 million is invested in control per year.[116,117] Assuming equal damages in several other ant-infested southern states, fire ant damage totals approximately $1 billion per year in the United States (Table 17.3).

In addition, the Formosan termite (*Coptotermes formosanus*) is reported to cause approximately $1 billion per year worth of damage in the southern United States, especially in the New Orleans region.[118]

The European green crab (*Carcinus maenas*) has been associated with the demise of the soft-shell clam (*Mya arenaria*) industry in New England and Nova Scotia.[119] It also destroys commercial shellfish beds and preys on large numbers of native oysters and crabs.[119] The annual estimated economic impact of the green crab is $44 million a year.[119] The crab has also invaded ecosystems in Australia and South Africa.[120]

An estimated 80,000 species of insects, 6000 species of spiders, and numerous other arthropod species exist in South Africa.[121,122] One of the most serious invaders is the

Argentine ant (*Linepithema humile*), which is causing major problems by destroying native vegetation, including endangered plants.[111] The same ant is also negatively affecting native ants and other beneficial species of arthropods.[123]

17.5.7 Mollusks

Eighty-eight species of mollusks have been introduced and become established in U.S. aquatic ecosystems.[61] Two of the most serious pests are the zebra mussel (*Dreissena polymorpha*) and the Asian clam (*Corbicula fluminea*).

The European zebra mussel was first found in Detroit's Lake St. Clair, having gained entrance via ballast water released in the Great Lakes by ships that traveled from Europe.[124] The zebra mussel has spread into most of the aquatic ecosystems in the eastern United States and is expected to invade most freshwater habitats throughout the nation.[124] Large mussel populations reduce food and oxygen for native fauna; mussel densities as high as 700,000 per m^2 have been recorded.[125] In addition, zebra mussels have been observed completely covering native mussels, clams, and snails, thereby further threatening the survival of native species.[124,125] Zebra mussels also invade and clog water intake pipes at water filtration and electric generating plants. It was estimated that this species would cause $5 billion a year in damage and associated control costs by the year 2000.[126]

Though the Asian clam grows and disperses less rapidly than the zebra mussel, it also causes significant fouling problems and threatens native species. Costs associated with the damage it causes are about $1 billion a year[61,127] (Table 17.3).

The introduced shipworm (*Teredo navalis*) in San Francisco Bay has been causing serious damage to docks and ships since the early 1990s. Currently, damage is estimated to be about $205 million a year[128] (Table 17.3).

17.6 Livestock pests

Microbes and other parasites were introduced when various species of livestock were brought into the six nations under consideration. In addition to the hundreds of pest microbes and parasites that have already been introduced, there are more than 60 additional microbes and parasites that could easily invade the United States and become serious pests to U.S. livestock.[129] A conservative estimate of the current losses to U.S. livestock from alien microbes and parasites is approximately $3 billion per year[130] (Table 17.3).

In Australia, there are an estimated 44 alien diseases of animals that could infect livestock if they gained entrance.[132] Already, three alien insect and mite species are causing $228 million a year in damage to the wool industry alone[133] (Table 17.3).

In India, there are about 50 alien diseases of livestock and wildlife that are causing significant losses, including foot-and-mouth disease.[134] During 8 months in 1996, nearly 50,000 cases of foot-and-mouth disease were reported,[135] with treatment costs of about $17,000.[48]

Several serious alien livestock diseases in South Africa, including tuberculosis, brucellosis East Coast fever, anthrax, and rinderpest are infecting livestock, wildlife, and other animals.[136] Estimates are that brucellosis alone is responsible for losses amounting to $100 million a year[137] (Table 17.3).

In Brazil and other Latin American countries, imported bovine tuberculosis (bovine TB) has become a serious threat to the development of the beef and dairy industries. Losses to bovine TB are reported to be approximately $63 million a year.[137]

17.7 Human diseases

17.7.1 AIDS, influenza, and syphilis

Perhaps the most notorious of all alien human diseases is acquired immune deficiency syndrome (AIDS), which originated in central Africa. Since the early 1980s the disease has spread to all inhabited parts of the globe. The number of cases of AIDS and HIV (human immunodeficiency virus) infections and treatment costs in the United States, the United Kingdom, Australia, South Africa, India, and Brazil are shown in Table 17.4.

New influenza strains, originating in the Far East, quickly spread to the United States and other nations. The influenza strains are reported to cause 5 to 6% of all deaths in 121 U.S. cities.[138] Costs of hospitalizations for a single outbreak of an influenza such as type A can exceed $300 million per year.[139] In total, AIDS and influenza take the lives of more than 40,000 people each year in the United States, and treatment costs for these diseases, plus for syphilis, total approximately $6.5 billion each year, and that does not include the cost of any other alien diseases (Table 17.3).

New influenza strains in the United Kingdom are reported to cause from 3000 to 4000 deaths a year.[140] In total, AIDS and influenza take the lives of approximately 4000 people each year in the United Kingdom, and treatment costs total approximately $1 billion a year, based on extrapolated data from the United States[4] (Table 17.3).

New influenza strains in Australia are reported to cause about 210 deaths a year.[140] In India, influenza cases totaled 3 million in 1984.[140]

17.7.2 Other diseases

Several other non-indigenous diseases infect humans in Brazil, including malaria, cholera, yellow fever, and dengue fever.[141,142] The numbers of people infected include: cholera, 2167; malaria, 425,000[142]; and dengue, 96,100.[143] If we assume that all of these infected people are hospitalized and that the average cost per year per person for hospitalization from these diseases in Brazil is $213,[143] then a minimum cost for these non-indigenous diseases is $133 million a year. In addition, the Brazilian government plans on spending about $1.4 billion annually to eradicate the *Aedes aegypti* mosquito, the vector of dengue fever and yellow fever[142] (Table 17.3).

