Mapping cryptic invaders and invasibility of tropical forest ecosystems: *Chromolaena odorata* in Nepal

Chudamani Joshi

Promotors:

Prof. Dr. Andrew K. Skidmore
Professor of Vegetation and Agricultural Land Use Survey
International Institute for Geo-information Science & Earth Observation (ITC),
and
Wageningen University, the Netherlands

Prof. Dr. Jelte van Andel
Professor of Plant Ecology, University of Groningen
the Netherlands

Co-promotors:

Dr. Jan De Leeuw
Associate Professor of Environmental Science
ITC, Enschede, the Netherlands

Dr. Iris C. van Duren
Assistant Professor of Landscape Ecology
ITC, Enschede, the Netherlands

Examining Committee:

Prof. Dr. Steven M. de Jong, Utrecht University, the Netherlands
Prof. Dr. David M. Richardson, Stellenbosch University, South Africa
Prof. Dr. Ken E. Giller, Wageningen University, the Netherlands
Prof. Dr. Marinus J.A. Werger, Utrecht University, the Netherlands

This research is carried out within the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) in Wageningen University, the Netherlands

Mapping cryptic invaders and invasibility of tropical forest ecosystems: *Chromolaena odorata* in Nepal

Chudamani Joshi
MSc (Botany), MSc (Natural Resource Management), LL-B (Administrative Law)

Thesis
To fulfil the requirements for the degree of Doctor
on the authority of the Rector Magnificus of Wageningen University
Prof. Dr. M.J. Kropff
to be publicly defended on Monday 22 May, 2006 at 15:00 hrs
in the auditorium at ITC, Enschede, The Netherlands

Mapping cryptic invaders and invasibility of tropical forest ecosystems: *Chromolaena odorata* in Nepal.

ISBN: 90-8504-470-7
International Institute for Geo-information Science & Earth Observation,
Enschede, the Netherlands (ITC)
ITC Dissertation Number: 133

Table of contents:

Figures:

Tables:

Dedicated to my *Guru*,

Dr. Jan De Leeuw

Abstract

For centuries, people continuously increased the rate of biological invasions and there is no sign of slowing down. From the depth of the Ocean to the crest of Himalayas, they are occupying pristine and semi-natural ecosystems at an alarming rate, threatening human, animal, plant as well as ecosystem health. Efforts to avoid or eradicate them are not achievable except for very few cases. Currently, therefore, their management aims at controlling invaders and mitigating their impact rather than eradication. Limitation of resources forces land managers to carefully plan and prioritize interventions only in areas most severely affected by invaders. Hence, information on the actual and potential distribution of invaders is considered crucial for their management.

It has long been recognized that remote sensing (RS) and geographical information systems (GIS) could contribute to help solving this problem. Remote sensing has so far been applied predominantly to invasive species that dominate the canopy of the ecosystem. The large majority of invasive species do however not show up in the canopy and thus remain difficult to detect by remote sensing in a direct and straightforward manner. Techniques for mapping such cryptic invaders have not been developed so far. In this thesis we explored methods to map the distribution of *Chromolaena odorata* (L.) RM King & H Robinson, one of the world's worst invasive species. This cryptic heliophyte originating from central America invaded the understorey of many tropical forest ecosystems throughout the world. *C. odorata* is a cryptic invader hidden under the forest canopy in the *Terai* of Nepal. It occurs predominately in opened up forest, with increased light intensity. The approach to map its distribution was to develop first remote sensing techniques to map forest canopy density and light intensity reaching the understorey and next relate these radiation maps to various aspects of the life history of *C. odorata*.

To set the scene we reviewed in chapter 2 existing methods to map the distribution of invasive species. Next, we explored the quality of four alternative methods to predict forest canopy density (Chapter 3). This comparative study revealed that an artificial neural network best explained canopy density in terms of variance explained and bias. A Landsat ETM+ image processed through a neural network predicted 81% of light intensity reaching the understorey. The resulting radiation map was the environmental data layer that was subsequently used to map the distribution of *C. odorata.* This study revealed that in the Nepalese *Terai C. odorata* failed to produce seed below a light intensity of 6.5 mJ m^{-2} day^{-1}, and that light intensity determined 93% of the variation in $\log_{10}$ seed production per plant

(Chapter 4). This enabled us to map its distribution in Nepal based on under-canopy light intensities.

C. odorata invades new areas by generative reproduction (wind dispersal of pappus-bearing achenes). Once established, clonal propagation through underground corms enhances further expansion of populations. We have discovered (Chapter 5) that the age of corms can be determined using corm rings in cross sections. While individual corms survive for only five years, we obtained evidence that multi-corm genets, which must have been much older, had developed in forests with opened canopy. We furthermore demonstrated that light intensity positively related to the rate of clonal growth. The light dependence of the expansion rate of plants is apparently a key attribute explaining the invasion success of this species. Any disturbances in forest canopy density leading to increased light intensities would ultimately trigger its clonal growth.

The canopy in tropical forest ecosystems in Nepal is severely degraded offering light conditions suitable for *C. odorata*. The degradation of the forest canopy was attributed to a series of interrelated causes including human dimensions and government policies (Chapter 6). Herbarium records revealed that *C. odorata* invaded southern Nepal shortly after the initiation of malaria control. We furthermore demonstrated that this was followed in Chitwan district by an influx of migrants, land use changes and degradation of the forests. It was therefore argued that the species invaded because of canopy degradation, which was caused by change in land use and demography and triggered by malaria control. This process of malaria control followed by migration increased human population growth rates that we described for Chitwan district occurred in tropical regions all over Nepal. Hence, we suggest that it must have contributed to the rapid spread of *C. odorata* in the lowland *Terai* forest belt.

Field observations revealed that *C. odorata* did not invade the tropical forests in the west of Nepal below 83°45’ east longitude. We were unable to explain this distribution pattern with simple climatic indicators. However, a bioclimatic indicator, the length of the growing season predicted the absence from west Nepal remarkably well. This suggests that *C. odorata* requires a minimum length of the growing period to accumulate sufficient resources to establish and persist. We thus conclude that an agro-ecological modeling approach yielded a better prediction than the commonly used bioclimatic approach (Chapter 7).

Since, deforestation and forest degradation are a point of concern for management of both biological invaders and native biota, there is a need to more closely monitor biological conservation areas because of the potentially irreversible impacts of deforestation and forest degradation. In chapter 8 we assess the rate of deforestation and the current degree of forest degradation in the *Terai* of Nepal. Forest canopy density class was predicted with 82% overall accuracy. Data analysis revealed that the forested area reduced from 21774 km^2 in 1958 to 12649 km^2 in 2000 corresponding to an annual rate of decline of 1.38%. Our analysis further revealed that 70% of the forested area outside conservation areas had canopies with density below 60%, thus confirming widespread degradation. More surprisingly, 50% percent of the forested area inside protected areas had such opened canopies. This indicates that canopy degradation is also very common inside protected areas. These areas play a special role in the conservation of internationally threatened forest communities, for instance Nepalese tropical rain forests. Our analysis revealed that canopy opening prevailed as well in these communities. We argue that from a biodiversity point of view conservation effort should focus on the preservation and restoration of these forest types. The forest degradation maps presented in this chapter could serve as a start to prioritize such interventions.

In this thesis, we demonstrated how the impressive developments in computational performance, the rapid growth of remote sensing and GIS technologies for spatial data acquisition and analysis could be used beyond their traditional application in mapping canopy-dominant invasive species. We have shown how a few of these cost-effective mapping techniques can reliably be up scaled to the national level to map the distribution of even those invasive species that do not dominate the canopy of forest ecosystems.

This thesis emphasizes the importance of site-specific microclimatic variation and empirical observations of the species' ecology, while applying remote sensing techniques in invasion studies. This could significantly reduce the uncertainties and the degree of "erroneous prediction". Maps displaying seed-producing sites could be used to significantly reduce the costs of controlling *C. odorata* infestation by providing information on the spatial segregation of source and sink populations. These will support efficient habitat ranking to restore invaded areas and protect non-invaded ecosystems. Such an approach may prove particularly valuable when implementing control measures under circumstances of limited capital and labour. This thesis also showed the necessity to understand the connection between the human historical, socio-economic, and cultural context with the environmental conditions and the ecology of the invader. It facilitates conceptualising the situation and hopefully it also helps in translating research results into appropriate policy measures.

Abstract in Dutch (Samenvatting)

Al eeuwenlang dragen menselijke activiteiten bij aan de verspreiding van soorten. In sommige gevallen levert dit geen probleem op. Maar, de enorme toename van biologische invasies is een steeds grotere zorg voor natuurbehoud en -beheer. Van de diepste oceanen tot de top van de Himalaya verstoren invasieve soorten natuurlijke en door de mens gebruikte ecosystemen. Dit vormt een bedreiging voor ons milieu; zowel voor de van oorsprong aanwezige planten en dieren als voor de gezondheid van het hele ecosysteem. De snelheid waarmee dit gebeurt, neemt alsmaar toe en er is geen enkel teken van vermindering van het probleem. Pogingen om biologische invasies te voorkomen of terug te dringen zijn slechts in enkele zeldzame gevallen succesvol geweest. Daarom concentreren natuurbeheerders zich nu voornamelijk op het onder controle houden van de schade van zulke soorten in plaats van het totaal terugdringen van de soort. Beheerders en managers hebben meestal te kampen met beperkte financiële middelen en beschikbare arbeidskracht. Daardoor moet men interventies zorgvuldig plannen zodat men een zo groot mogelijk rendement kan behalen. Het is duidelijk dat daarbij informatie over de werkelijke en de potentiële verspreiding van een invasieve soort van cruciaal belang is.

Het is inmiddels bekend dat het gebruik van geografische informatie systemen (GIS) en verschillende aardobservatie technieken, ofwel remote sensing (RS), een goede bijdrage kunnen leveren aan het in kaart brengen van de verspreiding van al dan niet invasieve soorten. Tot nu toe is remote sensing voornamelijk gebruikt om soorten te karteren die dominant voorkomen in de bovenste laag van de vegetatie van het ecosysteem. De grote meerderheid van alle invasieve soorten maakt echter geen deel uit van deze bovenste laag en blijft dus moeilijk rechtstreeks te detecteren met behulp van remote sensing technieken. Methoden om zulke wat moeilijker waarneembare en meer verborgen soorten in kaart te brengen zijn nog niet beschikbaar. In dit proefschrift worden methoden onderzocht om de verspreiding van een van de meest problematische invasieve soorten ter wereld, *Chromolaena odorata* (L.) RM King & H Robinson, in kaart te brengen.

Deze van oorsprong centraal Amerikaanse soort, waarvan men weet dat hij afhankelijk is van voldoende licht, komt tegenwoordig voor in de ondergroei van bossen in de gehele tropische zone van onze aardbol. *Chromolaena odorata* is in Nepal binnengedrongen in de bossen in het laaggelegen deel van het land, genaamd de Terai. De soort wordt voornamelijk gevonden op plaatsen waar het bos wat meer open is en waar voldoende licht door de boomkronen heen valt en de

ondergroei bereikt. In deze studie is geprobeerd eerst een methode te ontwikkelen om de dichtheid van het bladerdek van de bomen in kaart te brengen. Vervolgens werd dit vertaald naar de hoeveelheid licht die beschikbaar was voor de ondergroei. Tenslotte hebben we deze resultaten gerelateerd aan de verspreiding en verschillende eigenschappen van *C. odorata*.

De eerste stap in dit onderzoek was het creëren van een overzicht van alle bestaande methoden voor het in kaart brengen van de verspreiding van invasieve soorten. Dit is beschreven in hoofdstuk 2. Daarna werden vier verschillende methoden om de dichtheid van het bladerdek van de boomlaag te bepalen met elkaar vergeleken. Deze studie, beschreven in hoofdstuk 3, laat zien dat een neuraal netwerk het best presteerde in het meten van de dichtheid van het bladerdek. Een Landsat ETM+ satellietbeeld dat gebruikt werd in neuraal netwerk voorspelde met een nauwkeurigheid van 81% de lichtintensiteit die beschikbaar was voor de ondergroei in het bos. Deze lichtintensiteitkaart werd vervolgens als belangrijkste milieuvariabele gebruikt om de verspreiding van *C. odorata* te voorspellen. Hoofdstuk 4 van dit proefschrift laat zien dat in de Nepalese Terai, *C. odorata* geen zaad produceert als de lichtintensiteit lager is dan gemiddeld 6.5 mJ m^{-2} per dag. Verder blijkt lichtintensiteit 93% van de variatie te verklaren in de $\log_{10}$ getransformeerde zaadproductie per plant. Hierdoor is het dus mogelijk de verspreiding van deze soort in kaart te brengen op basis van lichtintensiteit die onder het bladerdek van het bos beschikbaar is.

Chromolaena.odorata dringt nieuwe gebieden binnen met behulp van generatieve voortplanting via zaad en windverspreiding. Zodra planten gevestigd zijn kan voortplanting en uitbreiding van populaties ook plaatsvinden via klonale groei en voortplanting via stengelknollen. We hebben ontdekt (hoofdstuk 5) dat de ouderdom van stengelknollen kan worden bepaald door een doorsnede te maken en jaarringen in de knol te tellen. Een individuele stengelknol blijkt maximaal 5 jaar oud te kunnen worden. Er werd echter ook gevonden dat in meer open bosvegetatie door aanmaak van steeds nieuwe stengelknollen, klompjes van die knollen een leeftijd bereikten die veel hoger lag. Tevens werd aangetoond dat lichtintensiteit positief gecorreleerd was met de mate van klonale groei. De afhankelijkheid van licht is blijkbaar de meest belangrijke eigenschap die het invasieve succes van deze soort bepaalt. Indien verstoring van de boomlaag in het bos leidt tot verhoging van de lichtintensiteit dan zal dit uiteindelijk *Chromolaena odorata* kunnen aanzetten tot klonale groei.

De kroonlaag van vrijwel alle tropische bosecosystemen in Nepal is zwaar gedegenereerd en biedt lichtcondities in de ondergroei die geschikt zijn voor *Chromolaena odorata*. Deze achteruitgang kan worden toegeschreven aan een serie van aan elkaar gerelateerde menselijke activiteiten, beleid en management (hoofdstuk 6). Uit herbariumgegevens blijkt dat *C. odorata* via het zuiden in Nepal binnendrong kort na de eerste pogingen malaria te verdrijven uit het laagland. We hebben ook laten zien dat dit in het district Chitwan resulteerde in een toestroom van migranten, veranderingen van landgebruik en bosdegradatie. Daarom betoogt dit onderzoek dat *C. odorata* binnendrong als resultante van bosdegradatie veroorzaakt door veranderingen in landgebruik en demografie en uiteindelijk geïnitieerd werd door maatregelen ter controle van malaria. Dit proces werd uitgebreid beschreven voor het Chitwan district maar vond in feite plaats in alle laagland gebieden binnen Nepal. Daarom wordt gesuggereerd dat dit de oorzaak is van de snelle verspreiding van *Chromolaena odorata* over een zeer groot deel van de gehele laagland zone van Nepal.

Uit veldwaarnemingen bleek echter dat *Chromolaena odorata* zich niet verspreidde in het westen van Nepal, ten westen van de lijn met breedtegraad 83°45’ oosterlengte. Het bleek onmogelijk dit verspreidingspatroon te verklaren met behulp van eenvoudige klimaatfactoren. Maar de factor “lengte van het groeiseizoen” bleek in dit geval een buitengewoon goede indicator voor de verspreiding van Chromolaena. Het lijkt erop dat deze soort een bepaalde minimumlengte van de groeiperiode nodig heeft om voldoende reserves te ontwikkelen om zich blijvend te kunnen vestigen. Daarom concluderen we dat bij het in kaart brengen van *Chromolaena odorata* in het laagland van Nepal, de toepassing een agro-ecologisch model een betere benadering is dan de meer gebruikelijke bioklimaatmodellen (hoofdstuk 7).

Ontbossing en bosdegradatie zijn punten van zorg voor zowel het behoud van oorspronkelijke biota als voor het in de hand houden van invasieve soorten. Daarom is een goed monitoring programma nodig vanwege de mogelijk onomkeerbare gevolgen van ontbossing en bosdegradatie. In hoofdstuk 8 wordt ingegaan op de snelheid van ontbossing en de huidige stand van zaken betreffende bosdegradatie in het Nepalese laagland. De dichtheid van het bladerdek van de bossen werd vastgesteld en geclassificeerd met een nauwkeurigheid van 82%. Analyses lieten zien dat het bosgebied verkleind werd van 21774 km^2 in 1958 tot 12649 km^2 in 2000. Dit komt overeen met een gemiddelde jaarlijkse afname van 1.38% van het bosoppervlak. Verder bleek dat 70% van de bossen buiten de beschermde gebieden een dichtheid van het bladerdek hadden wat lager lag dan

60%, wat op verregaande degradatie duidt. Nog verrassender was dat 50% van de bossen binnen beschermde gebieden ook een dergelijke open structuur bleken te hebben wat ook ernstige bosdegradatie binnen protectiegebieden betekent. Deze gebieden spelen een belangrijke rol in de bescherming van internationaal bedreigde bosgemeenschappen zoals bijvoorbeeld de Nepalese tropische regenwouden. Vermindering van de dichtheid van kroonlaag bleek ook in deze bossen een algemeen voorkomend verschijnsel. Vanuit het oogpunt van bescherming van biodiversiteit is dus bescherming en herstel van juist deze bostypen erg belangrijk. Kaarten die bosdegradatie weergeven, zoals beschreven in dit hoofdstuk, kunnen een goed beginpunt zijn voor het stellen van prioriteiten bij dit soort interventies.

Uit deze thesis blijkt hoe de enorme uitbreiding in rekencapaciteit, de snelle ontwikkeling van remote sensing en GIS technieken voor het verzamelen en verwerken van ruimtelijke gegevens gebruikt kan worden buiten de traditionele toepassingen. Voorheen bleef de kartering van invasieve soorten vaak beperkt tot rechtstreekse waarneming van soorten die deel uit maakten van de toplaag van de bosvegetatie. Nu kunnen we ook indirect minder goed zichtbare soorten in kaart brengen. Daarnaast hebben we laten zien dat een aantal kostefficiënte karteringstechnieken met goede betrouwbaarheid van lokaal naar nationaal niveau kunnen worden vertaald.

Deze thesis benadrukt het belang van standplaatsspecifieke variaties in microklimaat en veldwaarnemingen van eigenschappen van een soort binnen de toepassing van GIS en remote sensing modellen. Zorgvuldige toepassing hiervan bij het in kaart brengen van invasieve soorten kan onzekerheden in uitkomsten verminderen en het risico op verkeerde voorspellingen terugdringen. Voor *Chromolaena odorata* kan het gebruik van kaarten die de zaadproducerende “source-populaties” ruimtelijk scheiden van de niet reproductieve “sink-populaties” een aanzienlijke vermindering in kosten opleveren voor het in de hand houden van deze soort. Hiermee kunnen namelijk veel beter prioriteiten gesteld worden voor wat betreft in welke gebieden interventiemaatregelen genomen moeten worden. Dit is in alle gevallen van belang maar zeker in omstandigheden waar kapitaal en arbeidskracht slechts zeer beperkt beschikbaar is. Tenslotte blijkt uit deze thesis dat het noodzakelijk is een goed begrip te hebben van de historische, sociaal-economische en culturele context in samenhang met de milieuomstandigheden en de eigenschappen van de invasieve soort. Dit maakt het mogelijk de situatie goed te doorzien en hopelijk de onderzoeksresultaten te vertalen in doeltreffende beleidsmaatregelen.