Human disease transfers from one region to another continue to increase because of population growth, high population densities, rapid transportation, and the encroachment of civilization into new ecosystems.

17.8 Control and future implications

The economic damages associated with non-indigenous species invasions in the nations in the six continents total more than $336 billion per year. Of this, control costs account for more than $30 billion. Most of the control costs are associated with agricultural production.[35]

Most attempts at eradication have failed once the invading species has become well established. In the United States, the only pest species that has been eradicated has been the Mediterranean fruit fly (*Ceratitis capitata*) in Florida. The fly has invaded Florida at least three times and has been eradicated each time as a result of concerted efforts by the U.S. Department of Agriculture and the Florida Department of Agriculture.

In Britain, efforts are being made to eradicate the introduced muskrat (*Ondatra zibethicus*) and coypu (*Myocaster coypus*).[144] The populations have been significantly reduced, but the eradication effort has yet to succeed.[145]

The number of invading species worldwide have been increasing rapidly. For example, in the San Francisco Bay and Delta region, there has been a tenfold increase in the number of invading species since 1900.[146] With more people traveling and more goods moving from one country to another, there is greater opportunity for increasing numbers of species to invade most countries. There is a critical need for strict legislation to help prevent non-native species invasions, and for a major effort to educate the public concerning the dangers of invading species.

17.9 Conclusion

More than 120,000 non-indigenous species of plants, animals, and microbes have invaded the United States, the United Kingdom, Australia, India, South Africa, and Brazil (Table 17.1). An estimated 20 to 30% of the introduced species are pests and cause major environmental problems. Although relatively few of these species become serious pests, some species inflict significant damage to natural and managed ecosystems and cause public health problems. A variety of ecological factors allow alien species to become abundant and to emerge as ecological threats in their new ecosystem. These include alien plant or animal species introduced without their natural enemies (e.g., purple loosestrife); the development of new associations between alien parasite and host (e.g., the HIV virus and humans); effective predators in a new ecosystem (e.g., feral cats); artificial or disturbed habitats that provide favorable ecosystems for the invasive aliens (e.g., weeds in crop and lawn habitats); and invasion by some highly adaptable and successful alien species (e.g., the water hyacinth and the zebra mussel).

The study documents that the economic damage associated with non-indigenous species invasions in the six selected nations totals more than $336 billion per year. Precise economic costs associated with some of the most ecologically damaging alien species are not available. Cats and pigs, for example, have been responsible for the extinction of various animal species. Yet for these pest animals, only minimal cost data are known. Also, it is impossible to assess the value of a species that has been forced to extinction. If monetary values could be assigned to species extinctions, losses in biodiversity, ecosystem services, and aesthetics, then the cost of destructive non-indigenous species would undoubtedly be several times higher than the reported $336 billion. Yet even this conservatively stated economic loss indicates that alien species are exacting a significant environmental and economic toll worldwide.

The calculated dollar cost per capita for the losses incurred due to biological invaders in the six nations investigated were approximately $240 per year. Assuming similar costs worldwide, damages from invasive species would total more than $1.4 trillion a year. Based on an estimated $31 trillion in world GNP,[5] the $1.4 trillion in losses from invasive species represents nearly 5% of the world economy.

Nearly all crop and livestock species are non-indigenous. These alien crops (e.g., corn and wheat) and livestock (e.g., cattle and poultry) are vital to maintaining world agriculture and the global food system. However, these benefits do not diminish the enormous negative impacts of other non-indigenous species on agricultural and other managed and natural ecosystems.

A real challenge lies ahead in preventing further damage from invading alien species to natural and managed ecosystems worldwide, especially given the current rapid human population growth and related activities. The United States has taken a few steps in an effort to protect the environment from biological invaders. For example, President Clinton issued an executive order on Feb. 2, 1999, creating an interagency National Invasive Species Council and allocating $28 million to produce a plan within 18 months to mobilize the federal government to defend against non-indigenous species invasions. In addition, Australia, South Africa, India, Brazil, and the United Kingdom all have specific programs in place to prevent invasions of alien species in their countries. This suggests that a few million dollars spent on preventing future introduction of potentially harmful alien species in the United States and other nations will prevent billions of dollars in losses to agriculture, forestry, and other aspects of managed and natural environments worldwide.

Specific legislation is needed in all countries to slow or prevent non-native species introductions. All introductions of non-native plants, animals, and microbes, for whatever purposes — including agriculture, hunting, tourism, pets, recreation, and research — should be strictly regulated.[147] In addition, the government should make every effort to inform the public concerning the serious environmental and economic threats that are associated with alien species introductions.

Acknowledgments

We thank the following people for reading an earlier draft of this chapter and for their many helpful suggestions: V.L. Chopra, Indian Agricultural Research Institute, New Delhi, India; M.N. Clout, The University of Auckland, Auckland, New Zealand; M.L. Corn, Division of Resources, Science, and Industry, Congressional Research Service, Washington, DC; F. Fenner, John Curtin School of Medical Research, Canberra, Australia; M.D. Fox, University of New South Wales, Sydney, Australia; W. Ghiorse, Cornell University, Ithaca, NY; A. Hilsdorf, University of Taubate, Brazil; D.W. Macdonald, University of Oxford, Oxford, U.K., H. Momen, Instituto Oswaldo Cruz, Rio de Janeiro, Brazil; H. Peskin, Edgevale Associates, Nellysford, VA; D. Richardson, University of Cape Town, Cape Town, South Africa; M.J. Samways, University Natal, Pietermaritzburg, South Africa; D.N. Sen, University of Jodhpur, Jodhpur, India; I.K. Sharma, World Bank Global Environmental Facility Earth Council, Punjab, India; A. Vaidyanathan, Indian Society of Agricultural Economics, Mumbai, India; B. van Wilgen, CSIR Division of Water, Environment, and Forestry Technology, Stellenbosch, South Africa; R.L. Welcomme, Imperial College of Science and Technology, London, U.K.; M. Wingfield, University of Pretoria, Pretoria, South Africa.

References

1. Raven, P.H. and Johnson, G.B. *Biology*, 3rd ed., Mosby Year Book, St. Louis, 1992.
2. USBC. *Statistical Abstract of the United States*, U.S. Bureau of the Census, U.S. Government Printing Office, Washington, 1998.
3. Mack, M.C. and D'Antonio, C.M. Impacts of biological invasions on disturbance regimes, Trends Ecol. Evol., 13, 1998, 195.
4. Pimentel, D., et al. Environmental and economic costs associated with non-indigenous species in the United States, *BioScience,* 50, 2000, 53.
5. PRB. World Population Data Sheet, Population Reference Bureau, Washington, D.C., 1998.
6. Bryan, R.T. Alien species and emerging infectious diseases: past lessons and future applications, in *Proceedings of the Norway/UN Conference on Alien Species*, *July 1–5*, Sandlund, G.T. and Viken, P.J. (Eds.)., Norwegian Institute for Nature Research, Trondheim, Norway, 1996, 74.
7. Nature Conservancy. *America's Least Wanted: Alien Species Invasions of U.S. Ecosystems*, The Nature Conservancy, Arlington, Va., 1996.
8. Armstrong, S. Rare plants protect Cape's water supplies, *New Scient*., February 11, 1995, 8.
9. Elton, C.S. *The Ecology of Invasions by Animals and Plants*, Methuen, London, 1958.
10. Bright, C. Invasive Species: Pathogens of Globalization, *For. Pol.,* Fall, 1999, 51.

11. Pimentel, D., Hunter, M.S., LaGro, J.A., Efronymson, R.A., Landers, J.C., Mervis, F.T., McCarthy, C.A., and Boyd, A.E. Benefits and risks of genetic engineering in agriculture, *BioScience*, 39, 1989, 606.
12. Pimentel, D. Habitat factors in new pest invasions, in *Evolution of Insect Pests — Patterns of Variation*, Kim, K.C. and McPheron, B.A. (Eds.), John Wiley & Sons, New York, 1993, 165.
13. USDA. *Agricultural Statistics,* U.S. Department of Agriculture, Washington, D.C., 1998.
14. Pimentel, D. *Handbook on Pest Management in Agriculture,* Vols, 1–3, Pimentel, D. (Ed.), CRC Press, Boca Raton, FL.
15. Babbit, B. Statement by Secretary of the Interior on invasive alien species, *Proceedings, National Weed Symposium,* BLM Weed Page, April 8–10, 1998.
16. Clement, E.J. and Foster, M.C. *Alien Plants of the British Isles,* Botanical Society of the British Isles, London, 1994.
17. Crawley, M.J., Harvey, P.H., and Purvis, A. Comparative Ecology of the Native and alien floras of the British Isles, *Philo. Trans. Royal Soc. London Biol. Sci.*, 351, 1996, 1251.
18. Spedding, C.R.W. Weeds, pests and diseases in grassland and herbage legumes — an overview, in *Weeds, Pests and Diseases in Grassland and Herbage Legumes*, Brockman, J.S. (Ed.), British Crop Protection Council Publications, Croydon, UK, 1985, 1.
19. Oerke, E.C., et al. Crop Production and *Crop Protection: Estimated Losses in Major Food and Cash Crops,* Elsevier, Amsterdam, 1994.
20. Lemerle, D., et al. Survey of weeds and diseases in cereal crops in the southern wheat belt of New South Wales, *Austral. Jour. Exp. Agric.,* 36, 1996, 545.
21. Davis, S.D., Heywood, U.H., and Hamilton, A.C. *Centres of Plant Diversity: A Guide and Strategy for their Conservation,* Vol. 2, Asia, Australia & Pacific, IUCN, Geneva, 1995.
22. Londsdale, W.M. Inviting trouble: introduce pasture species in northern Australia, *Austral. Jour. Ecol.* 19, 1994, 345.
23. Delfosse, E.S. *Pests of Pastures: Weed, Invertebrates and Disease Pests of Australian Sheep Pastures*, CSIRO, East Melboune, Australia, 1993.
24. Groves, R.H. A short history of biological invasions of Australia, in *Biography of Mediterranean Invasions*, Groves, R.H. and DiCastri, F. (Eds.), Cambridge University Press, New York, 1991, 59.
25. Bromilow, C. *Problem Plants of South Africa*, Briza Publications, Arcadia South Africa, 1995.
26. Cilliers, C.J. and Nesser, S. Biological control of *Lantana camara* (Vrobenaceae) in South Africa, *Agric. Ecosys. Environ.*, 37, 1991, 57.
27. Kluge, R.L. and Nesser, S. Biological control of *Hakea sericea* (Proteaceae) in South Africa, *Agric. Ecosys. Environ.*, 37, 1991, 91.
28. Singh, C.M., Angras, N.M., and Kumar, S. *Weed Management*, MD Publications, PVT, LDT, New Delhi, 1996.
29. Khuspe, V.S., Subbaiah, R., and Mande, J.V. A *Compendium of Indian Weed Science Research 1950–1981,* Metropolitan Book Co., New Delhi, 1982.
30. Nandpuri, K.S., et al. *Field Problems of Important Crops,* Punjab Agricultural University, Ludhiana, India, 1986.
31. Echandi, E., et al. *Crop Protection in Brazil*, *Uruguay, Bolivia, Ecuador, and Dominican Republic,* United States Agency for International Development, Washington, D.C., 1972.
32. McKillop, I.G., et al. Long-term cost effectiveness of fences to manage European wild rabbits, *Crop Prot.*, 17, 1998, 393.
33. Thompson, H.V. The rabbit in Britain, in *The European Rabbit: The History and Biology of a Successful Colonizer*, Oxford University Press, Oxford, 1994, 64.
34. Emmerson, G. and McCulloch, J. Feral Peril: Queensland's Introduced Plants and Animals, Queensland Parliamentary Library, Brisbane, Australia, 1994.
35. Pimentel, D. *Techniques for Reducing Pesticides: Environmental and Economic Benefits*, John Wiley & Sons, Chichester, UK, 1997.
36. Liebold, A.M., et al. Invasion by exotic forest pests: a threat to forest ecosystems, *For. Sci.*, 41, 1995, 1.
37. Hall, J.P. and Moody, B. Forest depletions caused by insects and diseases in Canada 1982–1987, *Forest Insect and Disease Survey Information Report ST-X-8,* Forest Insect and Disease Survey, Canadian Forest Service, Natural Resources Canada, Ottawa, 1994.