Abstract in Nepali (सारांश)

शताब्दि देखिनै मानव द्वारा तिब्र पारीएको मिचाहा प्राणीको आक्रमणको गति कम हुने संकेत देखिएको छैन । प्रशान्त माहासागरका गहिराई देखि हिमालयका उचाई सम्म आफ्‌नो प्रकृतिक वासस्थान बाट विशुद्द प्राकृतिक जीवपरिबृत्ती तन्त्रमा द्रुत दरमा फैलिरहेका विदेशि प्राणीले मानव, जन्तु, बनस्पति तथा जीवपरिबृत्ती पद्दतीको स्वास्थ्यलाई चुनौति दिईरहेका छन् । यी प्राणीहरुको फैलावट रोक्ने तथा निर्मूल पार्ने जमर्को अति न्युन प्राणी संख्यामा बाहेक सफल देखिदैन । त्यसैले हालसालै मिचाहा प्राणीलाई निर्मुल पार्नु भन्दा ब्यबस्थापन गरि तिनहरुको प्रतिकुल प्रभाव न्यून गर्ने निति छ । श्रोतको न्यूनतमा मिचाहा प्राणी ब्यवस्थापनलाई प्रथमिकता आधार क्षेत्र चयन गर्न बाध्य बनाएको छ । त्यसैले मिचाहा प्राणीहरुको यथार्थ बस्तुस्थिति, तथा बासस्थानको जानकारी हुनु अति आवश्यक हन्छ ।

लामो समयदेखिनै दुर संबेदन तथा भू–सूचना प्रणाली यस कुरामा योगदान पुऱ्याउन सक्ने प्रमाणित भएको छ । तापनि दूर संबेदन प्रणाली प्राकृतिक पर्याबरणको माथिल्लो तहमा बास गर्ने प्रजातीहरुमा मात्र लागु गर्न सकिएको छ । मिचाहा प्राणीहरुको तथ्यांकमा अति ठूलो संख्यामा मिचाहा प्राणीहरु प्राकृतिक पर्याबरणको माथिल्लो तहमा बास नगर्ने देखिन्छ जहाँ दूर संबेदन प्रणाली को प्रत्यक्ष प्रयोग गर्न सकिदैन । अतः यस्ता लुकेर बस्ने प्राणीहरुको मापन प्रविधिको विकाश भएको छैन । त्यसैले प्राकृतिक जीवपरिबृत्ती पद्दतीको लुकेर बस्ने यस्ता मिचाहा प्राणहिरुको मापनमा मैले नयाँ प्रविधि सुझाएको छु । यस प्रविधिलाई मैले बनमारा *(Chromolaena odorata)* वनस्पतिको वास्तबिक तथा बस्तुगत बासस्थान मापनमा प्रयोग गरेको छु । यो वनस्पति संसारका १०० सबै भन्दा खराब मिचाहा वनस्पतिको श्रेणिमा पर्दछ । यो संसारका उष्ण प्रदेशिय देशहरुका वन जंगल भित्र फैलिएको छ । यस अध्ययनमा सर्व प्रथम प्राकृतिक वन जंगल को माथिल्लो तह तथा वन जंगल को तल्लो तह सम्म पुग्ने प्रकाश मापनका बिधिहरुको बिश्लेषण गरेको छु । तत् पश्चात यी प्रकाश मापनका नक्साहरु संग वनमाराको जीवन पद्धतिका बिभिन्न पक्षहरुको सम्बन्ध स्थापित गरिएको छ ।

आरम्भ खण्ड २ मा मिचाहा वनस्पति मापनमा विद्यमान विधिहरुको सिंहावलोकन गरिएको छ । तत् पश्चात वनछत्र घनत्व मापनका चार वैकल्पिक विधिहरुको श्रेष्ठताको अन्वेषण गरेको छु (खण्ड ३) । प्रकाश मापनको यस तुलनात्मक अध्ययनमा स्नायु तन्त्रको जन्जाल प्रविधि बाट वन वनछत्र घनत्व लाई लगभग ८१ प्रतिशत सत्यताका साथ मापन गर्न सकिने परिणामले देखाएको छ । यो प्रविधि अन्य तिन दाँजिएका प्रविधि भन्दा बढि सत्यता प्रदर्शन गर्ने तथा न्युन भ्रमांश परिणाम दिने देखिएको छ ।

यस प्रविधिको प्रयोगले ८१ प्रतिशत वन प्राकृतिक जीवपरिबृत्ती पद्दतीको तल्लो तह सम्म पुग्ने प्रकाश लाई नाप्न सकिने देखाँउछ । वनमारा वनस्पति ६.५ माईक्रो जूल प्रति मिटर प्रति दिन भन्दा न्यून प्रकाश पर्ने नेपालको तराइका स्थानहरुमा बिउ उत्पादन गर्न नसक्ने देखिएको छ । बीउ उत्पादन प्रक्रियामा ९३ प्रतिशत सत्यताको जाँच प्रकाश मापन बाट गर्न सकिन्छ । वनको तल्लो तहमा पुग्ने प्रकाश मापनबाट हामिलाई नेपालमा वनमाराको बितरण मापन गर्ने सामर्थ्य प्रदान गर्दछ (खण्ड ४) ।

वनमाराले वायु प्रसारित बिउहरुबाट नयाँ स्थानहरु मिच्ने गर्दछ । एक पटक स्थापित विरुवाले जमिन मुनि रहेको गानो द्वारा पनि आफ्नो संख्या बृद्धि गर्दछ । यो परिणामले “गाँनो औंठी” गणना गरि गानोको उमेर थाहापाउन सक्ने देखाउँछ (खण्ड ५)। सधन वनछत्र मा समेत वनमारा दीर्घजीवि हुनसक्छ । गानो बृद्धिमा प्रकाशको अहम् भूमिका रहने कुरा यस अध्ययन बाट प्रष्ट भएको छ । यो लचिलोपना यसको मिचाहा प्रबृत्तिको सफलताको मुख्य गुण भएको देखिन्छ । वन पारिस्थितिमा कुनै दखल हुँदा वनमाराको हु–बहु जीवोत्पादनलाई बढावा दिने गर्छ । एक्लो गानो पाँच बर्ष सम्म मात्र जीवित रहन सक्छ तापनि बहुगानोयुक्त बिरुवा खुला वनछत्र मुनि दीर्घ–जीवि हुन सक्छ ।

नेपालको उष्ण वन वातावरणको माथिल्लो तहमा निरन्तर रुपमा दखल हुने गरेको छ तथा वनमाराको गानोले निरन्तर मिचाहा प्रबृत्तिको भय देखाएको छ । यो जनसंख्या तथा सरकारी नीति लगायतका जटिल अन्तर–संबन्ध संग गासिएको छ (खण्ड ६) । वनसपति नमूना बिश्लेषणले नेपालको तराई क्षेत्रमा ओलो उन्मूलन सँगै वनमारा भित्रिएको संयोग देखिन्छ । औलो उन्मुलन पश्चात चितवनको तराई तिर बसाँई सर्ने कम तिब्र भयो । यसले भू–क्षय तथा भू–उपयोगमा परिवर्तन ल्याई चरन तथा घाँस दाउराको मागमा उल्लेख्य बृद्दि भई बन परिबृत्ती पद्दतीको माथिल्लो तहमा बिनास ल्यायो । अतः यो भन्न सकिन्छ की औलो उन्मूलनका कारण निम्तिएको वन बिनास र अति चरनले अन्य रैथाने विरुवाको बाहुल्यता रोकी वनमाराको बृद्धि तथा फैलावटका लागि सुहाँउदो वाताबरण सृजना गर्न टेवा पुर्‍यायो । यो अध्ययन चितवन जिल्लामा गरिएको भएतापनि यो प्रक्रियाले नेपालको तराई क्षेत्रकै प्रतिनिधित्व गर्दछ ।

हाल नेपालमा वनमारा ८३°४५’ पुर्बिय देशान्तर भन्दा पश्चिममा फैलिएको पाँइदैन । साधारण वातावरणिय सूचकहरु द्वारा यस प्रजातिको फैलावट प्रष्ट्याउन सकिदैंन तापनि वनस्पति हुर्कन आवश्यक मौषमि समय (Length of growing season) जस्ता कृषि–वातावरणिय सूचकहरुले पश्चिम नेपालमा यस विरुवाको अनुपस्थिति प्रष्ट्याउँछ । स्थानिय परिस्थितिमा जंगलभित्रको प्रकाश तथा मानव दखलको तिब्रताले बनमाराको बासस्थान तय गर्दछ भने राष्ट्रीय स्तरमा वनस्पति हुर्कन आवश्यक मौषमि समय तथा तापक्रमले वनमाराको बासस्थान सिमांङ्कन गर्ने पाईएको छ । बिरुवा बृद्दिको लागि चाहिने मौषमि समय पश्चिम नेपालमा पुर्वको दााजोमा उल्लेख्य रुपमा धेरै कम छ । जसले गर्दा वनमाराले हुर्कन, फुल्न तथा फल्न जम्मा पार्नु पर्ने भोजन श्रोत समय न्यून हुने देखिन्छ । त्यसैले यो भन्न सकिन्छ कि कृषि–वातावरणिय नमूना विधि वनस्पतिको उपस्थिति–अनुपस्थिति मापनमा बढी भरपर्दो हुन सक्छ । अतः कृषि–वातावरणिय सूचकहरु वनमारा मापनमा बढि उपयोगी सिद्ध भएको छ (खण्ड ७) ।

नेपालमा सरकार द्वारा नियन्त्रित वनजंगलको अवस्था दयनिय देखिन्छ । तथापि नीति निर्माताहरुलाइ यो स्थान केन्द्रित (Spatial) जानकारी कत्तिको छ भन्ने प्रश्न आफैमा गहन छ । मिचाहा तथा रैथाने प्राणि ब्यवस्थापनमा वन बिनास र क्षयले महत्वपूर्ण भूमिका खेल्ने भएकोले यो प्रकृया अति नै चासोको बिषय हो । खण्ड ८ मा नेपालको तराईमा वन बिनाश र क्षय दरको बिश्लेषण गरिएको छ । वनछत्र घनत्वलाई ८२ प्रतिशत सत्यताका साथ मापन गर्न सकिन्छ । तथ्यांक बिश्लेषणले सन् १९५८ मा फैलिएको २१७७४ बर्ग किलोमिटर वन क्षेत्र सन् २००० सम्म आई पुग्दा बार्षिक १.३८ प्रतिशतका दरले घटी १२६४९ बर्ग किलोमिटर बाँकी रहेको

देखिन्छ । यस अनुसन्धान वाट ७० प्रतिशत संरक्षित क्षेत्र बाहिरका जंगलको छत्र ६० प्रतिशत भन्दा कम रहेको देखिन्छ । यसले अत्याधिक वन बिनाश र क्षय भएको पुष्टी हुन्छ । अचम्मको अर्को कुरा, संरक्षित क्षेत्र भित्रको वनको ५० प्रतिशत छत्र समेत खुला छ । यसले नेपालका संरक्षित क्षेत्र भित्र समेत वन छत्र बिनाश भैरहेको तथ्य उजागर हुन्छ । यी क्षेत्रहरुले अन्तराष्ट्रीय रुपमै लोपोन्मुख वन प्रकारहरुको संरक्षणमा महत्वपूर्ण भूमिका खेल्ने गर्दछन् । यस अनुसन्धान बाट यस्ता बन प्रकारमा समेत वनछत्रको विनाश भएको देखिएको छ । जैविक विविधताको दृष्टिकोणले यी वनप्रकारको संरक्षणमा विशेष ध्यान दिनुपर्नेमा म जोड दिन चाहान्छु । यस खण्डमा प्रकाशित वन बिनाशका नक्साहरुले वन संरक्षणमा अग्रणि भूमिका खेल्न सक्छन् ।

यस शोध पत्रमा मैंले दूर संबेदन तथा भू–सूचना प्रणालीको प्रभावशाली बिकासले भू–तथ्यांक संकलन, प्रशोधन तथा बिश्लेषणका निकै संभावनाहरु देखाएको प्रमाणित गरें । यस पद्दति द्वारा परम्परागत भू–सूचना उपयोगको भन्दा पर वनजंगलको तल्लो तहमा लुकेर बस्ने मिचाहा वनस्पति मापन समेत गर्न सकिन्छ । खर्च प्रभावकारी यस विधि द्वारा वनजंगलको यथार्थ चित्रण तथा मिचाहा प्राणीको राष्ट्रीय स्तरसम्म भरपर्दो मापन को मेरो यस अनुसन्धानले वन वातावरण तथा वनमारा ब्यवस्थापनमा महत्वपूर्ण योगदान पुऱ्याउने छ भन्ने मैले आशा लिएको छु ।

यस अनुसन्धानले सुक्ष्म–मौषम तथा स्थलगत प्राणी अवलोकनलाई प्राथमिकता दिन्छ । बीउ उत्पादन मापन नक्साले बीउ उत्पादन हुने वा नहुने क्षेत्रको जानकारी दिई वनमाराको नियन्त्रण खर्चमा उल्लेख्य कमी ल्याउन मद्दत पुऱ्याउँनेछ । यसले प्रभावकारी प्राणी बासस्थानको तह निर्धारण गरि मिचिएका र नमिचिएका जीवपरिबृत्ती पद्दतीको पुर्नस्थापनमा सहयोग गर्ने छ । खासगरी अपुग बजेट तथा मानव संसाधन परिवेशमा यो नियन्त्रण उपाय बढी मूल्यवान् साबित हुन सक्छ । अन्त्यमा यस अनुसन्धानले मानव इतिहासको सामाजिक, आर्थिक तथा सांस्कृतिक पक्षहरुको पर्यावरण तथा मिचाहा प्राणि बिचको सम्बन्धको सूक्ष्म ज्ञान हुनुपर्नेमा जोड दिएको छ । यस अनुसन्धानले परिस्थितिको यथार्थ चित्रण गर्न तथा यसको निचोडले उपयुक्त नीति निर्माण गर्न मा समेत सहयोग पुऱ्याउने छ भन्ने मैले विश्वास लिएको छु ।

Acknowledgements

"Without friends no one would choose to live, though he had all other goods." -- Aristotle

On top of the list, my first and foremost gratitude goes to the great team behind me: my promoters and co-promoters Prof. Dr. Andrew K. Skidmore, Prof. Dr. Jelte van Andel, Dr. Jan De Leeuw, Dr. Iris van Duren for their determination, commitment, excellent guidance and innovative ideas. Prof. Dr. Hari Datta Lekhak, my motivation to do a PhD could not have been possible without your administrative support. I am very grateful for the confidence that you all have shown in me and for honouring me to be your student. Equally, I extend my heartfelt thanks to the family of all of you, who always made me feel at home.

Jan, I have realised that my real school is under your feet . Your zeal and devotion in environmental science studies inspired me and offered to me the spark to enlighten my pathway in this field. You always stood by me as a devoted *guru* and as a friend. I learnt the style of scientific writing, the art of critical thinking and handling complicated environmental issues. Einstein said, “The important thing is not to stop questioning. Curiosity has its own reason for existing” and thanks for tolerating all of my “curiosities”. Iris, I am most grateful to you for your “invaluable overall support” which gave me an enormous strength to face real world challenges. Jelte, Jan and Iris, it was an excellent and satisfactory feeling working with you there in the dense jungle of Nepal.

To WOTRO (The Netherlands Foundation for the Advancement of Tropical Research) and ITC (the International Institute for Geo-information Science and Earth Observation), I am deeply honoured to be financed to peruse my higher education in the Netherlands. No less thankful I am to my own organization: The Department of Plant Resources, Ministry of Forest and Soil Conservation. Her institutional contributions in all cases need to be highlighted. I need to point out the support of my former Director General, Mr. Madhu Sudhan Bista a superb supporter well beyond his formal commitments, who has always been ready to provide open and frank support, and many encouragements.

At ITC, several people contributed to the success of this thesis. Technically, Henk van Oosten, Wan Bakx, Jelger Kooistra, Boudewijn van Leeuwen, Gerard Reinink Fabio Corsi, Kees de Bie and Koert Sijmons offered welcomed help with their advanced remote sensing and GIS skills in downloading and processing raw image data. I thank the Library team: Marga, Carla, Petry and Margriet for offering me “every impossible literature”. I also owe a great deal to all who have taught me over the years about the remote sensing and GIS tools, techniques and application, to them before whom I have appeared, and to my colleagues who have shown me with their examples what it means to be a researcher.

Without the logistic support and data to work with, this research was not possible. In this regards, my brother in law Mr. Pushp Raj Joshi and Niranjan Joshi deserve very special thanks for their untiring support in every aspect of this research. My special thanks goes to the Department of Wildlife and National Parks for their administrative support. My thanks also goes to Claudia Pittiglio, Bed Kumar Dhakal, Nawang Norbu and Dadhi Lal Kandel and the people of Chitwan for their help in data collection.

I am particularly indebted to Prof. Dr. Dhadwal and Dr. P.K. Joshi (Indian Institute of Remote Sensing, Deharadun), Mr. Basant Shrestha and Mr. Govind Joshi (ICIMOD, Nepal) for providing me some of the maps to consult. The Department of Forest Survey and Research, Department of Survey and the Central Bureau of Statistics (Nepal) and Department of Health provided some aerial photographs, maps and data; to National Herbarium and Plant Research Laboratory, Godavari, Kathmandu Nepal (KATH) and British Museum (BM) London for providing Herbarium records.

Behind the scene, I have strong supportive hands of Prof. Dr. Ir. Martien Molenaar, Drs. Sjaak Beerens and their great team. I am equally thankful to Prof. Dr. Martin Hale and Ms. Loes Colenbrander for their administrative support which is part of what makes this research work possible. My special thanks goes to the Department of Natural Resources team in particular who shared and critically supported my research work in many scientific presentations, to Esther Hondebrink, Daniela Semeraro (now working with Dutch Ministry for foreign Affairs) and Petra Weber for their kind administrative support and friendship, to Benno Masselink, Job Duim, Gerard Leppink, Harold, Martin Blankestijn, Michel Erdmann, Ard Kosters and Cecille for their technical support. Equally I am thankful to the supporting team of the financial, marketing and project services, facility management, student affairs, ITC international hotel, communications departments and to all ITC family for everyday daily support. I thank you all.

The ITC is an institution that I have long held in reverence. During my 5 years as a student and researcher, I've had the opportunity to attached to ITC from a variety of perspectives: as a student in the Rural Land Ecology Group, organising MSc field work as a team member, served student community as a member of Student Association Board, supervise MSc field work as a researcher, and as a president of the PhD community of the ITC. During all of that time, my appreciation of the vital role that the ITC plays in the geo-information world has greatly deepened. I have enjoyed the close support and excellent company of all my PhD colleagues. I offer my sincere thanks to all of you for your vote, excellent friendship and support and wish you all a very successful and a bright future.

I am grateful to Manfred and Reena Pulles, (Netherlands), who gave me and my family a homely environment and whose friendship, affection, kindness and hospitality has made my stay pleasure. I am also thankful to Bimala Thapa (Netherlands), Zihni and Dr. Antje Erencin (Germany), Dr. Tom and Floor Bakx (Netherlands), Dr. Dhruba Pikha Shrestha and Dr. Arbind Tuladhar (Netherlands) for their kind hospitality. My special thanks also goes to all Nepalese friends who joined ITC between 1998-2006 gave me and my family a homely environment and to all global geo-friends who enriched me with invaluable memories.

The acknowledgement would not be complete with encouragement and support of my family and relatives at home. I am indebted to my parents-in-law, Mr. Bishnu Bhakta Joshi and Mrs. Kalavati Joshi and their family, who kept me in touch with normal life. Without their unreserved and invaluable support, care and encouragement, I would never complete this work. Both of my brothers-in-law Pushp Raj and Niranjan gave me a consistent encouragement and logistic support throughout this research.

I owe a great deal to the family of my both brothers and the family of my sister who have always been a great friend and an inspiration in every step of my life. On top of all, I owe a great deal to my mother, wife, daughter and son. I wish that my father, an extraordinary person who spent his all short of possible resources on me, had lived to see this day. My mother, who is still recording vegetation dynamics and species phenology for my research, is always inspiring me with an invaluable love of learning. My children, Richa and Rishav are the pride of my life. My wife Geeta has been a constant source of love, support and inspiration. Without her moral support and unprecedented sacrifice, I would be forced to work “in the real world” and not be allowed the freedom to pursue my dreams. Geetu, you have my most sincere thanks and gratitude.

P.S.: Please address all future communications to me at my ITC e-mail address: joshi@itc.nl or National Herbarium, Godavari, Kathmandu, Nepal.

CHAPTER ONE

Mapping cryptic invaders and invasibility of tropical forest ecosystems

1 Mapping cryptic invaders and invisibility of tropical forest ecosystems: *Chromolaena odorata* in Nepal

1.1 Consequences of biological invasions

Invasive species have intrigued ecologists for long (Elton 1958). In recent years, the establishment and spread of such species in areas where they do not occur naturally, are receiving increasing importance from scientists, policymakers and the public. Numerous studies demonstrated the dramatic effect of invaders on recipient ecosystems (Mack *et al*. 2000). Introduced species are among the greatest threats to native biological diversity and the functioning of natural ecosystems. Bio-invasion has been homogenizing the world's flora and fauna (Hobbs 2000a), altering bio-geo-chemical cycles (Polley *et al*. 1994) and is recognized as a primary cause of global biodiversity loss (Czech and Krausman 1997, Wilcove and Chen 1998) and species extinction (di Castri 1989).

1.2 Invasive species and biological invasion

What are "invaders"? The verb "to invade" meaning "to enter in a hostile manner" has existed for about 500 years (Oxford English Dictionary 1989). "Invasion" originally referred specifically to human warfare, the "action of invading a country or territory as an enemy". Its meaning then expanded to include "a harmful incursion of any kind". In law and philosophy, the meaning includes "encroachment upon the property, rights, privacy, etc of anyone". The current meaning combines the concept of entering a foreign territory and having some short of harmful impact.

What does the terms invaders and invasion stand for when used in an ecological context? Elton, " the father of invasion ecology" did not seem it necessary to define these terms in his classical book on invasion (Elton 1958). Since then, a number of definitions of invaders and invasion have been proposed. Kolar and Lodge (2001) defined invasive species as "a non-indigenous species that spreads from the point of introduction and becomes abundant". According to di Castri (1990) "A biological invader is a species of plant, animal or micro-organism which, most usually transported inadvertently or intentionally by man, colonizes and spreads into new territories some distance from its home territory". Definitions for the common term 'invasive' include non-indigenous species that have colonized natural areas (Burke and Grime 1996) as well as widespread species that have negative effects on the recipient ecosystem (Davis and Thompson 2000, Mack *et al*. 2000). Richardson *et al.* (2000) proposed a biogeographical approach that described invasions as a process in which non-indigenous species progress through ‘established’, ‘naturalised’, and ‘invasive’ stages. Based on similar approach, Colautti and MacIsaac (2004) proposed an invasion stage based neutral terminology under which, a non-indigenous species passes through localized and numerically rare to widespread and dominant stage.