38. Seymour, P.R. *Invertebrates of Economic Importance in Britain: Common and Scientific Names*, H.M.S.O., London, 1989.
39. van Lenteren, J.C. Frequency and consequences of insect invasions, in *Plant and Microbial Biotechnology Research Series,* Hokkanen, H.M.T. and Lynch, J.M. (Eds.), Cambridge University Press, Cambridge, 1995, 30.
40. Woods, W., Michael, P., and Grimm, M. *Insect and Allied Pests of Extensive Farming*, The Plant Protection Society of Western Australia, Victoria Park, Australia, 1990.
41. Myburgh, A.C. *Crop Pests in Southern Africa*, Plant Protection Research Institute, Department of Agriculture and Water Supply, Pretoria, South Africa, 1989.
42. David, K.V. and Kumaraswami, T. *Elements of Economic Entomology*, Madras, India, Popular Book Depot, 1975.
43. Lal, O.P. A survey of insect and non-insect pests parasitoids and predators from Kullu Valley, Himachal Pradesh, India, *J. Entomol. Res. (New Delhi)*, 14, 1990, 99.
44. Carlile, W.R. *Control of Crop Diseases,* Edward Arnold, London, 1988.
45. Persley, D.M. and Syme, J.R. *Field Crops and Pastures: A Disease Management Guide*, Queensland Department od Primary Industries, Brisbane, Australia, 1990.
46. Wager, V.A. *Common Diseases of Vegetables in South Africa,* Department of Agriculture, Pretoria, South Africa, 1956.
47. Nel, P.J. *Deciduous Fruits and Vines: Pests and Diseases and Their Control*, David Philip, Cape Town, South Africa, 1985.
48. Singh, S.P. Biological control, in *50 Years of Crop Science Research in India*, Paroda, R.S. and Chadha, K.L. (Eds.), Indian Council of Agricultural Research, New Delhi, 1996, 88.
49. Pandotra, V.R. *Illustrated Fungi of North India with Special Reference to J & K State*, International Book Distributors, Dhra Don, India, 1997.
50. Patel, B.N. and Patel, G.J. Virus diseases of tobacco and their management, in *Integrated Pest Diseases Management: Proceedings of the National Seminar,* Jayaraj, S. (Ed.), Tamil Nadu Agricultural University, Coimbatore, India, 1985, 261.
51. Singh, R.S. *Diseases of Vegetable Crops,* Oxford & IBH Publishers, New Delhi, India, 1985.
52. Atkinson, I.A.E. Introductions of wildlife as a cause of species extinctions, *Wild. Biol.*, 2, 1996, 135.
53. Frank, J.H. and McCoy, E.D. Introduction to insect behavioral ecology: the good, the bad and the beautiful: non-indigenous species in Florida, *Flor. Entomol.,* 78, 1995, 1.
54. Frank, J.H., et al. Immigration and introduction of insects, in *Strangers in Paradise,* Simberloff, D., Schmitz, D.C., Brown, T.C. (Eds.), 1997, 75.
55. Simberloff, D., Schmitz, D.C., and Brown, T.C. *Strangers in Paradise,* Island Press, Washington, D.C., 1997.
56. Morse, L.E., Cartes, J.T., and Center, L. Native vascular plants, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 205.
57. Malecki, R.A., et al. Biological control of purple loosestrife, *BioScience,* 43, 1993, 680.
58. Thompson, D.G., Stuckey, R.L., and Thompson, E.B. Spread, impact, and control of purple loosestrife *Lythrum salicaria* in North America wetland, U.S. Fish and Wildlife Service, Fish and Wildlife Research 2, Washington, D.C., 1987.
59. ATTRA. Purple Loosestrife: Public Enemy #1 on Federal Lands, ATTRA Interior Helper Internet, Washington, D.C., 1997, http://refuges.fws.gov/NWRSFiles/Habitat/Mgmt/PestMgmt/LoosestrifeProblem.html
60. Center, T.D., Frank, J.H. and Dray, F.A. Biological control, in *Strangers in Paradise,* Simberloff, D., Schmitz, D.C., and Brown, T.C. (Eds.), Island Press, Washington, D.C., 1997, 245.
61. OTA. *Harmful Non-Indigenous Species in the United States*, Office of Technology Assessment, United States Congress, Washington, D.C., 1993.
62. Richardson, D.M., et al. Alien plant invasions, in *Vegetation of Southern Africa*, Cowling, R.M., Richardson, D.M., and Pierce, S.M. (Eds.), Cambridge University Press, 1997, 535.

63. Huntley, B.J. South African experience regarding alien species: impacts and controls, *Proceedings of Norway/UN Conference on Alien Species,* Sandlund, O.T., Schei, P.J., and Viken, A. (Eds.), The Norwegian Ministry of Environment, Trondheim, 1996, 182.
64. Le Maitre, D.C., et al. Invasive plants and water resources in the Western Cape Province, South Africa: modeling the consequence of a lack of management, *J. Appl. Ecol.,* 33, 1996, 161.
65. van Wilgen, B.W. and van Wyk, E. Invading alien plants in South Africa: impacts and solution, *J. Arid Envir.,* 1999.
66. Alien Plants. http://www.wildnetafrica.co.za/bushcraft/dailynews/1997archive_4/archive_1997//28_alien plants.html.
67. Crawley, M.J., Harvey, P.H., and Purvis, A. Comparative Ecology of the Native and alien floras of the British Isles, *Philo. Trans. Royal Soc. London Biol. Sci.,* 351, 1996, 1251.
68. Castles, I. *Australia's Environment: Issues and Facts,* Australian Bureau of Statistics, Commonwealth of Australia, Sydney, 1992.
69. WRI. *World Resources 1998–1999*. World Resources Institute, Washington, D.C., 1998.
70. Vitousek, P.M., et al. Introduced species: a significant component of human-caused global change, *New Zealand J. Ecol.,* 21, 1997, 1.
71. Nepstead, D.C. and Serrao, C. Recuperation of a degraded Amazonian landscape: forest recovery and agricultural restoration, *Ambio,* 21, 1991, 248.
72. Lever, C. *Naturalized Animals: The Ecology of a Successfully Introduced Species*, T & AD Poser Natural History, London, 1994.
73. Hume, I.D. Native and introduced herbivores in Australia, in *The Nutrition of Herbivores,* Hacker, J.B., Ternouth, J.H. (Eds.), Academic Press, Sydney, 1987, 1.
74. Layne, J.N. Nonindigenous mammals, *Strangers in Paradise,* Simberloff, D., Schmitz, D.C. and Brown, T.C. (Eds.), Island Press, Washington, D.C., 1997, 157.
75. Vasantharaj, D.B. and Kumaraswami, T. *Elements of Economic Entomology*, Popular Book Depot, Madras, India, 1975.
76. Mountfort, G. and Cubitt, G. *Wild India*, William Collins & Co. LTD, London, 1985.
77. Srivastava, A.S. Survey, ecology and integrated control of rodents, in *Recent Advances in Entomology,* Mathur, Y., et al. (Eds.), New Gopal Printing Press, Parade, Kanpur, India, 1987, 454.
78. Nassar, R. and Mosier, J. Projections of pet population from census demographic data, *J. Amer. Vet. Med. Assoc.,* 198, 1991, 1157.
79. Luoma, J.R. *Catfight*, Audubon, 99, 1997, 85.
80. Dunn, E.H. and Tessaglia, D.L. Predation of birds at feeders in winter, *J. Field Ornith.,* 65, 1994, 8.
81. McKay, G.M. Feral cats in Australia: origin and impacts, in *Unwanted Aliens? Australia's Introduced Animals,* Nature Conservation Council of NSW, The Rocks, NSW, Australia, 1996.
82. Anon., Menace of moggies, *The Press*, The Christchurch Press Co. Limited, Christchurch, Australia, 1996, 21.
83. Maynard, R. and Hawkes, N. Australians call for cats on menu to save wildlife, *The London Times,* Times Newspapers Limited, October 17, Overseas News, 1996.
84. Low, T. *Feral Future*, Viking, Australia, 1999.
85. Temple, S.A. Exotic birds, a growing problem with no easy solution, *The Auk*, 109, 1992, 395.
86. Laycock, G. *The Alien Animals,* Natural History Press, New York, 1966.
87. Roots, C. *Animal Invaders*, Universe Books, New York, 1976.
88. Gooders, J. *Collins British Birds*, Collins, London, 1982.
89. Long, J.L. *Introduced Birds of the World: The Worldwide History, Distribution, and Influence of Birds Introduced to New Environments*, Universe Books, New York, 1981.
90. Alexander, D.J., et al. Newcastle disease outbreaks in fowl in Great Britain during 1984. *Vet. Rec.,* 117, 1985, 429.
91. Alexander, D.J. and Parsons, G., Pathogenicity for chickens of avian paramyxovirus type I isolates obtained from pigeons in Great Britain during 1983–1985, *Avian Path*. 15, 487,1986.
92. Bevan, B.J. and Bracewell, C.D. Chlamydiosis in birds in Great Britain: 2 isolations of chlamydiapsittaci from birds sampled between 1976 and 1984, *Jour. Hyg.,* 96,453, 1986.