Several terms have been used more or less synonymously to address invasive species including aliens, naturalized species, non-indigenous, invaders, pests, colonizers, weeds, immigrants, exotics, neophytes, transformers and introduced species (Jarvis 1979, Heywood 1989, Dukes and Mooney 1999, Lonsdale 1999b, Williamson 1999, Berdichevsky *et al*. 2000, Brabec and Pysek 2000, Colautti and MacIsaac 2004) or even more colourful terms, for instance bioterrorists (Meyerson and Reaser 2003). These terms however have different meanings (Binggeli 1996, Alpert *et al*. 2000, Richardson *et al*. 2000). The mixing of closely related terms that do not articulate exactly the same concept may confuse ecological debate (Thompson *et al*. 1995, Richardson *et al*. 2000, Daehler 2001, Davis and Thompson 2001, Rejmánek *et al*. 2002, Colautti and MacIsaac 2004). Surprisingly almost all authors used the word “invasive” to define "invasive species" (or "invasives") without a common definition.

The definitions given above differ in the ones, that include an element of harm and those who do not (Pys¡ek 1995, Daehler 2001). Richardson *et al.* (2000) remarked that there is a gradient of invasive species without any impact on their environment towards species with an increasing intensity of impact. He used the term “transformer species” proposed by Wells *et al.* (1986) for those invasive species, which ‘change the character, condition, form or nature of a natural ecosystem over a substantial area”. In the context of this thesis, we considered the definition that includes “spatial spread” and “harmful impact” of the elements associated with the concept of an “invasion” is more useful than one that does not.

1.3 Species invasiveness

The term invasiveness is used to describe species with inherent traits making them successful invaders. In other words invasiveness is traits that enable a species to invade a new habitat (Alpert *et al*. 2000). As Vitousek *et al.* (1995) clarified invasiveness is the establishment of self-sustaining plant populations that are expanding within a natural plant community with which they had not previously been associated".

Most introduced species do not find a tolerable environment nor an available ecological niche, hence do not survive (Lodge 1993). It is estimated that 10% of exotic species get established while only 1% become invasive (Groves 1991). So, only a fraction of all introduced species become invasive. What makes an introduced species a successful invader? Almost all successful invaders are opportunistic species characterized by a ruderal strategy (Grime 1979). It has been claimed that invasion success depends on inherent species' traits such as genotype, mechanism of seed dispersal, germination requirements, reproductive capability and their competitive ability (Baker 1965, Rejmanek and Richardson 1996, Molofsky *et al*. 1999).

1.4 Invasibility of ecosystems

Others maintain that exotic species become successful invaders because of the invasibility of the recipient ecosystem to invasion. Invasibility has been defined as a measure of an ecosystems' susceptibility to colonization by exotic species (modified from (Ewel 1986, Smallwood 1994)). When considering an invasive species from the angle of the invasibility of the recipient ecosystem one focuses on factors and site properties that make the ecosystem prone to invasion rather than traits of the species. Davis *et al.* (2000) formulated a number of testable predictions for plant invasibility, for example, that environments will be more susceptible to invasion during a period immediately following an abrupt increase in the rate of supply or a decline in the rate of uptake of a limiting resource. A better understanding of factors determining invasibility may offer a way to predict the spatial extent, rates and directions of spread of invaders.

1.5 Information requirements for invasive species management

Initially invasive species management aimed at eradication. Experience has learned however, that this is not achievable except for very few cases. Currently, therefore invasive species management aims at control of invaders and mitigation of their impact rather than eradication. Limitation of resources forces invasive species managers to carefully plan and prioritize interventions. Ideally, priority should be given to interventions, which optimize benefits in terms of damage reduction and durability and cost effectiveness. This could typically be realized in areas more severely affected by invaders. Information on the actual and potential distribution of invaders is considered crucial for their management (Reichard and Hamilton 1997, Rejmánek 2000). As Edward O. Wilson (Wilson 1997) has noted, "Extinction by habitat destruction is like death in an automobile accident: easy to see and assess. Extinction by the invasion of exotic species is like death by disease: gradual, insidious, requiring scientific methods to diagnose." Maps predicting the severity of the impact and damage could thus be used to localize areas requiring interventions most urgently. To this end, the development of advanced remote sensing and GIS technologies offers remarkable possibilities.

1.5.1 Remote sensing and Geographical Information System

Since Elton's time, ecologists have struggled to account for the distribution and spread of alien species. Remote sensing and GIS have successfully been applied to map the actual and potential distribution of invasive plant and animal species (McCormick 1999, Haltuch *et al*. 2000, Stow *et al*. 2000). Remote sensing applications appear to be restricted to detection of invasive species dominating the upper layer of the invaded community (McCormick 1999, Coulter *et al*. 2000, Haltuch *et al*. 2000). We however observed (Chapter 2) that 67% of the world’s 100 worst invasive species (ISSG 2004) do not dominate the ecosystem canopy, and hence can not be detected directly through remote sensing. Most of these cryptic invaders are small and go unnoticed or are hidden from remote sensing devices for instance when growing in the understorey of forest (Pysek and Prach 1995, Gerlach 1996, Chittibabu and Parthasarathy 2000). Indeed detection of forest understorey species using remote sensing has hardly been attempted (Jakubauskas and Price 1997).

Applying remote sensing to mapping cryptic invasive species is not straightforward since the captured spectral information will not be directly attributable to these species. Species spectra models can thus not be inverted to predict their distribution. Remote sensing thus has little to offer to cryptic invaders successful mapping because of their invasiveness. It could however offer far little explored possibilities in those cases where the invasibility of the recipient ecosystem determined the invaders’ success.

Such case might occur in degraded forests invaded by exotic species, which are adapted to increased light intensities associated with the opening of the forest canopy. Potentially, a combination of remote sensing, GIS and ecological knowledge could be used for mapping the distribution of understorey invasive species based on remotely sensed information on the degree of degradation of the forest overstorey. So far, such techniques have to our knowledge not been used to predict the distribution of invasive species.

1.5.2 Scientific challenge and objectives

We consider it scientifically challenging and desirable from a societal point of view to develop accurate and adequate means of mapping and modeling the distribution of cryptic invasive species over larger areas. The research proposed here aims at the development of remote sensing and geographical information techniques relevant for management of such cryptic invasive species.

The objectives of this thesis are:

- To review published approaches to mapping of invasive species, the application of remote sensing and GIS techniques in there, define the limitations of these approaches and identify possibilities for innovation of mapping and modeling invasive species distribution.
- To investigate the possibility of predicting from satellite remote sensing the forest canopy and light intensity penetrating through the canopy as indicators of the invasibility of forest ecosystems.
- To develop a technique to model spatial distribution of a cryptic forest understorey species *Chromolaena odorata* (L.) RM King & H Robinson in Nepal and its population ecology and to predict its reproductive success.
- To develop a model to predict the vegetative expansion of *C. odorata* based on knowledge of the influence of light intensity on the dynamics of its vegetative reproduction.
- To investigate the chain of causes that made lowland forest ecosystems in Nepal susceptible to invasion.
- To upscale local pattern to regional processes of ecosystems invasibility to localize areas or ecosystems most critically require interventions to control such invasive species.
- To make an integrated assessment of the complexity of deforestation and forest degradation for biological conservation in the tropical regions of Nepal.

1.6 Materials and method

1.6.1 The species: *Chromolaena odorata* (L.) R. M. King & H. Robinson and its biology

C. odorata (family Asteraceae, sub-family Lactucoideae tribe Eupatoriae; commonly known as Syam weed) is a Central American shrub that has become highly aggressive invasive throughout many tropical countries (McFadyen and Skarratt 1996). *C. odorata* was originally classified as *Eupatorium odoratum* L. (Syst. 10 (2):1905) but subsequent revision by King and Robinson (1970) resulted in the species being assigned to the genus *Chromolaena* DC (The name of the plant is based on Greek and refers to the bracts: *chroma* = colour and *chlaena* = cloak or blanket, also Latin *laena* = cloak). *C. odorata* is characterized by white, reddish or bluish flowers lacking ray florets. The species is distinguished by the three prominent veins in the leaves; rather consistent pattern of many rows of phyllaries (bracts) that are progressively longer and give a markedly cylindrical appearance to the head; straw-coloured, narrow-linear, angled bristly achenes having short white stiff cilia on the angles; and by the pungent odour emitted by the crushed leaves.

The worldwide range of expansion and the large economic and ecological impact of this species has been widely addressed (Epp 1987, De Rouw 1991, Gautier 1992b, Akpagana *et al*. 1993, McFadyen and Skarratt 1996, McWilliam 2000, Leslie and Spotila 2001). It has been therefore listed among the one hundred of the world's worst invasive species (ISSG 2004). *C. odo*rata is a clonally growing multi-stemmed erect, diffuse, scrambling heleophytic shrub grows up to 2m or higher; often forming dense thickets. Every clone consists of a root system with a corm (a modified stem to store food reserve under ground) supporting a number of shoots. *C. odorata* has a ruderal strategy and occurs in a wide range of environments including road verges, neglected agricultural fields and as an understorey species in forest. As a heliophyte (Gautier 1992a), it requires sufficient light to grow and produce seeds (Erasmus 1985, Witkowski and Wilson 2001).

1.7 Study area

1.7.1 Local scale

In order to represent the range of ecological conditions in Nepal, at local scale, a north-south cross section of central Nepal has been selected for this study. The area is located between 84° 15'– 85° 15'E longitude and 27° 00' – 27° 45'N latitude. This section is an eye view of vertical and horizontal (to some extant) biogeography of Nepal where east and west floristic region meets. It has an undulating extremely diverse hilly terrain with an altitude of below 100 to 2500m above mean sea level. It includes some parts of inner valleys of relatively flat plains, most parts of the east west running the *Churia* hills (first foot hills of massive Himalayan mountain chain) and some part of the *Mahabharat* (inner mountain range), representing most of the Nepalese ecosystems.

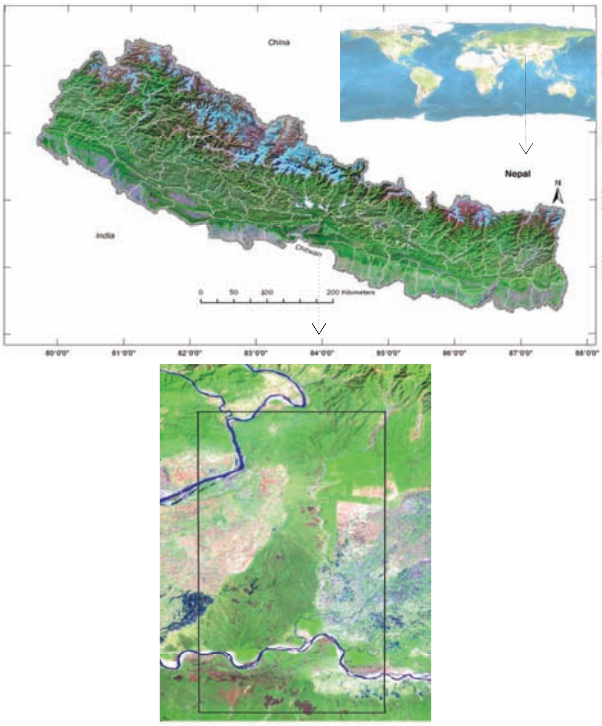

Figure 1-1. Map of the world (top); ETM+ mosaic image (band 7, 4, 2) of Nepal, the study area at national scale (middle) and ETM+ image (band 7, 4, 2) of part of the Chitwan district, the box showing the location of the study area at local scale (below).

1.7.2 National scale

Nepal (latitudes of 26° 22’ N and 30° 27’ N and longitudes of 80° 04’ E and 88° 12’) covers an area of 147, 181 sq. km, with a length of about 885 km, and an average width of 193 km (Figure 1-1). Although, it covers a small geographic area, it represents 118 ecosystems and is one of the world’s richest pockets in species diversity. Two-third of the area is occupied by hills and mountains that rises from the *Terai*, the northernmost part of Indo-Gangetic plain at about 59 m in the south to the crest of the Himalaya reaching upto 8848 m altitude in the north. The presence of extreme ranges of altitude has created a striking vertical zonation in climate, soil or land use within a small geographical area that makes it a paradise for many invasive plants including *C. odorata*.

1.8 Outline of the Thesis

This thesis represents a collection of research papers partly accepted or published in peer reviewed journals and international conference proceedings.

Chapter 1 provides a general introduction on invasion and invasive species and the application of GIS and remote sensing techniques in mapping and modeling the distribution of invasive species. It outlines shortcomings of current attempts in mapping invasive species. Next it states the research problems and objectives introduces the invasive shrub *C. odorata* and the study area.

Chapter 2 reviews the application of remote sensing and GIS in mapping actual and predicting potential distribution of invasive species. Distinction is made into four categories of invasive species based on whether they appear in and dominate the ecosystem canopy or not. The argument was that the possibility to apply remote sensing to map invaders differs between these categories. This review summarizes remote sensing techniques applied and outlines the potential of new remote sensing techniques. It also demonstrates that remote sensing has so far been applied predominantly to canopy dominant species. This contrasts with our findings revealing that the large majority of the world’s worst invasive species do not dominate the canopy. The mapping of these invaders received little attention so far. The chapter also reviews techniques used to map the risk of invasion for areas not invaded so far.

As we intended to map the distribution and biometry of *C. odorata* from the degree of openness of forest ecosystems, we first investigate the possibility of predicting forest canopy density from satellite imagery over large areas. In recent years a number of alternative methods have been proposed to predict forest canopy density from remotely sensed data. However, it remains difficult to decide which method to use, since their relative performance has never been evaluated. In **Chapter 3**, the performance of 1) an artificial neural network, 2) a multiple linear regression, 3) the forest canopy density mapper and 4) a maximum likelihood classification method was compared for prediction of forest canopy density using a Landsat ETM+ image. Comparison of confusion matrices revealed that the regression model performed significantly worse than the three other methods. As a continuous variable, about 89% of the variance of the observed canopy density was explained by the artificial neural network, which outperformed the other three methods in this respect. Moreover, the artificial neural network gave an unbiased prediction, while other methods systematically under or over predicted forest canopy density.

Remote sensing has so far been applied predominantly to canopy dominant species. The mapping of forest understorey invaders has received little attention so far. In **Chapter 4,** possibilities of not only mapping the presence and absence but also seed productivity of such invader were revealed by applying canopy density mapping classifier in combination with GIS expert knowledge, biophysical factors and field data. The relation between light intensity, forest canopy density and other environmental variables and the cover and seed production of *C. odorata* was established and these models were inverted to predict its seed productivity. Light intensity determined 93% of the variation in $\log_{10}$ seed production per plant. *C. odorata* failed to produce seed below a light intensity of 6.5 mJ m^{-2} day^{-1}. Further analysis revealed that *C. odorata* was absent above this light intensity in case of a high biomass of other shrub and herb species, a situation occurring in the absence of grazing. We therefore suggest that other species control *C. odorata* through competitive exclusion in the absence of grazing, whereas grazing breaks the dominance of these other species thus creating the conditions for *C. odorata* to rise to dominance. Predicted *C. odorata* cover and seed production per plant were combined into a map displaying an index of the seed production per unit area. Such map displaying seed producing sites could be used to significantly reduce the costs of controlling *C. odorata* infestation.

Plant invaders establish through dispersal of propagules, typically in the form of seeds. Once established, they may further increase their local dominance by vegetative regeneration. Vegetative regeneration in the short-lived perennial *C. odorata* is clonal. The corm stores food reserve under full light conditions and increases in size over time. In **Chapter 5**, we investigated how vegetative spread of

C. odorata was determined by light intensity. We first described the presence of year rings in the corms of *C. odorata*. These “corm rings” allowed us to estimate the age of individual plants. We next sampled a large number of plants under a range of light conditions. We demonstrated that the increase of corms biomass over time was controlled by light intensity. Corm biomass increased with plant age in open and medium open forest, but not in closed forest. We next demonstrate that the size of the plants as expressed by the number of shoots depended on the biomass of the corm system. Hence there appears to be a positive feed back under high light conditions more resources are allocated to corms, allowing next season to enlarge the photosynthetic apparatus, which subsequently allows allocation of more resources to the corms. Light intensity apparently controls this feed back and the possibility of *C. odorata* to develop vegetatively. We finally predict from forest canopy density areas where *C. odorata* populations are expected to expand vegetatively.

“What made this exotic species a successful invader?” is a fundamental question in managing invasive species. Was it the intrinsic ability of species to invade or the susceptibility of the ecosystem to invasion? **Chapter 6** addresses these questions for the invasion of *C. odorata* in the Nepalese *Terai*. In Nepal, malaria control increased human population growth rates to levels exceeding 10% per year in the 1960’s. This resulted in land use change and scarcity of grazing lands outside forests. The increasing demand for grazing and wood resources resulted in an opening up of the forest canopy. It is deduced that the pressure to exploit the forests for grazing increased over time as a result of land use change and increased human population density. Our results indicate that the neo-tropical shrub *C. odorata* maintains invasive populations at canopy densities below 60%, which prevail in forests with opened canopies. Hence, *C. odorata* invaded because canopy opening made the environment suitable for it. Herbarium record revealed that *C. odorata* invaded southern Nepal shortly after the initiation of malaria control and degradation of the forests. We therefore argue that the degradation and opening of the canopy of the low land Nepalese forests facilitated and accelerated *C. odorata* invasion which was caused by change in land use and demography triggered by malaria control.

Chapter 7 we presented the expert, empirical expert and principal component analysis based model to predict presence and absence of *C. odorata* in Nepal. Selection of explanatory variables was largely based on our understanding of species auticology and empirical observation. High resolution interpolated climate surfaces derived from the SRTM and a combination of altitude, precipitation and length of growing season for *C. odorata* allowed us to predict its distribution with 80% accuracy. While the principal component analysis could predict it's distribution with 86% accuracy. Furthermore, very high resolution interpolated surfaces data are more robust in delineating climatic variability within which *C. odorata* could thrive. Furthermore, coarse scale for species prediction is not appropriate for country similar to Nepal where topographic and climatic variation is high because course scale can not deal with all response shape.

Chapter 8 assess the spatial extent and severity of deforestation and forest degradation processes in the *Terai* district of Nepal using field survey records, existing vegetation maps, and remote sensing data to determine the different degrees of deforestation and forest degradation. Our results showed that 66% of the total forest area in *Terai* region had canopy cover below 60% thus confirming widespread degradation. More surprisingly fifty percent of the forested area inside protected areas had similar open canopies. This study demonstrates the effectiveness of remote sensing in retrieving and analyzing of such data, in particular for detection, extrapolation, and interpretation, area calculation, upscaling and monitoring to make geo-referenced information easily accessible to non-GIS users. This study provides useful information for improving planning strategies for sustainable forest management. The forest degradation maps presented in this article could serve as a start to prioritize such interventions

Finally, **Chapter 9** synthesizes the findings and conclusions of previous chapters and discusses the research approach applied in mapping, modeling and managing cryptic invasive species. It summarizes what we have learned and proposes a plan for dealing with invasibility of tropical ecosystems in Nepal. It outlines a road leading managers and restoration ecologists towards the restored biotic world we desire and deserve.

CHAPTER TWO

Remote sensing and GIS applications for mapping and spatial modeling of invasive species

This Chapter is Based on:

C. Joshi, De Leeuw, J., and van Duren, I. C. 2004. Application of remote sensing and Geographical Information System (GIS) technologies in mapping and modeling biological invaders. Proceedings of the XXth ISPRS Congress: International Archives of the Photogrammetry, Remote Sensing and Spatial Informational Sciences. Vol XXXV, Part B7,Istanbul 2004, 669-677.

2 Review: Remote sensing and GIS applications for mapping and spatial modeling of invasive species

Abstract

Biological invasions form a major threat to the sustainable provision of ecosystem products and services, both in man-made and natural ecosystems. Increasingly, efforts are made to avoid invasions or eradicate or control established invaders. It has long been recognized that remote sensing (RS) and geographical information system (GIS) could contribute to this, for instance through mapping actual invader distribution or areas at risk of invasion. Potentially GIS could also be used as a synthesising tool for management of interventions aiming at invasive species control. This chapter reviews the application of remote sensing and GIS in mapping the actual and predicting the potential distribution of invasive species. Distinction is made into four categories of invasive species based on whether they appear in and dominate the ecosystem canopy or not. We argue that the possibility to apply RS to map invaders differs between these categories. Our review summarizes RS techniques applied in here and outlines the potential of new RS techniques. It also demonstrates that RS has so far been applied predominantly to canopy dominant species. This contrasts with ecological databases revealing that the large majority of invasive species do not dominate the canopy. The mapping of these invaders received little attention so far. In this chapter, we reviewed various possibilities to map non-canopy invader species. The chapter also reviews techniques used to map the risk of invasion for areas not invaded so far.

Key words: Review, Remote sensing, GIS, Invasive species, Classification, Mapping techniques, Scale, Sensors, Challenges

2.1 Biological invaders

Invasive species are a current focus of interest of ecologists, biological conservationists and natural resources managers due to their rapid spread, threat to biodiversity and damage to ecosystems. Invasions may alter hydrology, nutrient accumulation and cycling, and carbon sequestration on grasslands (Polley *et al*. 1997). The global extent and rapid increase in invasive species is homogenising the world's flora and fauna (Mooney and Hobbs 2000) and is recognized as a primary cause of global biodiversity loss (Czech and Krausman 1997, Wilcove and Chen 1998). Bio-invasion may be considered as a significant component on global change and one of the major causes of species extinction (Drake *et al*. 1989).

This chapter attempts to provide a review of several studies that assess the utility of remote sensing (RS), or remote sensing coupled with geographical information system (GIS), in mapping and modeling the distribution of invasive species. The term invasive species is also more or less synonymously referred to as aliens, barriers, naturalized species, invaders, pests, colonisers, weeds, immigrants, exotics, neophytes, introduced species or transformers (Heywood 1989, Richardson *et al*. 2000). These terms come from studies having different view points on the problem but in the context of this paper they should be considered as similar.