93. McCoid, M.J. and Kleberg, C. Non-native reptiles and amphibians, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 443.
94. Lafferty, K.D. and Page, C.J. Predation of the endangered tidewater goby, *Eucyclogobius newberry,* by the introduced African Clawed frog, *Xenopus laevis,* with notes on the frog's parasites, *Copeia,* 3, 1997, 589.
95. Fritts, T.H. and Rodda, G.H. Invasions of the brown tree snake, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems,* LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 454.
96. Rodda, G.H., Fritts, T.H., and Chiszar, D., The disappearance of Guam's wildlife, *BioScience,* 47, 1997, 565.
97. Fritts, T. Personal communication, 1998.
98. Holt, A. 1997–1998. Hawaii's reptilian nightmare, *World Conservation,* 4/97–4/98, 31.
99. Fox, M.D. Conserving biodiversity: impact and management of exotic organisms, in *Conserving Biodiversity: Threats and Solutions*, Bradstock, R.A., et al. (Eds.), Surrey Beatty & Sons Pty LDT, Chipping Norton, New South Wales, 1995, 177.
100. Sabath, M.D., Boughton, W.C., and Easteal, S. Expansion of the range of the introduced toad Bufo marinus in Australia 1935–1974, *Copeia,* 3, 1981, 676.
101. Siegfried, W.R. Preservation of species in Southern African nature reserves, in *Biotic Diversity in Southern Africa: Concepts and Conservation*, Huntley, B.J. (Ed.), Oxford University Press, Oxford, 1989.
102. Boycott, R.C. and Bourquin, O. *The South African Tortoise Book: A Guide to South African Tortoises, Terrapins and Turtles*, Southern Book Publishers, Johannesburg, 1988.
103. Courtenay, W.R., Jennings, D.P., and Williams, J.D. Appendix 2. Exotic fishes of the United States and Canada, in *A List of Common and Scientific Names of Fishes from the United States and Canada,* Robins, C.R. (Ed.), 5th edition, Special Publication 20, American Fisheries Society, Bethesda, MD, 1991.
104. Courtenay, W.R. Biological pollution through fish introductions, in *Biological Pollution, The Control and Impact of Invasive Exotic Specie*s, McKnight, B.N. (Ed.), Indiana Academy of Science, Indianapolis, 1993, 35.
105. Courtenay, W.R., Nonindigenous fishes, *Strangers in Paradise,* Simberloff, D., Schmitz, D.C., Brown, T.C. (Eds.), Island Press, Washington, D.C., 1997, 109.
106. Dill, W.A. and Cordone, A.J. History and Status of Introduced Fishes in California, 1871–1996, *Fish Bulletin* 178, State of California, The Resources Agency, Department of Fish and Game, 1997.
107. Maciolek, J.A. Exotic fishes in Hawaii and other islands of Oceania, in *Distribution, Biology, and Management of Exotic Fishes,* Courtenay, W.R. and Stauffer, J.R. (Eds.), Johns Hopkins University Press, Baltimore, 1984, 131.
108. Wilcove, D.S. and Bean, M.J. *The Big Kill: Declining Biodiversity in America's Lakes and Rivers,* Environmental Defense Fund, Washington, D.C., 1994.
109. Bjergo, C., Boydstun, C., Crosby, M., and Kokkanakis, S., Sayers, R. Non-native aquatic species in the United States and coastal water, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems*, LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 428.
110. Bruton, M.N. and Van As, J. Faunal invasions of aquatic ecosystems in Southern Africa, with suggestions for their management, in Proceedings of the National Synthesis Symposium on the Ecology and Management of Biological Invasions in Southern Africa, Macdonald, I.A.W., Kruger, F.J. and Ferrar, A.A. (Eds.), Oxford University Press, Cape Town, 1985, 47.
111. Jenkins, J.C. *Measuring and Modeling Northeastern Forest Response to Environmental Stresses,* Ph.D. dissertation submitted to the University of New Hampshire, Durham, N.H, 1998.
112. Alsop, F.J. and Laughlin T.F. Changes in the spruce-fir avifauna of Mt. Guyot, Tennessee, 1967–1985, *J. Tenn. Acad. Sci.,* 66, 1991, 207.