2.2 Application of RS and GIS techniques in mapping biological invasions

Remote sensing technology has received considerable interest in the field of biological invasion in the recent years. It is a tool offering well-documented advantages including a synoptic view, multi-spectral and multi-temporal coverage data since 1960s (Pearson *et al*. 1976, Lillesand and Kiefer 1994). It is now widely applied on collecting and processing data. It has proved to be a practical approach to study complex geographic terrain types and diverse inaccessible ecosystems. It provides a wide range of sensor systems including aerial photographs, airborne multi-spectral scanners, satellite imagery, low and high spatial and spectral resolution and ground based spectrometer measurements.

Remote sensing technology has many attributes that would be beneficial to detecting, mapping and monitoring invaders. Spatial heterogeneity complicates the study of seasonal and long-term trends of biological invasion. Remote sensing, however, with its broad view has the potential to deliver the relevant information. Satellite imagery is available for most of the world since 1972 (Lillesand and Kiefer 1994). The multi-date nature of satellite imagery permits monitoring dynamic features of landscape and thus provides a means to detect major land cover changes and quantify the rates of change.

Integrated GIS and remote sensing have already successfully been applied to map the distribution of several plant and animal species, their ecosystems, landscapes, bio-climatic conditions and factors facilitating invasions (McCormick 1999, Rowlinson *et al*. 1999, Haltuch *et al*. 2000, Los *et al*. 2002). An increasing number of publications (Figure 2-1) is dealing with the application of remote sensing and GIS in the data collection and analysis of invasive animal and plant species, their abundance, distribution, mapping, modeling and factors influencing their distribution.

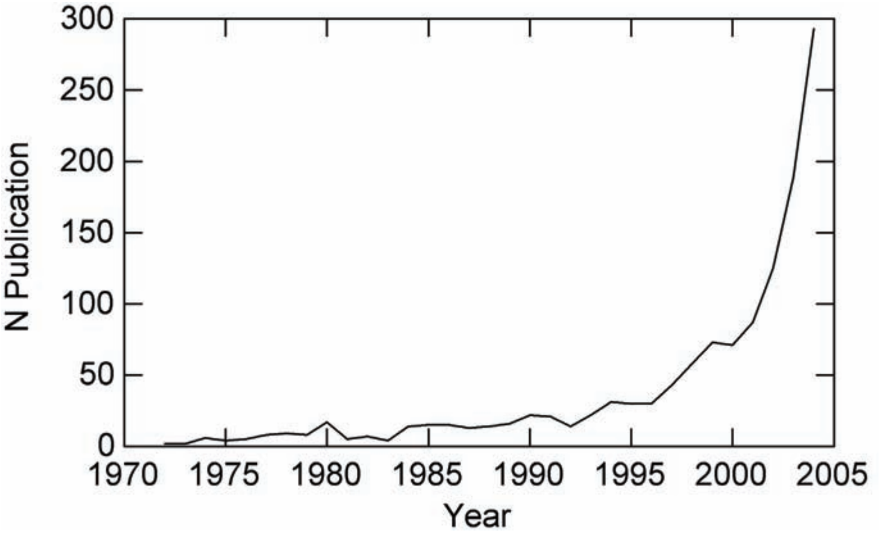

Figure 2-1. Publications on application of remote sensing and GIS techniques in mapping invasive species.

Mapping the type and extent of bio-invasions, the impact of invasions or potential risks of invasions requires accurate assessment and modeling of species distributions. So far no synoptic literature review has been published in the field of mapping invasive species. To sketch the possibilities, limitations and challenges of remote sensing techniques in mitigation of invasive species, this paper provides an overview of the application of remote sensing and GIS technologies in mapping biological invasions. We addressed the following questions.

- What mapping techniques have been used to map and predict the potential distribution of invasive species?

- What sensors and what image processing and classification techniques have been used to map the actual distribution of invasive species?

- For what species groups (canopy versus non-canopy members, plant versus animal species) has successful mapping been reported? Is there any evidence that the reported successful applications tend to be biased towards any particular species groups?

- To what extent has sensitivity to scale and the reliability of the mapping product been addressed?

- Which available mapping techniques so far not applied to invaders could be used to improve the mapping of invasive species?

We searched for articles on biological invasions using several electronic databases (AgECONCD, GEOBASE and SOILCD), covering international agricultural, economical and rural development literature, Journals, monographs, conferences, books and annual reports. We also searched other sources such as scientific abstracts, worldwide web, CD ROMs and libraries within and outside the Netherlands. Several experts were contacted who provided additional references.

2.3 Mapping actual and potential distribution: from global to local scale

At national or continental level, maps of invasive species distribution are mainly interpolations from recorded observations compiled and stored in herbaria, zoological collections and research institutes. Maps are often generated by manually drawing polygons (boundaries) around areas where the species is known to occur or alternatively using some automated interpolation procedures. For example the distribution map of *Chromolaena odorata*, one of the world's worst invaders is displayed in Figure 2-2. The map displays the distribution as a continuous surface. This suggests that the species occurs throughout the area represented by the map polygons. In reality, species are not homogeneously distributed across their distribution range. Instead they prevail in certain environments while they are absent from others. Maps showing discontinuous patches would more realistically represent such a distribution pattern. However, at small scale (typically <1 to a million) we prefer to use interpolations, while realizing that they are generalizations, displaying the broad geographic range within which the species is known to occur.

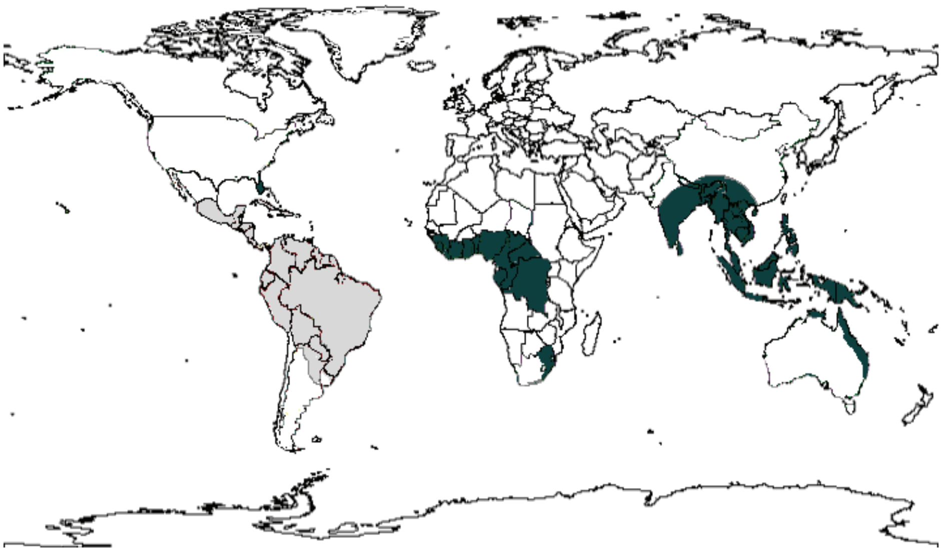

Figure 2-2. The world distribution of *Chromolaena odorata*: area under infestation (Black) and native range (Grey). Map based on global invasive species database (ISSG 2004).

The need to display the discontinuity and patchiness in distribution patterns emerges while moving towards larger scales. Here, it would be impractical to derive maps through interpolation, because it would require sampling every patch, a costly operation particularly when larger areas are to be mapped.

2.4 Remote sensing techniques

Digital camera photography and videography are recently introduced as cheap, easily available and flexible alternatives to standard photography, particularly when the data are to be transferred onto a computer system. There are systems available that cover the near infrared (NIR) and infrared (IR) as well. Multispectral scanners register reflectance in a number of spectral bands throughout the visible, near- to far-infrared portions of the electromagnetic spectrum. Broad-band scanners have few spectral bands of one hundred or more nm wide. Hyperspectral scanners have more (tens up to several hundreds) but narrower (from tens to a few nm wide) spectral bands. Broad-band scanners have been successfully applied to discriminate between broad land cover types such as forest versus bare soil and built up area. The higher spectral resolution of hyperspectral scanners allows discrimination of more subtle differences such as those between individual species (Figure 2-3).

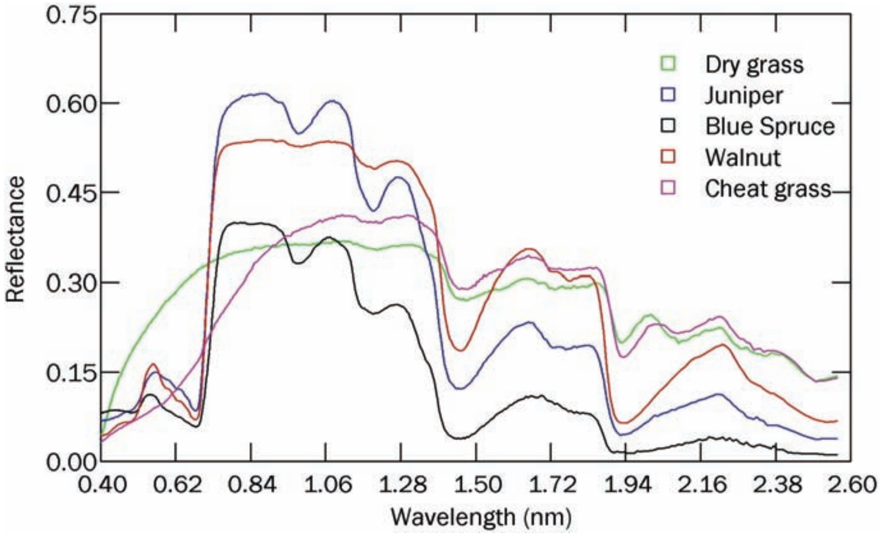

Figure 2-3. Electromagnetic spectrum and spectral reflectance profiles for different species (adopted from the spectral library of the Environment for Visualizing Images software (ENVI 2003).

2.5 Classification of invasive species

The data captured by remote sensing devices will be most directly related to the properties of that canopy. We introduced a classification of species based on their remotely sensed canopy reflectance response (Figure 2-4). It is the canopy of an ecosystem (be it vegetation or fauna) that reflects the electro-magnetic radiation that is captured by remote sensing devices.

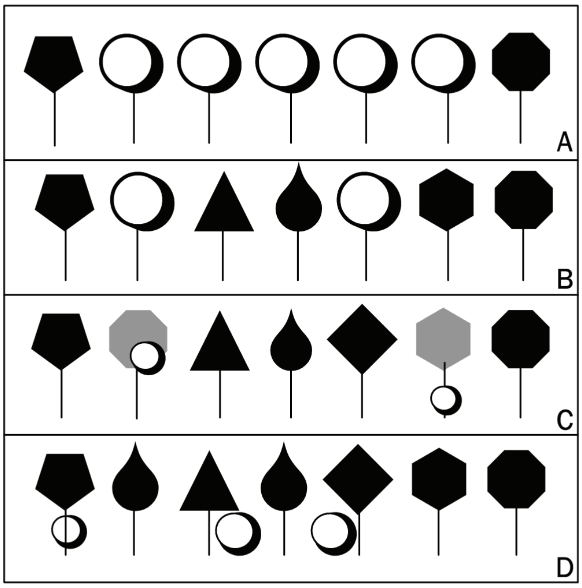

Figure 2-4. Application of remote sensing in detecting individual invasive species (may be an animal or plant) as represented in white colour object. Class I: Canopy dominant species (A), class II: Mixed canopy dominant species (B), class III: Pseudo-canopy dominant species (C) and class IV: Understorey species (D).

Class I includes species dominating the canopy and forming homogeneous single species stands. Class II includes species that are members of a multi species canopy and directly reflects electro-magnetic radiation. Class III includes species not reflecting, but influencing the reflective properties of canopy members belonging in class II and I. Class IV finally includes all species that neither dominate canopy nor influence the reflective properties of other species in class I and II.

2.5.1 Canopy dominant species

Several invasive species dominate the canopy of the earth surface forming homogeneous single species stands that extend over larger areas. Included are a large number of tree species such as *Melaleuca quinquenervia.* Canopy dominance among invaders is not restricted to tree species, it also occurs in grasses (e.g. *Arundo donax, Spartina anglica*), floating water hyacinth (*Eichhornia crassipes*) and submerged aquatic vegetation (*Caulerpa taxifolia, Undaria pinnatifida, Oscillatoria* sp.) and among colonial animals such as zebra mussels (*Dreissena polymorpha*).

From aerial photography and videography to multispectral and multitemporal scanners have been used for identifying and mapping invasive species (Venugopal

1998, Shepherd and Dymond 2000, Everitt *et al*. 2001a). For instance, detection of invasive *Prosopis glandulosa* var. *torreyana* using TM images (Harding and Bate 1991), *Gutierrezia sarothrae* with NOAA-10 low resolution spectral image (Peters *et al*. 1992), *Kalmia angustifolia* (Franklin 1994) and *Arundo donax* using high spatial resolution (~4m) AVIRIS data (Ustin *et al*. 2002), *Cynodon dactylon* with aerial video and colour-IR photographs (Everitt and Nixon 1985), *Populus tremuloides* clones using hand-held video (Stohlgren *et al*. 2000) are some of the examples of mapping canopy dominating species.

Some of the reported invasive species dominate submerged aquatic ecosystems. For those ecosystems, remote sensing methods described so far, are limited, because little light is reflected back by submerged organisms. Budd *et al.* (2001) used Advanced Very High Resolution Radiometer (AVHRR) remote sensing reflectance imagery and found a significant relationship between reflectance before and after *Dreissena polymorpha* invasion. Hill *et al.* (1998) modelled the propagation of the green alga *Caulerpa taxifolia* and predicted the local pattern of expansion, increase of biomass and covered surfaces, and invasive behaviour. Gross *et al.* (1988) estimated biomass of the *Spartina alterniflora* using a hand-held fixed band radiometer configured to collect data in Landsat TM. Welch *et al.* (1988) related 13 invasive macrophytes distributions (including *Hydrilla verticillata*, *Potamogeton* sp., and *Lemna perpusilla*) to environmental factors influencing aquatic plant growth using bathymetry and herbicide applications maps and statistical data on nutrients, dissolved oxygen, biological oxygen demand, and turbidity into a PC-based GIS. A significant change was found in the ratio of emergents to subemergents as well as the total area of aquatic macrophytes.

2.5.2 Mixed canopy dominant species

Plant characteristics such as life form, leaves, flowers etc determine reflectance. If a species is dominant enough in the canopy and characteristics can be distinguished from other species, than it is possible to detect such individual species based on spectral reflectance. The ability of high spectral and spatial resolution sensors to discriminate between invasive and native species depends on intra-specific variability in spectral reflectance. Everitt & Nixon (1985) demonstrated that a family of spectra can represent a particular species, and invasive species are easily separated using low altitude aerial photographs or field spectrographs. For example, Menges *et al.* (1985) found colour IR (CIR) aerial photography to be useful for detecting *Sarcostemma cyanchoides*; *Parthenium hysterophorus; Sorghum halepense; Sisymbrium irio* and *Amaranthus palmeri* in different crops. Young *et al.*

(1976) detected growth timing *of Chrysothamnus viscidiflorus* using colour photography. Feyaerts & van Gool (2001) proposed an online system that distinguishes crop from weeds based on multispectral reflectance gathered with an imaging spectrograph.

2.5.3 Pseudo-canopy dominant species

Numerous investigators have worked on developing techniques for using multispectral data in invasive species mapping and detection (Eav *et al*. 1984, Medlin *et al*. 2000, Vrindts *et al*. 2002, Zhang *et al*. 2002b). Analysis of hyperspectral data has produced encouraging results in the discrimination of healthy and infected canopy dominant species. For example, Fouche (1995) identified rootrot-infested cashew nut trees, *Phytophthora cinnamomi* infestation in avocado orchards and infected citrus trees. They could be differentiated from their healthy neighbours, using low-altitude aerial colour infrared (CIR) imagery. Gebhardt (1986) used IR measurements of crop canopy temperature to detect differences in water supply and nematode infestation. Smirnov & Kotova (1994) monitored the infection by *Heterobasidion annosum* in areas with pollution levels exceeding 15 Ci/km2 after the Chernobyl nuclear disaster in Russia. Kharuk *et al.* (2001) analysed large-scale outbreak of the *Dendrolimus sibiricus* in the forests using NOAA/AVHRR imagery and found that the imagery could be used for detecting dying and dead trees. Using principal component and cluster analyses Zhang *et al.* (2002b) observed that the sensitive spectral wavelengths and reflectance values could discriminate *Phytophthora infestants* infection on tomatoes.

Performing spatial correlations, GIS tools often does identification of invaders influencing canopy dominant species. For example, Kazmi & Usery (2001) monitored vector-borne diseases, Bell (1995) detected grape phylloxera spread and Terry & Edwards (1989) analysed the effect of insecticides and parasites released for invasive species control.

2.5.4 Understorey species

Few researchers have pointed out the possibilities of application of remote sensing in studying forest understorey invaders. Plant species such as *Chromolaena odorata*, *Ulex europaeus, Clidemia hirta, Lantana camara, Mimosa pigra, Psidium cattleianum, Rubus ellipticus, Schinus terebinthifolius* and most of the invasive animal species are examples of this category. Most of the invasive animals, lower flora, herbs, shrubs and fauna are found to be understorey vegetation, making detection using direct remote sensing techniques almost impossible. Nevertheless, a combination of remote sensing techniques, GIS and expert knowledge still offer potential to detect understorey invasion through development of models and risk maps. May *et al.* (2000) quantified remotely sensed airborne data into physical and ecological variables, obtaining an improved spatial and temporal representation of the dynamics of native and exotic plant communities. Hence, such information could potentially be used in mapping non-canopy dominant invasive species as well as predicting the probability of actual and potential sites and areas where environmental conditions are susceptible to infestation.

2.6 Monitoring and prediction of invasion risk

Predicting the probability of biological invasion and probable invaders has been a long-standing goal of ecologists. A major challenge of invasion biology lies in the development of pre and post predictive models and understanding of the invasion processes. Introduced species vary in their invasive behaviour in different regions (Krueger *et al*. 1998). Prediction is more difficult than finding an explanation. Predicting the ecological behaviour of a species in a new environment may be effectively difficult, therefore attempts at predicting invasions have generally been unsatisfactory (Williamson 1999). Furthermore, estimating animal species numbers, population size and related features is rather difficult in comparison to plants. However, Kolar & Lodge (2001) indicated clear relationships between the characteristics of releases and the species involved, and the successful establishment and spread of invaders. Allen & Kupfer (2000) developed a modified change vector analysis (CVA) using normalized multi-date data from Landsat TM and examined *Adelges piceae* infestation. Luther *et al.* (1997) pointed out the importance of logistic regression techniques to develop models for predicting forest susceptibility and vulnerability and to assess the accuracy of forecasts. Using an integrated multimedia approach in the vegetation database for invasive species provides a unique way to represent geographic features and associated information on interrelationships between flora, fauna, and human activities (Hu 1999). Predictions of malaria risk mapping (Kleinschmidt *et al*. 2000) and microbiological risk assessment for drinking water (Gale 2001) are some examples of risk mapping and prediction that have been done in the field of biological invasions.

Applications of these promising quantitative approaches in an Integrated GIS environment may allow us to predict patterns of invading species more successfully. Although remote sensing and GIS technology provides particularly valuable insight into the effectiveness of an ongoing invasive species control programme, the cost of monitoring should be within the affordable range as pointed by Benton & Newnam (1976).

2.7 Issues of scale and accuracy

Scale is one of the central issues in invasion ecology. All observations depend upon the spatial scale, size of the study area investigated and resolution of the remote sensor. Habitat evaluation of a species is influenced strongly by spatial scale (Cogan 2002). The variations in the landscape patterns are scale-dependent (Rescia *et al*. 1997). Unfortunately there is no “correct” scale; it depends on survey purpose (Trani 2002). Hence, in most of the cases, landscape scale is used as an appropriate scale for modeling.

McCormick (1999) reported 91 % accuracy in mapping *Melaleuca quinquenervia* and pointed out the superiority of large scale of colour infrared aerial photographs for identifying, mapping and effective management of exotic species. Carson *et al.* (1995) found that the Landsat TM and SPOT data with ground resolution of 30 and 20 meters respectively, are not considered useful for mapping at species level, unless the stand of an invasive species is large enough. Multi-date imagery therefore appears to improve mapping and modeling the infestation pattern of canopy dominant species (Mast *et al*. 1997, Hessburg *et al*. 2000). Bren (Bren 1992) examined the invasion of *Eucalyptus camaldulensis* into an extensive, natural grassland in a high flood frequency site using 45 years time series aerial photographs (taken in 1945, 1957, 1970, and 1985) extrapolated model showed the almost complete extinction of extensive grass plains. Hessburg *et al.* (2000) used aerial photo (from 1932 to 93) of interior northwest forests, USA and found emergent non-native herb lands. Mast *et al.* (1997) provided a quantitative description of the *Pinus ponderosa* tree invasion process at a landscape scale using historical aerial photography, image processing and GIS approaches. Welch *et al.* (1988) applied GIS to analyse aerial photo for monitoring the growth and distribution of 13 aquatic emergent, submergent and free-floating species. They produced vegetation maps using large scale (1:8000-1:12000) color infrared aerial photographs of different years (1972, ’76, ’83, ’84, ’85).

Current developments in sensor technology have the potential to enable improved accuracy in the mapping of vegetation and its productivity. Rowlinson *et al.* (1999) indicated that using manual techniques to identify infested riparian vegetation from 1:10,000 scale black and white aerial photographs yielded the most accurate and cost-effective results. The least accurate data sources for this purpose were aerial videography and Landsat thematic mapper (TM) satellite imagery. High spatial (less or equal to 1m) but low spectral resolution remote sensing data appeared to be useful in mapping invasive Chinese tallow trees with an accuracy of greater than 95 percent (Ramsey *et al.* 2002). Medlin *et al.* (2000) detected infestations of *Senna obtusifolia, Ipomoea lacunosa,* and *Solanum carolinense* with at least 75% accuracy using multi-spectral digital images. Vrindts *et al.* (2002) distinguished seven weed species with a more than 97% correct classification using a limited number of wavelength band ratios. Everitt *et al.* (2001b) noted that *Juniperus pinchotii* had lower visible and higher near-infrared (NIR) reflectance than associated species and mixtures of species allowing a mapping accuracy of 100 percent. Lass *et al.* (2000) tested accuracy of detection of a homogenous population of *Centaurea solstitialis* at different spatial resolution. Their result showed a low commission and omission errors with 0.5m spatial resolution than 4.0m. Very-high spatial resolution (0.5 m) colour infrared (CIR) digital image data from colour-infrared digital camera imagery showed potential for discriminating Acacia species from native fynbos vegetation, other alien vegetation and bare ground (Stow *et al*. 2000). In cases where different spatial resolutions resulted in equal detection accuracy, the larger spatial resolution was selected due to lower costs of acquiring and processing the data.

All these studies noted the importance of image resolution, spectral characteristics, superiority of lower scaled aerial photographs and images. It also shows clearly that for accurate mapping of invasive species it is important to consider the phenological stage in aiming of taking the aerial photographs or images. Although high spectral and spatial resolution provide the ability to classify canopy dominant species, precise classification of a species is still difficult. Several such studies of the spectral properties of invasive species have been derived, mostly from low altitude aerial photography or field spectrographs. However, the information reaching the remote observer will be minimum.

Other factors for instance, atmospheric noise, humidity, shadow, contribution from soil may add to the confusion and the chance of discrimination of separate species low. Furthermore, variation in orientation of leaves, age of a leaf, variation in leaf area index, different slopes of the locations where the individuals are found could make the spectral signature of a species difficult to define. It is not however practically feasible to determine the ideal wavelengths for discrimination when large numbers of invasive species are present. Furthermore, if the presence of number of invasive species per pixel increases, the difficulty in identifying the individual components that contribute to the mixed spectrum also increases. These problems will be further aggravated if species variability in spectral signatures is high. For large scale direct remotely sensed monitoring of several invasive species, the possibility of correctly identifying all individuals through direct mapping thus appears doubtful.

2.8 Summary and conclusions

In this article, we attempted to evaluate the potential of remote sensing and GIS techniques for the critical task of invasion mapping. Although the use of RS and GIS techniques for mapping invasive species and invaded ecosystems is increasing rapidly, the literature on means and methods for invasive species mapping remains scattered and often contradictory. Most of the IUCN's worst invasive species fall under our class IV species, in which straightforward application of remote sensing is almost impossible. Recent remote sensing and GIS applications on detecting invasive species were mainly dealing with species belonging to class I (Figure 2-5).

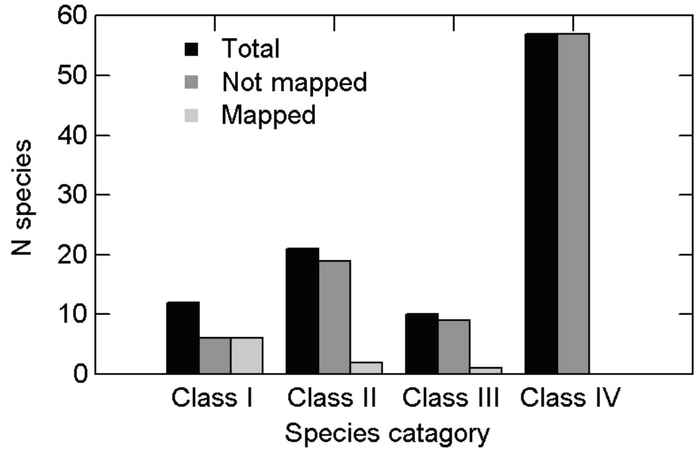

Figure 2-5. Mapped and not mapped 100 world's worst invasive species included in the list of International Union for Conservation of Nature and Natural Resources. Class I: Canopy dominant species, II: Mixed canopy dominant species, III: Pseudo-canopy dominant species and IV: Ecosystem understorey species.

For instance, most of the understorey species that have been declared as the world's worst invaders by the ecologists have not caught the attention of remote sensing experts. In the same way species such as *Melaleuca quinquenervia* or *Tamarix ramosissima*, which dominate entire ecosystems forming a monotypic dense canopy, do not necessarily need the use of high spectral resolution imagery and vice versa. It is not clear whether RS and GIS techniques will prove equally strong for mapping mobile invasive species such as *Acridotheres tristis* (Bird), *Aedes albopictus* (mosquito) or *Boiga irregularis* (snake). This needs further testing in the near future.

The status of many exotic species with respects to their invasiveness is not well documented. Therefore, the ability of remote sensing and GIS techniques to monitor changes in different ecosystems may be crucial if the effect and the cause of rarity are to be assessed. Cases of actual applications are still not much more than the traditional investigations. Rapidly shifting interest in remote sensing and IGIS of bio-invasion mapping has resulted in the development of a diverse range of mapping techniques. However, the technology needs further development in terms of real world applications in the mapping of invasive species. Moreover, mapping, modeling and predicting biological invasion will still be a major challenge for ecologists because the biological processes involved are very complex. This complexity makes it difficult to retrieve or delineate invasions that occur in diverse ecosystems. As Specter and Gayle (1990) pointed out the proliferation of new technologies does not guarantee their application to real world problems.

Although restricted to few taxa, studies revealed the potential of remote sensing and GIS application in mapping and modeling invasive species. There are possibilities of generating in-depth information in detecting, mapping and analyzing the impact of invasion on an area or entire ecosystem and species level properties. To enhance the result of invasion mapping, there is a clear need of combined use of remote sensing, GIS and expert knowledge. Management dealing with invasive species requires accurate mapping and modeling techniques at relative low costs. Development of those will be a valuable step towards conservation of native biodiversity.

2.9 Future research

The increasing number of sensors and classification algorithm have provided spatial ecologists with tremendous opportunities to advance the application of RS and GIS techniques in mapping and modeling the distribution of invasive species. However, application of remote sensing is strongly limited when dealing with world's worst forest understorey plant species. Direct detection of such species using remote sensing tool are hardly explained (Jakubauskas and Price 1997).

Understanding of distribution pattern of understorey species may only be possible when interrelation between ecological complexities of forest overstorey and understorey species could be established. Consequently, mapping forest canopy openings could be a potential factor to predict the distribution of forest understorey heliophytic invaders. Although, the literature review revealed a wide variety of proposed methods (Boyd *et al*. 2002), there is little indication which method would be superior in performance in comparison to others, because very few of the reported methods have been validated for their accuracy, and so far no comparative studies have been undertaken. We argue that high accuracy in forest canopy classification may holds particular promise since growth of forest understorey heliophytic invasive species depends on light intensity reaching to the forest floor, which is regulated by the canopy itself.

CHAPTER THREE

Remotely sensed estimation of forest canopy density: a comparison of the performance of four methods

This Chapter is based on:

C. Joshi, J. De Leeuw, A. K. Skidmore, I. C. Van Duren and H. van Oosten 2006. Remotely sensed estimation of forest canopy density: a comparison of the performance of four methods. International Journal of Geoinformatics and Earth Observation, (8) 2: 84-95.

3 Remotely sensed estimation of forest canopy density: a comparison of the performance of four methods

Abstract

In recent years, a number of alternative methods have been proposed to predict forest canopy density from remotely sensed data. To date however, it remains difficult to decide which method to use, since their relative performance has never been evaluated. In this study, the performance of 1) an artificial neural network, 2) a multiple linear regression, 3) the forest canopy density mapper and 4) a maximum likelihood classification method was compared for prediction of forest canopy density using a Landsat ETM+ image. Comparison of confusion matrices revealed that the regression model performed significantly worse than the three other methods. These results were based on a z-test for comparison of weighted Kappa statistics, which is an appropriate statistic for analysis of ranked categories. About 89% of the variance of the observed canopy density was explained by the artificial neural networks, which outperformed the other three methods in this respect. Moreover, the artificial neural networks gave an unbiased prediction, while other methods systematically under or over predicted forest canopy density. The choice of biased method could have a high impact on canopy density inventories.

Key words: Forest canopy density, Classification, Remote sensing, Neural networks, Weighted Kappa, Bias prediction.

3.1 Mapping forest canopy density

The distribution and rate of decline of tropical forests has successfully been assessed using remote sensing images (Myers 1980, Prince 1987, Ringrose and Matheson 1987, Ford and Casey 1988, Iverson *et al*. 1989, Sussman *et al*. 1996). About 13.5 million hectares of tropical forest is cleared annually for agriculture, pastures, timber products and infrastructure development (FAO 2001, Geist and Lambin 2002). These figures only consider complete clearing and do not include forest degradation, a more gradual process, which (Tanner *et al*. 1996) considered relevant to account for as well.

The world-wide extent of forest degradation is unknown. Forest degradation is characterized by a reduction in forest quality and biomass by an opening up of the canopy. This is particularly so in the tropics, where local population exploit forest resources.

Forest canopy density expressing the stocking status constitutes the single major stand physiognomic characteristics of the forest. The degree of forest canopy density is expressed in percentages. Remotely sensed estimates of forest density have been used to indicate the degradation status of forest (Tiwari *et al*. 1986, Prince 1987, Ringrose and Matheson 1987, Ford and Casey 1988). There are a variety of empirical approaches that can be used to map forest canopy: 1) object based classification (Dorren *et al*. 2003), 2) artificial neural network (Boyd *et al*. 2002), 3) linear regression (Iverson *et al*. 1989, Zhu and Evans 1994, DeFries *et al*. 1997, Levesque and King 2003), 4) decision tree classification (Souza Jr. *et al*. 2003), and 5) forest canopy density mapper (Rikimaru 1996, Rikimaru and Miyatake 1997). Hence, the literature review reveals a wide variety of proposed methods. So far, there is little indication which method would be superior in performance in comparison to others, because very few of the reported methods have been validated for their accuracy, and so far, no comparative studies have been undertaken.

Typically, error matrices have been used to assess the performance of methods to estimate canopy density (Congalton and Green 1999, Oetter *et al*. 2000, Souza Jr. *et al*. 2003, Peddle *et al*. 2004). Error matrices are appropriate to reveal the performance of classifications resulting in categories. In the case of canopy density, predictions have been made in the form of ranked categories (Rikimaru and Miyatake 1997, Saatchi *et al*. 1997, Scarth and Phinn 2000). Forest canopy density, however, is a continuous variable by nature. Hence, it would be more appropriate to predict forest canopy density in a continuous variable, rather than categories, which allows to assess the performance of predictions in terms of random and systematic error components.

The main objective of this study is to compare the performance of different methods, namely 1) artificial neural networks (ANN), 2) multiple linear regression technique (MLR), 3) a semi-expert deductive approach i.e. forest canopy density mapper (FCD) and 4) maximum likelihood classification (MLC) to predict forest canopy density in percent as a continuous variable.

3.2 Material and methods

3.2.1 Study area

The field study was conducted in Feb.-Mar. 2003 and Sept.-Oct. 2003 in a forest corridor (Figure 3-1) linking the Himalayan middle mountains to the Royal Chitwan national park in Chitwan district, Nepal (27° 31´ to 27° 44´ N and 84° 24´ to 84° 31´ E). The forests consists of almost pure stands of *Shorea robusta* Gaertn. interspersed by riverine *Acacia catechu* (L.f.) Willd., *Dahlbergia sissoo* Roxb. and *Trewia nudiflora* Linn. mixed forests and grasslands in shallow depressions and along riverbanks. The area was chosen for this particular study because the high population density surrounding the forest resulted in strong gradients of degradation and forest canopy density from the forest edge to the interior.

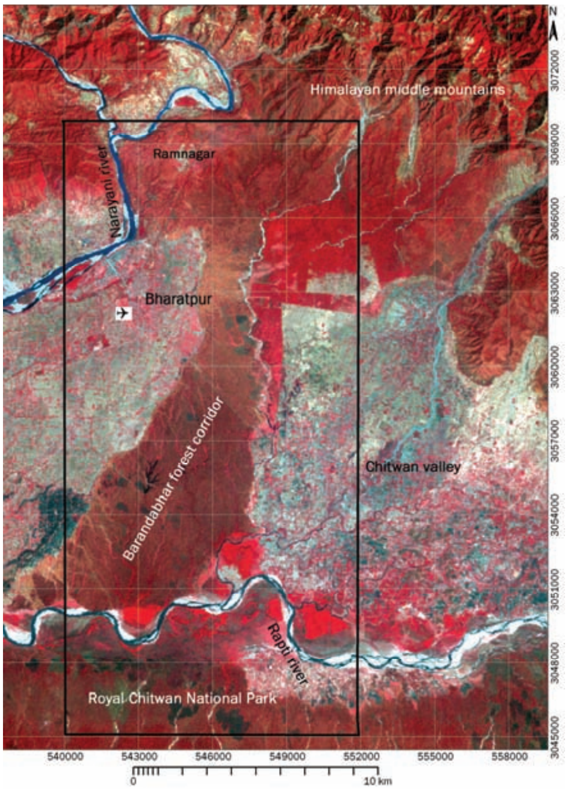

Figure 3-1. False colour composite (band 4, 3, 2) of ETM+ image of October 24, 2001 used in the present study. The area within black straight lines is the subset shown in figure 3-2.

3.2.2 Field and image data

A Landsat ETM+ image of October 24, 2001 was geo-referenced with subpixel accuracy of 17m. The thermal band was resampled to 30 m pixel size to normalise the spatial scale differences between the optical and the thermal band. The spectral radiance of each seven bands (band 1 to 7 of the ETM+ imagery) was calculated and than converted to reflectance which was finally used as an input for further analysis (NASA 2005).

Field data were collected from 372 sites selected by uniform random sampling using a land cover map of the area. Estimation of canopy density was based on two methods: 1) Densitometer estimation and 2) hemispherical photographs. At every site, densiometer estimation of canopy was recorded and hemispherical photographs were taken at the centre of a 30 x 30m plot.

Figure 3-2. Hemispherical photograph of forest canopy taken with fisheye lens attached to a Nikon digital camera.

Hemispherical canopy photography (Figure 3-2) is one indirect optical technique that has been widely used in studies of canopy structure and forest light transmission. Photographs taken skyward from the forest floor at the height of 1.2

m with a 180° hemispherical (fisheye) lens produce circular images that record the size, shape, and location of gaps in the forest canopy. These data are subsequently combined to produce estimates of growing-season light transmission, as well as other measures more directly related to canopy structure, such as openness, leaf area, and sunfleck frequency (Chazdon and Field 1987, Rich 1990). An index of sky openness corresponding to sky directly visible from the leaf (i.e. from the camera position) was computed from each photograph using Gap Light Analyzer (GLA) software version 2 (Frazer *et al*. 1999) and related to the canopy openness.

A view angle of 114° for canopy density calculation was considered. This corresponded to a 900m^2 canopy area for an average tree height of 22.5m. From this larger sample a sample of 50% of the sites (n=186) was randomly selected to develop method to predict canopy density (see 3.2.3). The remaining subset of n=186 sites was used to validate the predictions of these methods (see 3.2.4).

3.2.3 Forest canopy density prediction

3.2.3.1 Artificial neural networks (ANN)

Artificial neural networks (here after neural networks) are neurologically inspired statistical mechanisms also employed in classification of forest cover using various sensors (Barrow *et al*. 1993, Boyd *et al*. 2002). Here, a three-layer feedforward error-backpropagation artificial neural network implemented in Interactive Data Language (IDL) was used (Skidmore *et al*. 1997) in order to predict forest canopy density as a continuous variable. The algorithm minimises the root mean square error between the observed and the predicted values We followed (Atkinson and Tatnall 1997) to search for system parameters to increase the accuracy of the method and avoid overtraining of the neural network.

The neural networks with the sub sample of 186 sites with canopy density and the seven ETM+ bands was trained and the best combination of optimum learning rate and momentum to minimize the root mean square error (RSME) was empirically established. The best results were obtained with a learning rate of 0.7, a momentum of 0.7 and four hidden nodes. The RSME stabilized after approximately 7000 epochs. Finally, 20 iterations of 7000 epochs were performed and the best one was selected based on root mean square error.

3.2.3.2 Multiple linear regression techniques (MLR)

Multiple linear regression techniques (here after multiple regression) have been used to model the relation between spectral response and closed canopy conifer forest cover (Ripple 1994). In this study, we developed a multiple linear regression model, which best described the relation between canopy density and the seven ETM+ spectral bands. The regression eq. using n = 186 observations is:

$$\mathrm{Y} = 3.32 + 0.021 * \mathrm{B}_1 - 0.002 * \mathrm{B}_2 + 0.003 * \mathrm{B}_3$$

$$+ 0.024 * \mathrm{B}_4 \text{ - } 0.023 * \mathrm{B}_5 + 0.021 * \mathrm{B}_6 - 0.029 * \mathrm{B}_7$$

Equation 3-1

$R^2 = 0.62$, $F_{7,178} = 41.44$, $p < = 0.01$, where Y= Predicted forest canopy density and B_1 to B_7 is the reflectance value of band 1 to band 7 of Landsat ETM+ image.

3.2.3.3 Forest canopy density mapper (FCD)

Rikimaru (1996) introduced an alternative deductive approach i.e. forest canopy density mapper (here after density mapper) to map forest canopy density using four indices (vegetation, bare soil, shadow and surface temperature) derived from Landsat TM imagery. Based on these four variables, nine canopy density classes namely 0, 1-10, 11-20,....., 71-80+ were obtained.

This model involves bio-spectral phenomenon modeling and analysis utilizing data derived from four indices namely: Advance Vegetation Index (AVI), Bare Soil Index (BI), Shadow Index (SI), and Thermal Index (TI). Using these four indices, the canopy density for each pixel was calculated in percentage. Advanced vegetation index AVI reacts sensitively for the vegetation quantity compared with NDVI. Shadow index increases as the forest density increases. Thermal index increase as the vegetation quantity decreases and bare soil index increases as the bare soil exposure degrees of ground increase. After normalization of the Landsat ETM+ bands (except band 6) these indices has been calculated using equation:

$$if B_4 - B_3 < 0, AVI = 0 .... (Case \quad a)$$

$$if B_4 - B_3 > 0, AVI = ((B_4 + 1) * (256 - B_3) * (B_4 - B_{3)})^{1/3} .... (Case \quad b)$$

Equation 3-2

$$BI = \frac{(B_5 + B_3) - (B_4 - B_1)}{(B_5 + B_3) + (B_4 + B_1)} * 100 + 100$$

Equation 3-3

$$SI = (256 - B_1) * (256 - B_2) * (256 - B_3)$$

Equation 3-4

where: B is the spectral band of ETM+ image.

The DN value of thermal band (band 6) of ETM+ image was converted to units of absolute radiance using 32 bit floating point calculations (NASA 2005). The temperature calibration of the thermal infrared band (band 6) into the value of ground temperature has been done using equation:

$$T = \frac{K_2}{\ln\left(\frac{K_1}{L\lambda} + 1\right)}$$

Equation 3-5

where: T = Effective at-satellite temperature in Kelvin; K1 = 666.09 watts/(meter squared * ster * µm) K2 = 1282.71 Kelvin; L = Spectral radiance in watts/(meter squared * ster * µm).

As an intermediate step, VI and BI were synthesized in to Vegetation Density (VD) value using principal component analysis and VI and SI into Scaled Shadow Index (SSI) by linear transformation of SI. In areas where the SSI value is zero, this corresponds with forests that have the lowest shadow value (i.e. 0%). In areas were the SSI value is 100, this corresponds with forests that have the highest possible shadow value (i.e. 100%). Details in Rikimaru, 1996). Finally, VD and SSI were synthesized into a forest canopy density value as:

$$FCD = (VD * SSIO) + 1)^{1/2} - 1$$

Equation 3-6

The method requires intervention by an operator to establish threshold values. The accuracy obtained in three SE Asian countries averaged 92% (Rikimaru and Miyatake 1997). For this study, FCD- mapper software ver. 2 (Rikimaru and Miyatake 1997) and ILWIS 3.2 (ILWIS 2005) were used.

3.2.3.4 Maximum likelihood classification (MLC)

As a parametric classifier, maximum likelihood classification method (here after maximum likelihood) calculates the probability that a given pixel belongs to a specific class and assigns the pixel to the class having the highest probability (Richards 1999). The training set of 186 pixels into ten canopy classes namely 0, 1-10, 11-20,....., 71-80+ was classified. The Interactive Data Language (IDL 6.0) and ENVI 4.0 (ENVI 2003) was used for image classification.