113. Neufeld, H.S. personal communication, 1998.
114. Vinson, S.B. Impact of the invasion of *Solenopsis invicta* (Buren) on native food webs, in *Exotic Ants: Biology, Impact, and Control of Introduced Species,* Williams, D.F. (Ed.), Westview Press, Boulder, CO, 1994, 241.
115. Allen, C.R., Lutz, R.S., and Demarais, S. Red imported fire ant impacts on northern bobwhite populations, *Ecol. App.*, 5, 1995, 632.
116. Vinson, S.B. The economic impact of the imported fire ant infestation on the State of Texas, *Report,* Texas A&M University, College Station, TX, 1992.
117. TAES, Texas Imported Fire Ant Research & Management Plan, Report, Texas Agricultural Extension Service, Texas A&M University, College Station, TX, 1998.
118. Corn, M.L., et al. *Harmful Non-Native Species: Issues for Congress,* Congressional Research Service, Library of Congress, Washington, D.C., 1999.
119. Lafferty, K.D. and Kuris, A.M. Biological control of marine pests, *Ecology,* 77, 1996, 1989.
120. Lexis-Nexis. http://web/lexis-nexis.com/universe/do...&_md5 = e58d78ae0ab24daccb7bc83, 1999.
121. Anon. *Southern African Wildlife: A Guide to Our Mammals, Birds, Reptiles, Insects, Fishes, Amphibians, Invertebrates,* **Readers Digest**, New York, 1989.
122. South Africa, South Africa Government on Line: http://www.gov.za, 1998.
123. Kruger, F.J., et al. The characteristics of invaded Mediterranean-climate regions, *in Biological Invasions: A Global Perspective,* Drake, J.A., et al. (Eds.), John Wiley & Sons, New York, 1989, 181.
124. Benson, A.J. and Boydstun, C.P. Invasion of the zebra mussel into the United States, in *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems,* LaRoe, E.T., et al. (Eds.), U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995, 445.
125. Griffiths, D.W., et al. Distribution and dispersal of the zebra mussel (*Dreissena polymorpha*) in the Great Lakes Region, *Can.J. Fish. Aqua. Sci.*, 48, 1991, 1381.
126. Khalanski, M. Industrial and ecological consequences of the introduction of new species in continental aquatic ecosystems: the zebra mussel and other invasive species, *Bulletin Francais de la Peche et de la Pisciculture*, 0, 1997, 344
127. Isom, B.G. Rationale for Sampling and Interpretation of Ecological Data in the Assessment of Freshwater Ecosystems. American Society for Testing and Materials, *ASTM* (*American Society for Testing and Materials*) *Special Technical Publication*, 894, Philadelphia, PA, 1986.
128. Cohen, A.N. and Carlton, J.T. *Nonindigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta*, United States Fish and Wildlife Service, Washington, D.C., 1995.
129. USAHA. *Foreign Animal Diseases: Their Prevention, Diagnosis and Control*, Committee on Foreign Animal Diseases of the United States Animal Health Association, Richmond, VA., 1984.
130. Drummond, R.O., et al. Estimated losses of livestock to pests, in *Handbook of Pest Management in Agriculture,* Pimentel, D. (Ed.), CRC Press, Boca Raton, FL, 1981, 111.
131. Morgan, N.O. Potential impact of alien arthropod pests and vectors of animal diseases on the U.S. livestock industry, in *Handbook of Pest Management in Agriculture*, Pimentel, D. (Ed.), CRC Press, Boca Raton, FL, 1981, 129.
132. Meischke, H.R.C. and Geering, W.A. Exotic animal diseases, in *Pests and Parasites as Migrants,* Gibbs, A. and Meischke, R. (Eds.), Cambridge University Press, Cambridge, 1985.
133. Slater, P.D., Lewington, D., and Prathey, J.E. An investigation into the effects of redlegged earth mite and Lucerne flea on the performance of subterranean clover in annual pasture in S. New South Wales, *Pl. Prot. Quar.,* 11, 1996, 6.
134. Khera, S.S. and Sharma, G.L. *Important Exotic Diseases of Livestock Including Poultry,* Indian Council of Agricultural Research, New Delhi, 1967.
135. Foot and Mouth Disease. http://www.ss.niah.affrc.go.jp, 1998.
136. Coetzer, J.A.W., Thomson, G.R., and Tustin, R.C. *Infectious Disease of Livestock: With Special Reference to Southern Africa,* Oxford University Press, Cape Town, 1994.
137. Cosivi, O., et al. Zoonotic tuberculosis due to *Mycobacterium bovis* in developing countries. *Emerg. Infec. Dis.,* 4, 1998, 1.

138. Kent, J.H., et al. Influenza surveillance — United States, 1991–92. *Mort. Morb. Weekly Rep. Surv. Summ*. 41, 1992, 35.
139. Chapman, L.E., et al. Influenza — United States, 1989–90 and 1990–91 seasons, *Mort. Morb. Weekly Rep. Surv. Summ*., 41, 1992, 35.
140. Influenza. www.influenzanews.com, January 31, 1999.
141. Alves, M.C.G.P., Gurgel, S.D.M., and Almeida, M.D.C.R.R.D. Sampling design for larval density computation of *Aedes aegypti* and *Aedes albopictis* in the state of S. Paulo Brazil, *Revista de Saude Publica,* 25, 1991, 251.
142. PAHO. *Health in the Americas, Scientific Pub. No. 569, Vol. 1 & 2*, World Health Organization, Pan American Health Organization, Washington, D.C., 1998.
143. Nascimento, E.L. Sick and tired. http://www.brazil-brasil.com/cvrap96.htm February 6, 1998.
144. Gosling, L.M. and S.J. Baker. The eradication of muskrats and coypus from Britain, U.K. *Biol.J. Linn. Soc.,* 38, 1989, 39.
145. Keymer, I.F., Wells, G.A.H., and Ainsworth, H.L. Renal neoplasia in coypus (*Myocaster coypus*). *Vet. Jour.*, 158, 1999, 144.
146. Cohen, A.N. and Carlton, J.T. *Nonindigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta*, United States Fish and Wildlife Service, Washington, D.C., 1995.
147. Sjoberg, G. and Hokkanen, H.M.T. Conclusions and recommendations of the OECD workshop on the ecology of introduced, exotic wildlife: fundamental and economic aspects, *Wild. Biol.,* 2, 1996, 131.
148. Palmer, E.L., Fowler, H.S. *Fieldbook of Natural History*, McGraw-Hill, New York, 1975.
149. McKnight, K.H. and McKnight, V.B. *Peterson's Field Guide to Mushrooms*, Houghton Mifflin, New York, 1987.
150. Horst, R.K. *Westcott's Plant Disease Handbook*, Avi Book, Van Nostrand Reinhold, New York, 1990.
151. Fauquet, C.M. Taxonomy and classification general, in *Encyclopedia of Virology*, Webster, R.G. and Granoff, A. (Eds.), Academic Press, London, 1994, 1396.
152. LaRoe, E.T., et al. (Eds.), *Our Living Resources: A Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems*, U.S. Department of the Interior, National Biological Service, Washington, D.C., 1995
153. Hawksworth, D.L., et al. *Dictionary of the Fungi*, 8th ed., CAB International, Cambridge, U.K, 1995.
154. Matthews, L.H. *Mammals in the British Isles,* Collins, London, 1982.
155. Baker, S.J. Escaped exotic mammals in Britain, *Mammal Rev.* 20, 1990, 75.
156. Frazer, J.F.D. *Amphibians*, Wykeham Science Series, Vol. 25, 1973.
157. Cacutt, L. *British Freshwater Fishes*, Croom Helm, London, 1979.
158. Maitland, P.S. and Campbell, R.N. *Freshwater Fishes of the British Isles*, Harper Collins, London, 1992.
159. Kitching, R.L. *The Ecology of Exotic Animals and Plants: Some Australian Case Histories,* John Wiley & Sons, New York, 1986.
160. Russel, G.G.E., et al. List of species of Southern African plants, *Mem. Bot. Surv.S. Africa,* 56, 1987, 270.
161. van Wilgen, B.W. and van Wyk, E., Invading alien plants in South Africa: impacts and solution. *J. Arid Envir.,*1999.
162. Cock, J. and Koch, E. *Going Green: People, Politics and the Environment in South Africa*, Oxford University Press, Cape Town, 1991.
163. Stuart, S.N. and Adams, R.J. South Africa, in *Biodiversity in Sub-Saharan Africa and its Islands*, Stuart, S.N. and Adams, R.J. (Eds.), International Union for the Conservation of Nature and Natural Resources, Oxford, U.K., 1990, 186.
164. Sharma, B.D., et al. Flora of India, *J. Bot. Surv. Ind.,* 1, 1993, 1.
165. Harcourt, C.S. and Sayes, J.A., *The Conservation Atlas of Tropical Forests — The Americas*. Simon and Schuster, New York, 1996.