3.2.4 Comparison of classifier performance

ANN and MLR predicted forest canopy density as a continuous variable, where as FCD and MLC methods predicted canopy density as a categorical variable. Therefore, to compare the performance of all four classifiers using Kappa statistics, we grouped ANN and MLR classification results into ten canopy classes same as FCD and MLC. Normally in remote sensing the performance of image classification methods is analysed using error matrices and Kappa statistics (Congalton and Green 1999). In this case, with ranked rather than nominal categories in the table, the weighted Kappa statistic (Naesset 1996) was used. Table 3-1 gives the agreement weights used for the weighted Kappa. A z-test was performed for a pairwise comparison of the methods, testing the null hypothesis H0: $\kappa_1 = \kappa_2$, where κ is the weighted Kappa value for the map produced by the method.

Table 3-1. Matrix with agreement weights used for calculation of weighted Kappa

Observed category	Predicted category								
	0	10	20	30	40	50	60	70	80+
0	1.00	0.89	0.78	0.67	0.56	0.44	0.33	0.22	0.11
10	0.89	1.00	0.89	0.78	0.67	0.56	0.44	0.33	0.22
20	0.78	0.89	1.00	0.89	0.78	0.67	0.56	0.44	0.33
30	0.67	0.78	0.89	1.00	0.89	0.78	0.67	0.56	0.44
40	0.56	0.67	0.78	0.89	1.00	0.89	0.78	0.67	0.56
50	0.44	0.56	0.67	0.78	0.89	1.00	0.89	0.78	0.67
60	0.33	0.44	0.56	0.67	0.78	0.89	1.00	0.89	0.78
70	0.22	0.33	0.44	0.56	0.67	0.78	0.89	1.00	0.89
80+	0.11	0.22	0.33	0.44	0.56	0.67	0.78	0.89	1.00

Furthermore, the predicted against observed canopy density values were plotted for the four methods and their agreement tested by a simple linear regression for each. The coefficient of determination and slopes and intercepts were used to assess the performance of predictors. An unbiased predictor would be expected to have a slope (β) of 1 and intercept (α) of zero. Hence the hypothesis: H0: $\alpha = 0$ and H0: $\beta = 1$ was tested. Besides, the variance of the observed canopy density explained by the predicted density was used to assess the random errors in the model.

3.3 Results

Table 3-2 presents the error matrix for the artificial neural network classifier. The matrix shows that 110 out of 186 observations had been correctly classified. There were few severe outliers, only 6 observations were more than two classes and 22 observations were more than one class from the diagonal. The linear regression method correctly classified 64 out of 186 observations. There were many more severe outliers, 29 observations more than two classes and 57 observations were more than one class off diagonal (Table 3-3). The forest canopy density mapper with 92 out of 186 observations correctly predicted, 13 observations more than two classes and 42 observations were more than one class off diagonal (Table 3-4). The maximum likelihood classification predicted 82 out of 186 observations correctly, 21 observations more than two classes and 51 observations were more than one class off diagonal (Table 3-5).

Table 3-2.Error matrix for observed versus predicted canopy density class by an artificial neural network

Predicted class	Observed class									
	0	10	20	30	40	50	60	70	80+	Total
0	50	5	3							58
10	4	1	3							8
20		2	3	2	3					10
30	1		1	5	3					10
40		1	2	4	5	3				15
50	1		3	3	4	15	4	1		31
60					4	7	12	3		26
70							2	10	5	17
80+								2	9	11
Total	56	9	15	14	19	25	18	16	14	186

Table 3-3. Error matrix for observed versus predicted canopy density class by a multiple regression model

Predicted class	Observed class									
	0	10	20	30	40	50	60	70	80+	Total
0	31	2								33
10	1									1
20	8		2							10
30	7	4	4	3	1	1				20
40	6	2	4	6	5	6	1			30
50	2	1	4	4	11	10	11	3	3	49
60	1		1	1	1	7	6	10	5	32
70					1			2	1	4
80+						1		1	5	7
Total	56	9	15	14	19	25	18	16	14	186

Table 3-4. Error matrix for observed versus predicted canopy density class by the forest canopy density mapper

Predicted class	Observed class									
	0	10	20	30	40	50	60	70	80+	Total
0	50	6	4	2	1	2				65
10	1	2								3
20			2	1	1					4
30	4	1	2	4	6	5				22
40	1		5	4	5	9	3			27
50			2	2	3	7	7	2		23
60				1	3	1	6	1	2	14
70						1	2	9	5	17
80+								4	7	11
Total	56	9	15	14	19	25	18	16	14	186

Table 3-5. Error matrix for observed versus predicted canopy density class by a maximum likelihood classification

Predicted class	Observed class									
	0	10	20	30	40	50	60	70	80+	Total
0	45	3	1	1						50
10			1	1						2
20	6	2	4	2	3	2				19
30	2	2	3	4	2	3	1			17
40	1	2	3	1	4	2		1		14
50	1			2	3	5	2	1		14
60	1		3	2	4	11	6	1	1	29
70				1	3	2	8	2	1	17
80+							1	11	12	24
Total	56	9	15	14	19	25	18	16	14	186

The Kappa statistics (Table 3-6) showed that neural network resulted in an overall weighted Kappa accuracy of 63.1%, which was 6.7% larger than the nominal Kappa. The weighted Kappa accuracy was increased by 5.6% for multiple regression, 10.1% for density mapper and 11.2% for maximum likelihood.

Table 3-6. Values of the overall accuracy, weighted Kappa accuracy, weighted Kappa statistic and their variances for the four methods

Method	Overall accuracy	Weighted Kappa accuracy	Kappa	Weighted Kappa	Weighted sigma^2
Artificial neural networks	59.1	63.1	0.51	0.55	0.00159
Multiple linear regression	34.4	39.8	0.23	0.29	0.00178
Forest canopy density mapper	49.5	54.5	0.39	0.44	0.00158
Maximum likelihood classification	44.3	49.8	0.34	0.40	0.00159

Table 3-6 further showed that neural network resulted in a nominal Kappa value of 0.55 and increased by 7.8% with the weighted Kappa. The weighted Kappa was 0.29 and increased by 26% for multiple regression, 0.44 and increased by 12.8% for density mapper and 0.40 and increased by 17.6% for maximum likelihood. The variance of weighted Kappa value for multiple regression was 0.00178 which was significantly higher than the values of other three methods. The variance of weighted Kappa value for other three methods was almost similar.

The z tests comparing the performance of the four methods based on weighted Kappa and their variances (Table 3-7) showed that the calculated z value for neural network and multiple regression was 4.52 which was higher than 1.96 at $\alpha = 0.05$ conclude that maps produced by the two methods are significantly different from each other. Similarly, the map produced by the neural networks was significantly different from the map produced by maximum likelihood classification. The calculated z value for density mapper and multiple regression was 2.70 which showed a significant difference between them. We failed to detect significant differences between the maps classified by neural networks and the forest canopy density mapper. Figure 3-3 shows the forest canopy density class map produced by the four methods.

Table 3-7. Values for z tests comparing the performance of the four methods based on weighted Kappa and their variances (significant at higher than 1.96 at $\alpha = 0.05$)

Method	ANN	MLR	FCD	MLC
ANN	-	4.52	1.93	2.70
MLR		-	2.64	1.89
FCD			-	0.77
MLC				-

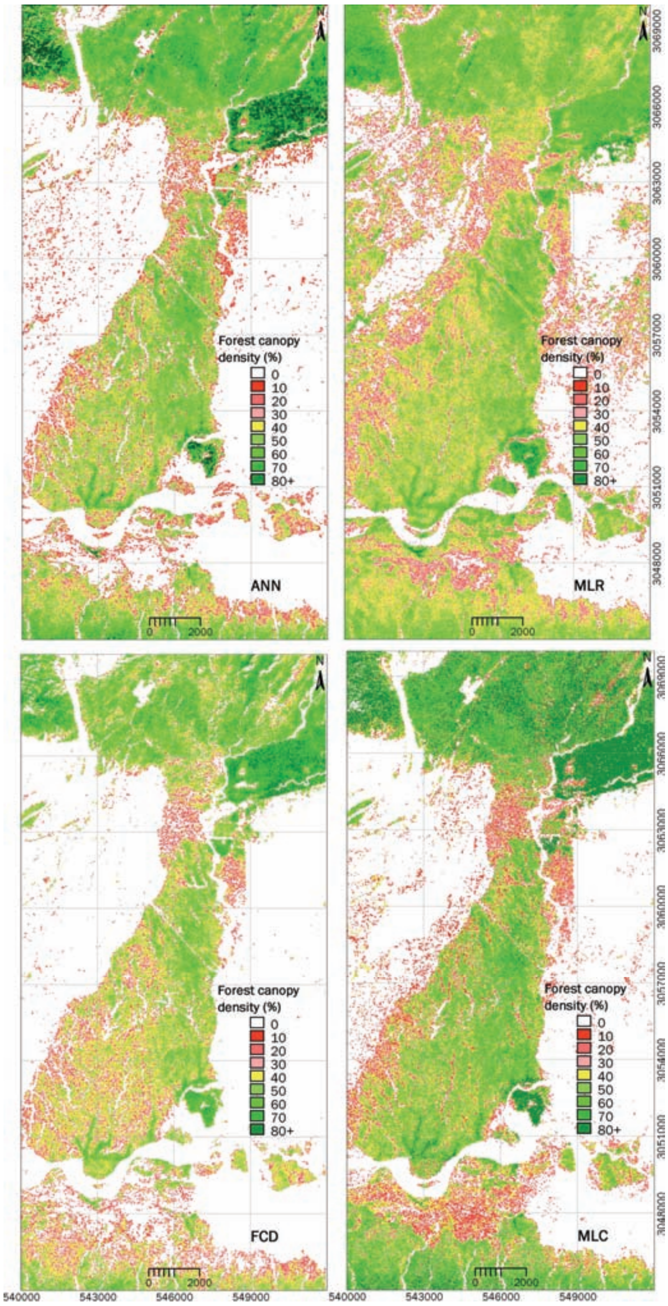

Figure 3-3. Forest canopy density maps produced by the artificial neural network (ANN), multiple linear regression (MLR), the forest canopy density mapper (FCD) and maximum likelihood classification (MLC).

The ANN produces the highest overall accuracy of 63 percent when compared to other three methods. The MLR incorrectly mapped the “lower” as well as “higher” canopy classes in the middle canopy classes where as MLC incorrectly mapped most of the “lower” canopy classes (Figure 3-3). The ANN classification appears similar to the output of the FCD. In summary, both ANN and FCD were significantly different from both the MLR and MLC, whilst the ANN produces the highest mapping accuracy.

Figure 3-4 presents the performance of the four methods in scatterplots of observed versus predicted canopy density. The figure shows the fitted regression line against the expected line for an unbiased predictor with intercept (α) = 0 and slope (β) = 1. Visual comparisons of these pairs of regression lines suggest bias, which was further tested using the results presented in Table 3-8. The observed slope and intercept significantly differed from one and zero for the forest canopy density mapper as well as the maximum likelihood classification. Hence, we considered these two methods as biased predictors.

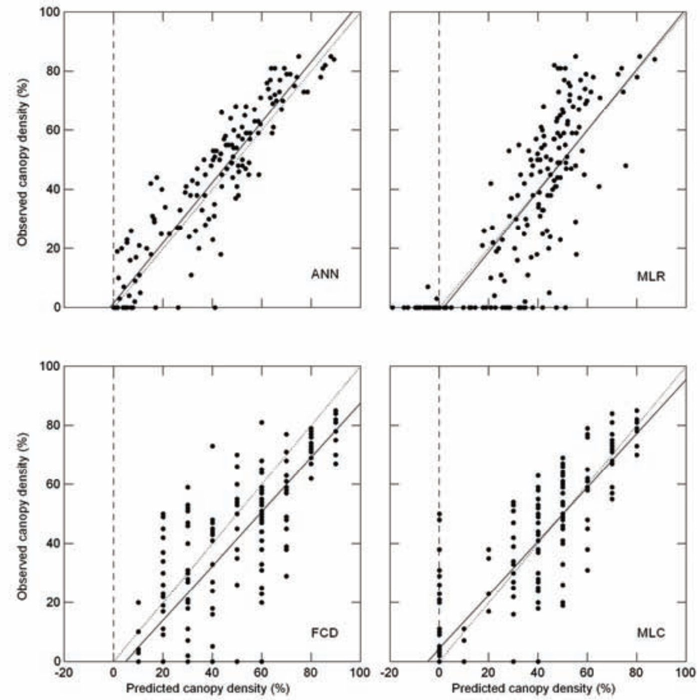

Figure 3-4. Scatterplot of observed versus predicted forest canopy density for the artificial neural network (ANN), multiple linear regression (MLR), the forest canopy density (FCD) mapper and maximum likelihood classification (MLR). The dotted line corresponds to an unbiased predictor, while the solid line represents the observed linear regression relationship.

We failed to reject both null hypotheses for the artificial neural network and linear regression models. The fit of the neural network predictions against the observations was quite good, with observations equally distributed across the range of observations (Figure 3-4). This was not the case for the multiple linear regression model. Here the model overpredicted at low canopy density while it underpredicted at high canopy density. Hence, we considered this also a biased predictor. We therefore conclude that the artificial neural network method did not produce biased predictions.

Table 3-8 reveals that 89% of variance of observed canopy density was explained by the predictions in case of the neural network, 64% by the multiple regression, 79% by the density mapper and 74% by the maximum likelihood. The standard error of the estimate of 9% for the neural network was considerably lower than the other three methods.

Table 3-8. Regression statistics describing the relationship between predicted and observed canopy density for the four methods

Method	R^2	Intercept (α)	standard errorα	tα	Slope (β)	Standard errorβ	tβ	Standard error of estimation
Artificial neural networks	0.89	0.017	0.011	1.545	1.019	0.026	0.731	0.092
Multiple linear regression	0.64	-0.001	0.022	-0.045	0.992	0.054	-0.148	0.168
Forest canopy density mapper	0.79	0.044	0.015	2.933	0.912	0.035	-2.514	0.131
Maximum likelihood classification	0.74	0.044	0.020	2.246	0.920	0.040	-2.015	0.144

The methods tested in this study perform differently and produce different classification results. However, in this case the artificial neural network method is unbiased than the other three methods, achieved the highest classification accuracy as well as it could map forest canopy density as a continuous variable. The predicted canopy density map obtained from the artificial neural network classification method is shown in Figure 3-5.

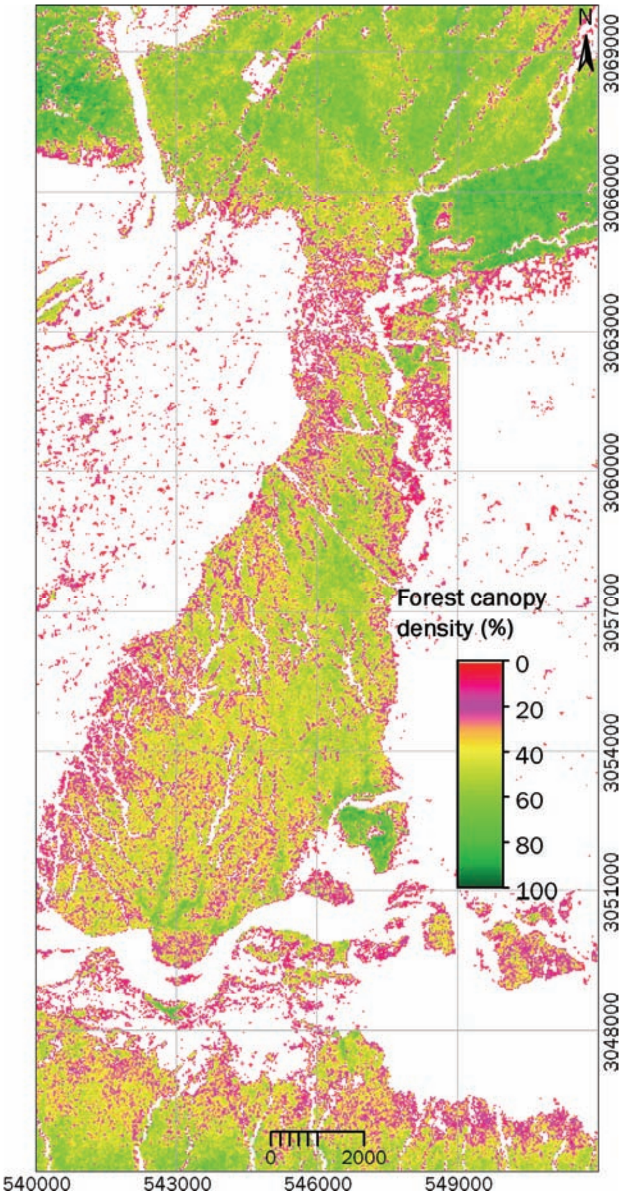

Figure 3-5. Forest canopy density map produced by the artificial neural network classification method in Chitwan region, Nepal.

3.4 Discussion

Our results indicated differences in performance between the four methods. The artificial neural network performed better than multiple linear regressions, forest canopy density mapper or the maximum likelihood classification since it explained a higher proportion of the variance and its predictions were unbiased. This confirms the experience that neural networks approaches in image classification tend to be more accurate than other statistical techniques (McClelland *et al*. 1989, Hepner *et al*. 1990a, Downey *et al*. 1992, Kanellopoulos *et al*. 1992, Atkinson and Tatnall 1997, Boyd *et al*. 2002, Liu *et al*. 2003).

Hence, the neural networks classification method was superior in performance and would appear to be a first choice for mapping forest canopy density. However, the neural networks approach has some drawbacks. Firstly, there are no hard rules about neural topology and training algorithm and it is not always possible to know how a neural network obtained a given result. Secondly, like the other empirical methods it requires relatively large dataset which increases the cost of the survey.

Although a classical multiple linear regression technique is unbiased in explaining of variance of the predicted canopy density explained by the observations, poor performance of the technique showed that canopy density may not be well explained by the multiple regression analysis using TM data, a conclusion reinforced by the findings of (Jakubauskas and Price 1997).

The maximum likelihood classification method is a well-known parametric method, which looks at the shape, size and orientation of the training samples (Tou and Gonzalez 1974). If normality assumption in feature space for each training class area is valid, then the classification has a minimum overall error probability and the method is the optimal choice (Swain 1978). However, the normality assumption of the training set is sometimes violated. Here, the method has a lower overall accuracy, significantly different from neural networks and biased prediction because forest canopy density was assumed as a continuous variable and the decision boundaries do overlap (Skidmore *et al*. 1988). Hence, maximum likelihood classification method could not well classify forest canopy density as a continuous variable.

Some of these problems do not hold for the other methods presented. The forest canopy density mapper for instance can be used with existing expert knowledge to select the best combination of the parameter thresholds or the combination of indices to produce an optimal classification. It was designed to increase the information collecting capability of foresters reducing cost and saving time (Rikimaru *et al*. 2002). Drawbacks of the forest canopy density mapper are that the performance of the method depends on the user's personal performance, skill and use of expert knowledge. It requires intervention by an operator to establish threshold values and make specific adjustments according to vegetation type and terrain condition (Rikimaru and Miyatake 1997). The choices made influence the class boundaries in the resulting map. This can lead to subjectivity and impact on the predicted canopy density. This method is a more subjective than the three other methods. The forest canopy density mapper is thus a somewhat less accurate method, which could be used, if the analyst is aware of risk of bias and subjectivity. We propose to use this method as a quick and less accurate method for forest canopy density estimation in discrete class in those cases where neural networks methods are not an alternative. The coefficient of determination explained in this study was lower than the 92% as reported by Rikimaru (1996). This may possibly be the extreme local variation of canopy cover in our study. The better result of his study probably reflects the regional approach, where such errors played a smaller role.

Previous approaches to map forest canopy density estimated in two (Boyd *et al*. 2002) or a few classes (Rikimaru and Miyatake 1997, Rikimaru *et al*. 2002) rather than a continuous variable. The latter approach is preferred to derive information on ecological forest conditions. In this study, we demonstrated that it is possible to predict canopy density as a continuous variable with a standard error of 9%. This error is the result of an empirical neural network method development. Part of the error in the model may have been introduced by the geometric accuracy of matching the imagery to the field observations used for training and validation. The root mean square error of image registration equalled 0.56 pixels, corresponding to ± 17m in the field. Additionally seasonality and the time difference between the date of image acquisition and field data collection may have introduced some error. It is very apparent from the results matrices (Figure 3-4) that the most of the tested methods were less predictive at intermediate canopy densities (i.e. for observed classes 20 to 40%), it could be the effect of the understorey vegetation cover, which sometimes changed considerably over such short distances due to management practices. We assume that canopy estimation by neural networks might well be more precise than the 9% error reported in this study, when the geometric accuracy could be improved, a possibility pointed out by (Rennolls 2002, Waltham *et al*. 2002).

3.5 Conclusions

The methods tested in this study perform differently, and produce different classifications. However, a three-layer feedforward error-backpropagation artificial neural network is an unbiased method and performed better than the maximum likelihood classification, forest canopy density mapper or traditional multiple linear regression methods in forest canopy density prediction.

The performance of a method, as expressed in the traditional confusion matrix, is often not a superior indicator of thematic map accuracy, especially when the map is based on continuous variables (e.g. forest canopy density) as a confusion matrix may not express the biased nature of a method. Presentation of data in a scatterplot rather than a traditional error matrix could be an alternative approach to evaluation of bias and unbiased methods.

The weighted Kappa could be an appropriate statistic for analysis of ranked categories. Forest canopy density is quantified as a continuous or ranked variable and an alternative approach could be using regression diagnostics. Confusion matrices and associated statistics, which have been used in previous forest canopy density studies, revealed little difference between methods in this study. The regression techniques provided better insight into the performance of the respective methods and allowed to distinguish between systematic and random errors.

The findings of our study illustrate different classification techniques can provide ecologists and land managers with reasonable accuracy in mapping forest canopy density at local landscape scales. However, canopy density differences that have been obtained for a forest canopy with the different classification methods clearly point out that the choice of biased method could have a high impact on canopy density inventories.

CHAPTER FOUR

Mapping the cover and seed production of the cryptic forest understorey invasive shrub *Chromolaena odorata*

This Chapter is based on:

C. Joshi, J. De Leeuw, J. van Andel , A. K. Skidmore, H. D. Lekhak, I. C. van Duren and N. Norbu 2006. Indirect remote sensing of a cryptic forest understorey invasive species. Journal of Forest Ecology and Management (225) 1-3: 245-256.

and

C. Joshi, J. De Leeuw, J. van Andel , A. K. Skidmore, H. D. Lekhak, I. C. Van Duren 2005. Remote sensing and GIS for mapping and management of invasive shrub *C. odorata* in Nepal. AGILE 2005, 8th conference on Geographic Information Science, 26-28, May, 2005, Portugal pp. 71 – 80.

4 Mapping the cover and seed production of the cryptic forest understorey invasive shrub *Chromolaena odorata*[*]

Abstract

Remote sensing has successfully been applied to map the distribution of canopy dominating invasive species. Many invaders however, do not dominate the canopy, and remote sensing has so far not been applied to map such species. In this study, an indirect method was used to map the seed production of *Chromolaena odorata*, one of the world's one hundred worst invasive species. The study was executed in lowland *Shorea robusta* forest in Nepal, where *C. odorata* invaded the understorey of degraded forest. A Landsat ETM+ image processed through a neural network predicted 89% and 81% of forest canopy density and light intensity reaching the understorey respectively. We inverted these models to predict *C. odorata* seed productivity. Light intensity determined 93% of the variation in $\log_{10}$ seed production per plant. *C. odorata* failed to produce seed below a light intensity of 6.5 mJ m^{-2} day^{-1}. Further analysis revealed that *C. odorata* was absent above this light intensity in case of a high biomass of other shrub and herb species, a situation occurring in the absence of grazing. We therefore suggest that other species control *C. odorata* through competitive exclusion in the absence of grazing, whereas grazing breaks the dominance of these other species thus creating the conditions for *C. odorata* attain canopy dominance. The presence of grazing was related to distance from the forest edge, a variable that together with light intensity allowed us to map 64% of variation in *C. odorata* cover. Predicted *C. odorata* cover and seed production per plant were combined into a map displaying the total seed production per unit area. Such map displaying seed producing sites could be used to significantly reduce the costs of controlling *C. odorata* infestation by providing information on the spatial segregation of source and sink populations, which will support efficient habitat ranking to restore invaded areas and protect non-invaded ecosystems. This may prove particularly valuable when implementing control measures under circumstances of limited capital and manpower.

Key words: Remote sensing, GIS, *Chromolaena odorata*, seed production, light intensity, invasive species, mapping

---

[*] Winner: Two best paper awards: best remote sensing paper and best conference paper, 8th AGILE conference, Portugal 2005.

4.1 Mapping cryptic invaders

The spread of invasive species has generated interest in mapping their present distribution. Predicting the current or future distributions of species has principally been conducted using a 'bioclimatic envelope' that assume that climate ultimately restricts species distributions (Ungerer *et al*. 1999, Kriticos and Randall 2001). In addition to actual distribution, the potential distribution is increasingly mapped, in an attempt to localize areas at risk of invasion. Rouget *et al.* (2004) for instance predicted the potential distribution of invasive species in southern Africa from climatic variables.

Potential distribution maps may be useful for invasion risk assessment as well as for country level baseline studies. They do not however, suffice for local management of invasive species aimed at eradication or control. For such purpose, one may wish to identify, at a larger geographical scale, areas where the invader is actually present. Presence data alone may, however not suffice. With limited resources available for invasive species management one may wish to localize areas that deserve priority. Prioritization could be based on various criteria (Moody and Mack 1988, Mack *et al*. 2000, Pressey and Taffs 2001, Cowling *et al*. 2003, Reyers 2004), and for example focus on areas where the invader dominates and outcompetes native species. Alternatively, large scale maps could be useful to prioritize areas deserving control of invaders producing seeds, which potentially could infest areas not yet invaded.

Since the first satellite images reached the scientific community, remote sensing has been recognized as a possibility to map the distribution of invasive species at local scale (Tucker 1979, Anderson *et al*. 1993, McCormick 1999, Ramsey *et al*. 2002, Zhang *et al*. 2002a, Underwood *et al*. 2003). Applications of remote sensing, however have been restricted to species dominating the canopy of ecosystems (Chapter 2). Dominant species determine the spectral signature and thus species detection using their spectral signature is straightforward. Many invasive species however, do not dominate the canopy. We observed (Chapter 2) that 67% of the one hundred world's worst invasive species (ISSG 2004) were rare or hidden below the ecosystem canopy. Such species would be difficult to localize using their spectral signature, since they contribute partly or not at all to the radiation received by a remote sensing device. Not surprisingly, none of these 67 species has successfully been mapped.

We proposed indirect mapping approaches to map this distribution of invasive species (Chapter 2). Indirect mapping approaches predict the distribution of species using knowledge about the ecological relation between a species and it's environment. At regional scales, this approach is commonly used for predicting potential distribution of a species under current and future climates (Messenger 1959, Baker *et al*. 2000, Warren *et al*. 2001, Kriticos *et al*. 2003, Sutherst 2003, Yonow *et al*. 2004). At local scales, it has been recognized that the distribution of invaders is related to variability of the environment. Land-use changes such as the replacement of natural ecosystems by agricultural systems clearly alter many ecosystems and may promote biological invasions (Hobbs 2000a). Frenot *et al.* (2001) and Hong *et al.* (2003) for instance related the current distribution of invasive species to land cover types and suggested that land use change might have contributed to their invasive success.

The above paragraph suggest consensus that land use change stimulates invasions. There is however, an ongoing debate to what extent the invasive success of exotic species depends on traits of the species or the condition of the environment (Harper 1977, Johnstone 1986). Some researchers attributed invasions to inherited traits (Elton 1958, Lodge 1993, Williamson 1996). No doubt, highly invasive species combine successful reproductive, competitive and dispersal traits. The expression of these traits however, is the result of the interaction with its environment (Pysek *et al*. 1998). Species highly invasive in one environment may well be incapable of maintaining viable populations under different conditions. Hence, spatial information on environmental conditions affecting an invaders reproductive and competitive trait could be used to predict its distribution.

The shrub species *Chromolaena odorata* (L.) R. M. King & Robinson, originating from Central America, has invaded tropical forests around the world (Goodall and Erasmus 1996, McWilliam 2000). It typically occurs in the forest understorey in the Southern Himalayas (pers. obs.). The invasiveness of *C. odorata* is thought to depend on the combination of production of large quantities of easily dispersed achenes (Cypsela) and its capability to suppress the native vegetation through light competition (Erasmus 1985, Honu and Dang 2000). These traits however, appear to be phenotypical, since the species shows a remarkable response to light (Witkowski and Wilson 2001). It remains virtually absent from dense forest, but dominates and vigorously produces seed in forest clearings and ruderal environments with more open canopies. This information suggests that light intensity determines the competitive and reproductive traits of *C. odorata*. Hence, light intensity in the forest could possibly be used to map the distribution and reproduction of *C. odorata*.

The aim of this study was to investigate the possibility of indirect mapping techniques to localize areas, where *C. odorata* was capable of expressing its invasive traits. For this purpose, we analyzed the ecological relationships between *C. odorata* and its environment and used the acquired understanding to map its distribution.

4.2 Methods and Materials

4.2.1 Study Area

The study area in south central Nepal is a forest corridor linking the Himalayan middle mountains to Royal Chitwan National Park (27°31´ to 27°44´N and 84°24´ to 84°31´E). The forest is dominated by *Shorea robusta* Gaertn., a drought deciduous tree species. Grasslands occur in depressions within the forest. Densely populated irrigated croplands surround the study area.

Forests in the National Park were generally well conserved. Other forests, although protected by legislation, have been severely degraded through logging and lopping of the trees, grazing of the understorey vegetation, frequent fires and collection of non-timber forest products. The resulting local variation in canopy degradation status made the study area particularly suitable for the current research.

4.2.2 Data collection

The study area was visited twice. In September and October 2003, 302 plots were visited to measure forest canopy density percentage and light intensity at 1.2 m above the ground using hemispherical photography taken with fisheye lens attached to a Nikon digital camera. Total light intensity was calculated from the digital hemispherical photographs using Gap Light Analyzer software version 2 (Frazer *et al*. 1999).

The study area was revisited in February - April 2004, when *C. odorata* was bearing seeds. Field data were collected from 153 randomly located plots. A plot size of 30 by 30 m was used to visually estimate *C. odorata* cover percentage (Figure 4-1A). For each plot, five reproductive plants were randomly selected, and for each plant: a) number of flowering stems per plant (st), b) number of flowering branches per stem (br), c) number of inflorescences per flowering branch (in), d) number of capitulum per inflorescence (ca) and e) number of cypselum per capitulum (sy) were measured (Figure 4-1B).

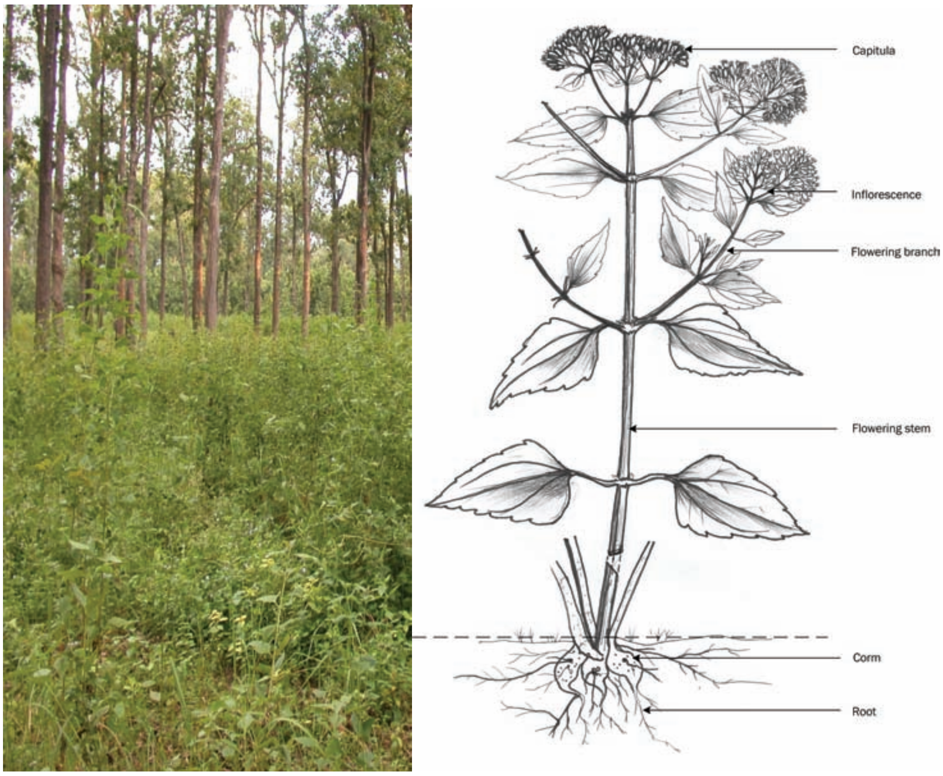

Figure 4-1. *Chromolaena odorata* growing under *Shorea* forest in the Chitwan region of Nepal (Left) and morphology of *C. odorata* (Right).

Next, the response of the seed production of *C. odorata* to the same environmental variables was analyzed. For this, only those sites where *C. odorata* was present were selected. The seed production per plant was reflected by the size and architecture of plants. From the five above described properties, we calculated the average seed production per plant (SPP):

$$SPP = st * br * in * ca * cy \qquad \text{Equation 4-1}$$

Other species in the forest understorey may have competed with *C. odorata*. Therefore, the cover (c, %) and height (h, m) of the herb (H) and shrub (S) layer in the understorey were visually estimated and from these we calculated a biomass index (BI) of these other understorey species:

$$BI = \oint (Hc * Hh + Sc * Sh) \qquad \text{Equation 4-2}$$

Furthermore, grazing intensity was assessed based on height (m) and cover (%) of palatable plant species and the density of dung.

Topographic maps of 1996 of the study area were digitized to obtain maps of settlements, roads, forest edge, land cover and land use.

4.2.3 Data analysis

Exploratory data analysis was used to investigate the relation between the biometry of *C. odorata* and a number of environmental factors. Curve Expert 1.3 (Hyams 2005) was used to explore sigmoid functions. A path analysis (Sokal and Rohlf 1995) was conducted to determine the degree to which single environmental factors made an independent contribution to the prediction of *C. odorata* cover. The model included the multiple-way interaction of forest canopy density, grazing intensity, light intensity, biomass of the forest understorey vegetation and distance from road/trail and forest edge. Systat 11 (Systat 2004) was used for statistical analysis.

4.2.4 Mapping

4.2.4.1 Grasslands

Waterlogged depressions with open grassland, which did not represent the true forest environment, were classified using maximum likelihood image classification method as to separate only grassland from the rest and masked from further analysis.

4.2.4.2 Light intensity and forest canopy density

An artificial neural network was found to be unbiased and more accurate classifier than the other traditional methods and could well classify physical parameters as a continuous variable (Chapter 3). Hence, a three-layer feedforward backpropagation artificial neural network (ANN) consisting of an input, a hidden and an output layer (Skidmore *et al*. 1997) was used to map light intensity using the first seven bands (band 1-7) of Landsat ETM+ image of October 24, 2001 (path 142, row 42). The image was registered with sub-pixel accuracy (±17m). Light intensity reaching the forest understorey derived from the hemispherical photographs was used to train the artificial neural network. All 302 observations were randomly divided into two groups. One subset of 151 observations was used for training and the other 151 for testing. The backpropagation algorithm used is designed to minimize the root mean square error (RMSE) between the observed and the predicted output. The combination of learning rate and momentum optimally minimizing the root mean square error was empirically established. The best results were obtained with a learning rate of 0.9, a momentum of 0.7 and two hidden nodes. Finally 20 iterations of 7000 epochs were performed (the RMSE stabilized after approximately 7000 epochs) and the best classification based on least root mean square error was selected. Forest canopy density in percentage was also calculated using the same data set and procedure. Image classification and GIS analyses were performed with ENVI 4.1 (2003), ILWIS 3.3 (2005) and ArcGIS 9.0 (ESRI 2004).

4.3 Results

4.3.1 Environment data layers

Figure 4-2 shows the environmental data layers used for analysis in section 4.3.2 and 4.3.3 and spatial predictions in section 4.3.4.

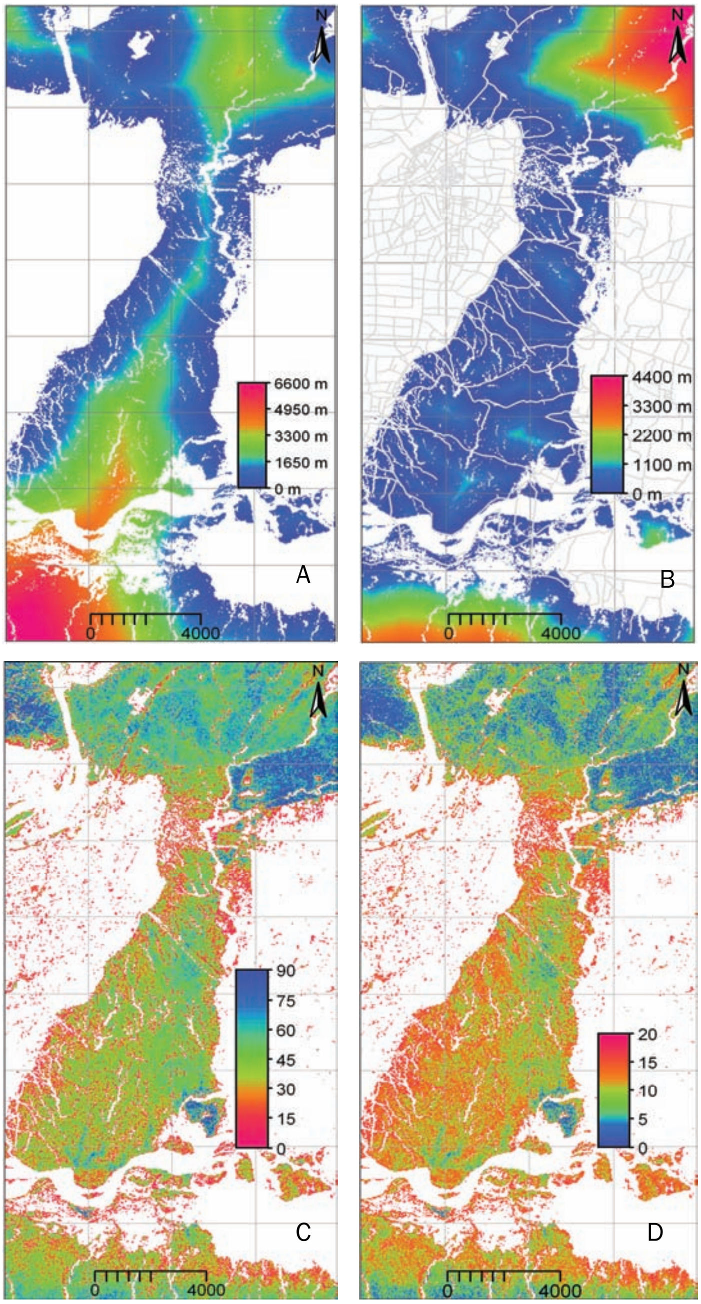

Figure 4-2. Maps of A) distance from the forest edge (m); B) distance from roads and trails (m); C) forest canopy density (%); D) average daily light intensity (mJ m-2day-1) reaching the forest floor. Forest canopy density and light intensity maps were generated using a Landsat ETM+ image of October 24, 2001.

Figure 4-3 reveals that the light intensity map from the neural network classifier (Figure 4-2D) explained about 81 % of the variance in light intensity of an independent validation sample. The prediction of light intensity was unbiased, since the slope (b) and intercept (a) did not differ significantly from 1 and 0 respectively (a = -0.002, SEa = 0.005, ta = -0.361, b = 1.003, SEb = 0.035, tb = 0.086). Similarly, in chapter 3 we reported a high reliability of the forest canopy density map.

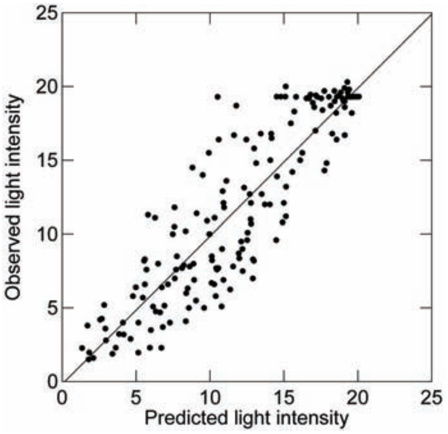

Figure 4-3. Scatterplot of light intensity reaching the forest floor observed in the field, against the light intensity predicted by a neural network using a Landsat ETM+ image.

4.3.2 Cover

The cover of *C. odorata* was positively correlated to light and grazing intensity and negatively related to all other variables (Table 4-1).

Table 4-1. Correlation (r) between six environmental variables and the cover of *C. odorata*. Significance levels: *p< 0.001

Environmental variable	r
Forest canopy density	-0.69*
Light intensity	0.71*
Grazing intensity	0.32*
Distance from road/trail	-0.34*
Distance from forest edge	-0.25*
Understorey biomass	-0.50*

Stepwise multiple regression model (Equation 4-3) included light intensity (X1) and biomass of the other forest under-storey species (X2) as significant variables explaining 64% of the variance in *C. odorata* cover (Y):

$$y = -0.28 + 2.88 * x_1 - 16.0 * x_2$$ Equation 4-3

Light intensity had a positive influence, while the biomass of the other under-storey species negatively influenced *C. odorata* cover. Hence, only two of the variables in Table 4-1 were included in this model. The six variables displayed in table 4-1 were, however, strongly interrelated. Some may have affected cover indirectly through other variables. Path analysis (Figure 4-4) revealed that distances from forest edges and from roads had a direct influence on *C. odorata* cover as well as an indirect influence through grazing that influenced the biomass of other forest under-storey species. Similarly, forest canopy density indirectly influenced *C. odorata* cover through light intensity.

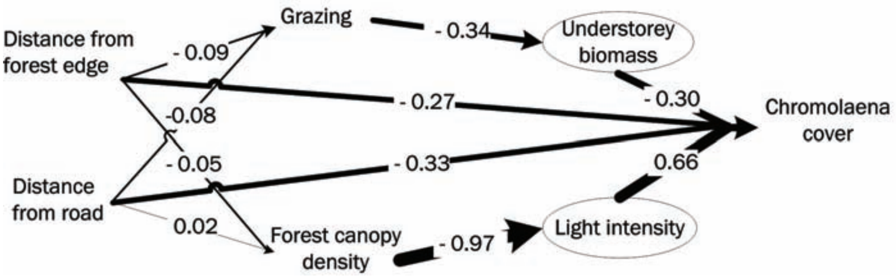

Figure 4-4. Path diagram showing the effect of six variables on the cover of *C. odorata*. Solid arrows indicate significant effects with path coefficients. Signs indicate negative and positive relationships accordingly, line width is proportional to the size of path coefficients.