166. Mares, M.A., Braun, J.K., and Gettinger, D. Observations on the distribution and ecology of the mammals of the Cerrado grasslands of central Brazil, *Annals of Carnegie Museum*, 58, 1989,1.
167. Sick, H. *Birds in Brazil,* Princeton University Press, Princeton, NJ, 1993.
168. Brazil Diversity. *First National Report on the Convention on Biological Diversity,* Brazil. Minister of State for the Environment, Water Resources and the Amazon Region, Brasilia, Brazil, 1998.
169. Rodrigues, G.S. Personal communication, 1999.
170. Saxena, K.G. Biological invasions in the Indian subcontinent: review of invasion plants, in *Ecology of Biological Invasions in the Tropics,* Ramakrishnan, P.S. (Ed.), International Scientific Publications, New Delhi, 1991, 53.
171. Species Diversity. http://www.wcmc.org.uk/igcmc/main.html# Species Diversity WWW Virtual Library. October 3, 1998.
172. Mammals Checklist for India, 1998. http:www.wcmc.org.uk/igcmc/rl_anmal/mammals.html, October 3, 1998.
173. WCMC. Fishes, India. World Conservation Monitoring Center. Species Diversity in India. http://www.wcmc.org.uk/igcmc/main.html, 1998.
174. Saini, M.S. and Kaur, H., Damage by blue rock pigeons *Columba livia* to peas *Pisum sativum* var. arvense. *Ind. Jour. Agric. Sci.,* 56, 1986, 487.
175. WCMC. www.emb/-heidelbergde/vetz/living.~reptiles.html,1998.
176. Talwar, P.K. and Jhigran, A.G. *Inland Fishes of India and Adjacent Countries*, Oxford and IBH Pub. Co, New Delhi, 1991.
177. Ghosh, A.K. *The Red Data Book on Indian Animals Part 1,* Zoological Survey of India, Calcutta, 1994.
178. Gupta, S.K. *Plant Mites of India, Handbook,* Zoological Survey of India Publications, Calcutta, 1985.
179. Redhead, T.D., et al. Mammals introduced to Southern Australia, in *Biogeography of Mediterranean Invasions,* Groves, R.H. and DiCastri, F. (Eds.), Cambridge University Press, New York, 1991, 293.
180. CDC. Summary of Notifiable Diseases, United States, 1995. *Mort. Morb. Weekly Rep.* Commun. Dis. Cent. 44, 1996, 1.
181. Berridge, V. *AIDS in the UK*, Oxford University Press, Oxford, UK, 1996.
182. CDHFS. 1997–98. Commonwealth Department of Health & Family Services Annual Report 1997–98, National Capital Printing, Australia, 1997–1998.
183. South Africa, South Africa Government on Line: http://www.gov.za, 1999.
184. www.nic.in,1998.
185. USPHS. *For a Healthy Nation: Returns on Investments in Public Health,* U.S. Department of Health and Human Services, Public Health Service, Washington, D.C., 1994.
186. Hurley, S.F., et al. AIDS. *AIDS*, 9, 1995, 777.
187. Webb, D. *HIV and AIDS in Africa,* Pluto, London.
188. http://www.unaids.org./highband/document/epdidemio/june98/fact sheets/index.html
189. l = http://www.avert.org/usastata.htm 1999
190. m = http://www.avert.org./statsage/htm 1999; Charles Bowman, charlesb@atp.org.uk
191. http://158.232.21.21.5/Revised/Cube_frame.html 1999
192. http:hivinsite.ucsf.edu/international/asia/latin_america/2098.3bd9.html 1999.

Index

A

C

D

E

F

G

H

I

L

Q

R

S

T

U