Biomass of the other forest understorey species was a variable significantly contributing to the above regression model. It was not however, included in the spatial prediction of *C. odorata* cover because it was impossible to map the spatial distribution of this factor. Only light intensity, canopy density and distance from forest edges and roads were mappable. Hence, we used stepwise linear regression to select the best fitting model (Equation 4-4) using these variables, which explained 64% of the variation of *C. odorata* cover abundance:

$$y = -9.56 + 3.14 * x_1 - 0.005 * x_2$$ Equation 4-4

where X_1 = Light intensity and X_2 = distance from forest edge

4.3.3 Seed productivity of *C. odorata*

The seed production per plant differed between highly and poorly reproductive populations by more than five orders of magnitude. The reproductive biometry of *C.*

odorata was significantly related to a number of environmental variables (Table 4-2).

Table 4-2. Correlation between six environmental variables and the logarithm of five biometric properties and the seed production per plant (SPP) of *C. odorata*. Significance levels: *p< 0.05, **p< 0.01 and ***p<= 0.001

Environmental variable	*Chromolaena odorata*					
	Flowering stems	Flowering branches	N inflorescence	N Capitula	N Cypsela	SPP
Forest canopy density	-0.84***	-0.85***	-0.84***	-0.80***	-0.70***	-0.86***
Light intensity	0.85***	0.87***	0.85***	0.79***	0.69***	0.86***
Grazing intensity	0.17*	0.18*	0.15 ns	0.14 ns	0.12 ns	0.16*
Distance road/trail	-0.54***	-0.47***	-0.54***	-0.54***	-0.53***	-0.55***
Distance forest edge	-0.27***	-0.22**	-0.27**	-0.25**	-0.24**	-0.25**
Under-storey biomass	-0.16*	-0.17*	-0.08 ns	-0.01 ns	0.09 ns	-0.11 ns

Light intensity and forest canopy density, which were strongly inter-correlated (Pearson r = 0.98), had a stronger relationship with these biometric properties than any other environmental variable. Figure 4-5 reveals sigmoid patterns in the relation between light intensity and the five reproductive properties of *C. odorata*. All five properties reduced to almost zero at light intensity below 6.5 mJ m^{-2} day^{-1}.

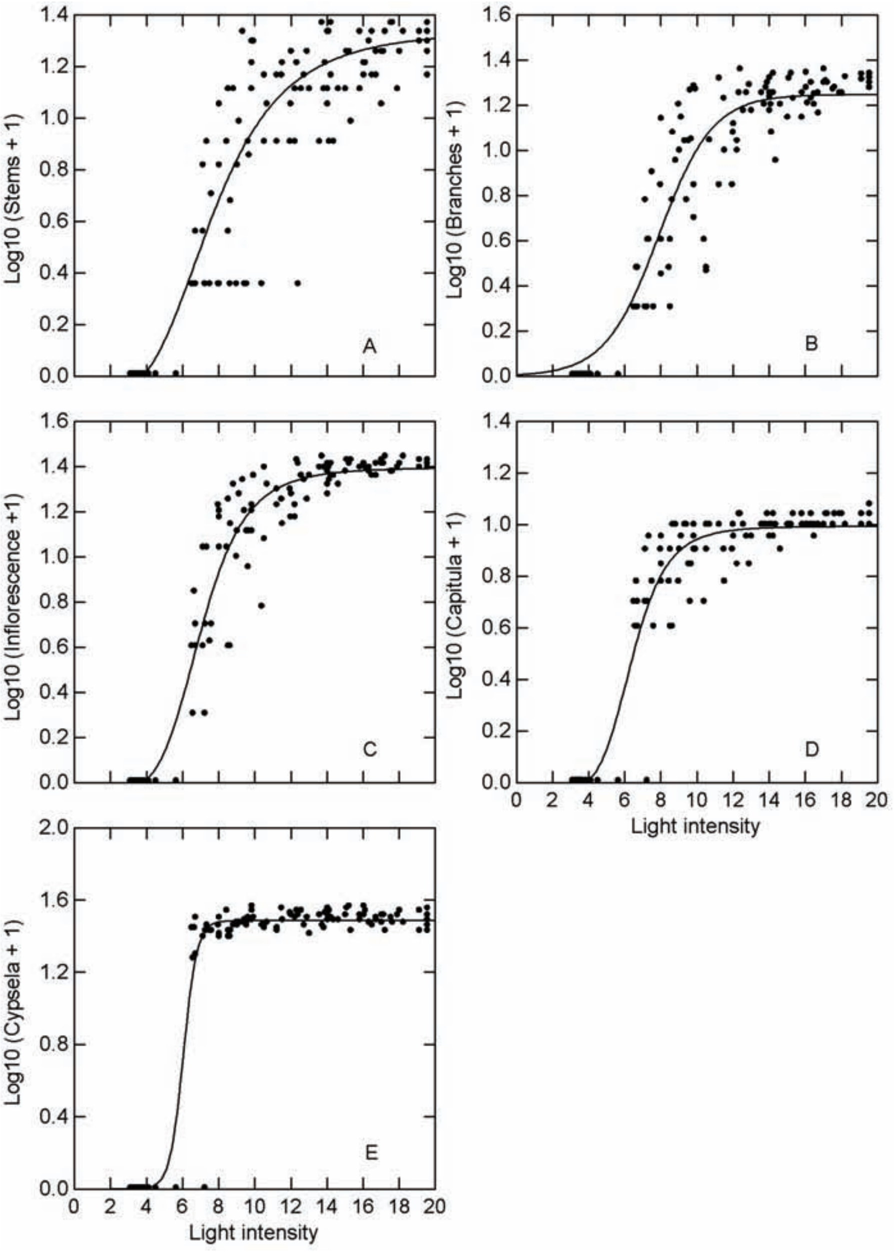

Figure 4-5. Relation between light intensity (mJ m^{-2} day^{-1}) and the log$_{10}$ of: A) number of flowering stems per plant, B) number of flowering branch per flowering stem, (C) number of inflorescence per flowering branch, D) number of capitulum per inflorescence and E) number of cypselum per capitulum.

Multiple multiplicative factor (MMF) and logistic models best described the relation between light intensity and *C. odorata* reproductive properties (Table 4-3).

Table 4-3. Regression equations of the best fitting non-linear model between light intensity (X) and reproductive properties of *Chromolaena odorata* (Y)

Dependent: Log$_{10}$ (Y+1)	Equation	Type	R2	S.E.
N flowering stem	= (-0.09*2645 + 1.34 * X3.86)/ (2645 + X 3.86)	MMF	0.86	0.17
N flowering branch/stem	= 1.24/(1+203.5*Exp(-0.67*X))	Logistic	0.86	0.17
N head/ branch	= (-0.04 * 65850 + 1.39*X5.69)/ (65850+X5.69)	MMF	0.92	0.13
N capitulum/ inflorescence	= (-0.03*238039 + 0.99* X 6.66)/ (238039+X6.66)	MMF	0.90	0.11
N cypsela/ capitulum	= 1.48/(1+9706186.4*Exp(-2.66*X))	Logistic	0.92	0.15

Hence, light intensity was significantly related to five independent biometric properties of *C. odorata*, which together determine its seed productivity. These biometric properties were combined in a reproductive index (see section 4.2.2) reflecting the seed production per plant. Figure 4-6 reveals that *C. odorata* did not produce seed below light intensity of 6.5 mJ m^{-2} day^{-1}.

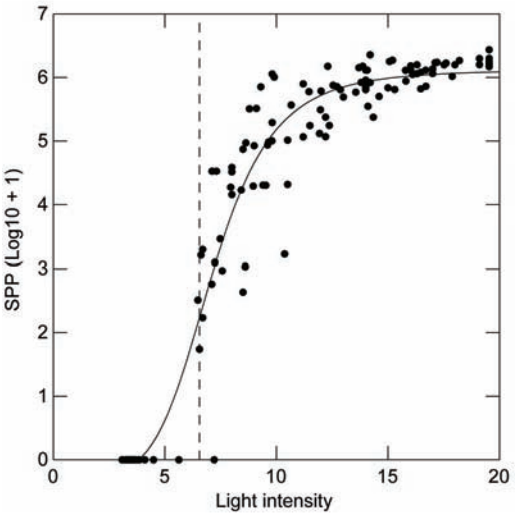

Figure 4-6. Relation between light intensity and the logarithm of the seed production per plant (SPP) of *C. odorata*. The solid line represents the selected MMF model, while the dotted line indicates the light intensity threshold of 6.5 mJ m^{-2} day^{-1} separating reproductive and non-reproductive populations of *C. odorata.*

The relation between the seed production and light intensity was best explained by an Multiple Multiplicative Factor model (Equation 4-5):

$$Log_{10}(Y+1)=\frac{-6624.39+6.11*X^{5.219}}{28590.4+X^{5.219}} \qquad \text{Equation 4-5}$$

where, Y is the seed production per plant and X is light intensity received at the forest floor.

The equation explained 93% of the variation in Log_{10} Y+1. Further analysis revealed that none of the other variables in Table 4-2 was related to the residuals of the seed-light model described in equation 4-5.

4.3.4 Other factors affecting seed productivity of *Chromolaena odorata*

The results presented above indicate that *C. odorata* produced seed at light intensities above 6.5 mJ m^{-2} day^{-1}. This does not imply however, that *C. odorata* was always present above or absent below this threshold. Out of 153 cases, *C. odorata* was absent in 22 cases where light conditions were suitable for seed production. In addition, the species was present in 17 cases where light conditions were unsuitable for seed production.

Why was *C. odorata* absent from sites suitable for seed production? Logistic regression (Equation 4-6) revealed that the presence of *C. odorata* (Y) in areas with light intensity above 6.5 mJ m^{-2} day^{-1} was significantly related to the biomass (X) of the forest understorey vegetation (-2LL = 44.08, df = 1, $p<0.0001$; McFadden's $Rho^2 = 0.38$):

$$y=\frac{\exp^{4.99-3.65*x}}{1+\exp^{4.99-3.65*x}} \qquad \text{Equation 4-6}$$

We classified the probabilities values predicted by the logistic regression into cases of 'predicted presence' and 'predicted absence'. To do this classification, a threshold probability had to be determined. We chose the threshold that produced the highest value for Cohen's Kappa. The threshold for classifying the predicted probabilities into cases of 'predicted presence' and 'predicted absence' was 0.74 and produced a Kappa value of 0.56. Table 4-4 presents the model confusion matrix which shows that the model predicts *C. odorata* to be absent from sites with a high biomass of other herb and shrub species.

Table 4-4. Logistic regression model confusion matrix

	Observed	
Predicted	Absence	Presence
Absence	17	13
Presence	5	86

4.3.5 Spatial prediction of cover and potential seed productivity of *C. odorata*

Figure 4-7A shows the cover percentage of *C. odorata* was predicted based on Equation. 4-4. Figure 4-7B shows the potential for seed production by *C. odorata* as predicted from light intensity using Equation 4-4 and 4-5 and Figure 4-7B shows the potential seed production based on *C. odorata* cover and light suitability map.

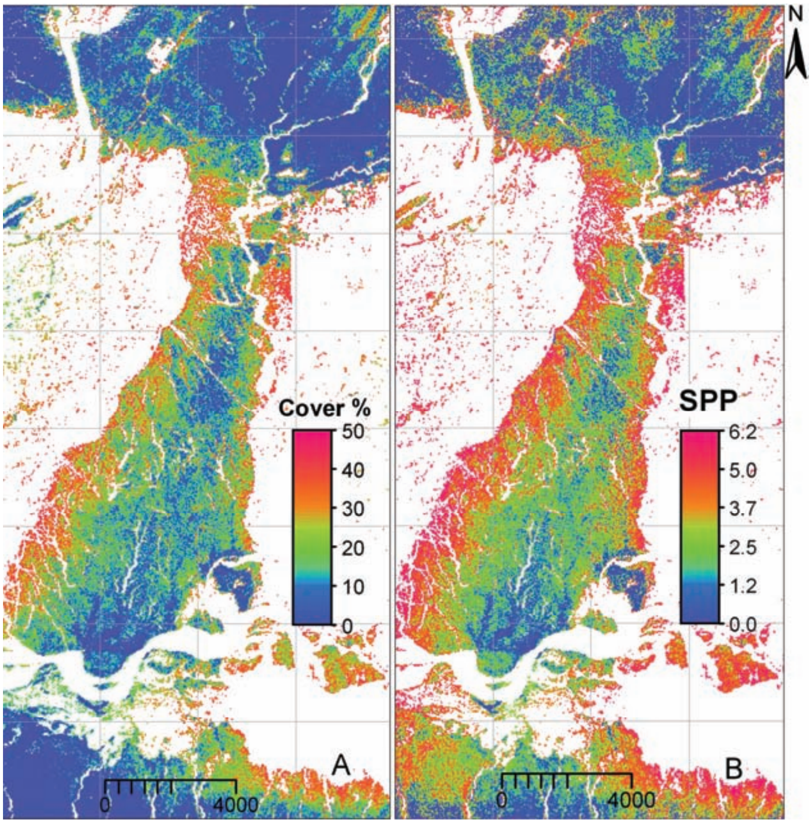

Figure 4-7. Map of A) potential cover % of *C. odorata* and B) Potential areas of seed producing *C. odorata.*

Our predictive map shows that reproductive populations of *C. odorata* currently present along forest edges and roads/trails, however it has a high potential to substantially increase its range. It could potentially invade large areas of the forest interior which are suitable for seed production.

4.4 Discussion

This study revealed a strong relationship between light and cover abundance and seed production of *C. odorata*. Light intensity influenced four out of five plant architectural characteristics determining total seed output per plant: the number of flowering stems, branches, the number of inflorescences and capitulum increased with light intensities. The number of cypsela (seed) per capitulum was genetically determined hence, not related to light intensity. These findings confirm Witkowski *et al.* (2001), who reported significant differences between sun, semi-shade and shaded sites for flower head, capitula and seed per plant but not seeds per capitula.

A positive influence of light intensity on the growth and seed production has been reported for various invasive species characteristic of forest understorey and forest clearings (Bush and Van Auken 1986, Silander and Klepeis 1999, Meekins and McCarthy 2000). Many of these species have a ruderal strategy (Grime 1979), characterized by high reproductive potential, excellent dispersal, fast germination, rapid initial growth and a high morphological and physiological growth plasticity. *C. odorata* combined all of these. The marked response to light of four out of five plant architectural characteristics demonstrated its extremely high morphological growth plasticity.

C. odorata thrived in areas with high light intensities, which corresponded to reduced canopy density. Light intensity under closed canopy in natural forest remained below the threshold for seed production. Open canopies with sufficient light reaching the understorey were predominantly found outside conserved forests. Here canopy density was reduced due to logging and lopping (Joshi, pers. obs.).

C. odorata was not always present in forests, that would otherwise have been suitable for seed production given their understorey light climate. Path analysis showed that grazing indirectly influenced the cover of *C. odorata*, most likely through its impact on the biomass of other understorey species. *C. odorata* was absent in the understorey of natural forest gaps in well conserved areas such as Chitwan National Park where native grass species reached a height of 2 m. Outside these well-conserved areas most of the forest was intensively grazed and the density and height of the other grass and herb species was generally much lower (Joshi, pers. obs.). We therefore suggest that these other species control and suppress *C. odorata* through competitive exclusion in the absence of grazing, whereas grazing breaks the dominance of these other species thus creating the conditions for unpalatable *C. odorata* to rise to dominance. Following this, we further suggest that the intensive grazing of the *S. robusta* forest in the Nepalese *Terai* may well have been a secondary factor responsible for the invasive success of *C. odorata*.

Our results indicate that *C. odorata* was frequently observed in forest too shaded for seed production. Anatomical analysis of the corm (an underground swollen stem base specialized for food storage, detail in Chapter 5) exhibited continuously developed increment zones (Figure 4-8).

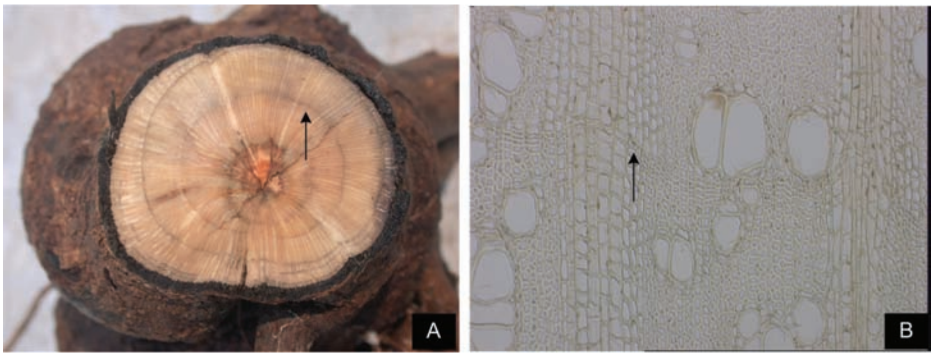

Figure 4-8. Cross sections (A) and radial sections (B) of the xylem of corm of *C. odorata*. Increment zones (arrows) are labeled ray parenchyma with reduced cell lumina and dark stained accessory compounds within (ray parenchyma).

Counting the number of increment zones found in the xylem at the shoot-root transition zone of plants indicated the annual formation of “tree rings” in the xylem of *C. odorata*. Tree ring analyses revealed that plants recorded under these conditions were mostly young and few old plants were found sterile. The presence of *C. odorata* under conditions unfavourable for seed production implies continuous re-colonization of such sites from adjacent seed producing populations. Blackmore (1994) reported that wind disperses seed up to 80m as short distance dispersal agent, while animals, people and vehicles transport seeds over significantly greater distances. The observation that *C. odorata* distribution was generally restricted to the first few metres along tracks in dark forest supports the importance of local people and domestic as well as wild animals as long-distance dispersal agents and forest canopy density as a factor limiting the invasion ability of *C. odorata*.

The above indicates that *C. odorata* maintains sterile populations in shaded forests, which are maintained through continuous colonization from adjacent seed producing populations. This implies that *C. odorata* is locally segregated into source and sink populations. The source-sink concept (Pulliam 1988) is interesting when applied to invaders, since source populations pose a risk of further spread and infestation while sink populations do not. Sink populations could rapidly develop into source population if trees are removed.

Our results indicated that *C. odorata* thrives in relatively open forest with sufficient light penetration, and suggested that competitive exclusion of *C. odorata* by other understorey species is prevented by grazing. The canopy of well-protected natural *S. robusta* forest is closed and its understorey hardly grazed. We suggest that it is the lopping and logging of the trees and grazing by livestock which degrade the forest canopy and understorey and creates conditions which allows *C. odorata* to dominate. On the one hand, it could be argued that the success of *C. odorata* depended on the degradation of the canopy and understorey of these forests, which created optimal conditions for *C. odorata*. Following this, it could be argued that the condition and degradation of the environment played a primordial role. On the other hand, it could be argued that seed dispersal and a strong reaction of seed production to light conditions point at an important role of the inherited traits of *C. odorata*. We think that the degradation of the forest environment which created the conditions allowing *C. odorata* to express its latent invasiveness. Hence, it could also be argued that degradation created an invadable environment which offered *C. odorata* to express its latent invasible traits.

Although, Civco (1993), reported that neural networks were under certain circumstances less accurate than conventional approaches for classifying land cover, several researchers have demonstrated that neural networks allow the combination of multi-source spatial data into the same classification or estimation (Benediktsson *et al*. 1993, Wilkinson 1993). A potential seed production map was created using a neural network, an approach generally leading to more accurate results than other traditional land cover classification techniques (Hepner *et al*. 1990b, Bischof *et al*. 1992, Atkinson and Tatnall 1997). The seed production of *C. odorata* was predicted from maps displaying light intensity reaching the forest understorey.

Perhaps the broadest use of remote sensing has been to identify and map vegetation types. In this study, we mapped the degradation stages of one particular vegetation type, rather than multiple separate vegetation categories. In forestry, remote sensing has been used for a long time to monitor deforestation (Tucker *et al*. 1984). More recently, however, it has been recognized that gradual forest degradation is a process deserving equal attention (Souza Jr. *et al*. 2003). Remote sensing techniques to map forest canopy density have been developed in response to this. Approaches to map forest canopy density, produced categorical maps with two (Boyd *et al*. 2002) or more classes (Rikimaru 1996, Rikimaru *et al*. 2002) rather than a continuous variable. We argued in chapter 3 that canopy density should be treated as a continuous rather than discrete variable.

This study indicates that an indirect approach could be very successful to predict life cycle attributes of invasive species. This success could possibly be due to a number of factors such as *C. odorata* being a shade tolerant species, the strong relationship between seed production and light intensity proved surprisingly strong and light intensity and the ability to reliably predict light intensity. We further suggest that there are possibly many more species that react in similar ways and the success of this approach might well be applicable to other cryptic invasive species.

We demonstrated the possibility to map and localize source and sink populations in plant species using remote sensing and GIS techniques. The reproductive success in many species is determined by physical factors such as light, moisture or nutrient availability. Remote sensing offers continuously improving possibilities to map such physical variables across landscapes (Baker *et al*. 2000, Corsi *et al*. 2000, Guisan and Zimmermann 2000). We therefore suggest that there is wider application for the indirect mapping approach described in this study. This could be helpful in invasive species management to direct interventions aiming at eradication or control to those areas which produce viable seeds and pose the greatest risk for further spread.

The findings of our study illustrate how remote sensing and GIS technologies can provide ecologists and land managers with an innovative perspective with which to study the factors influencing the patterns of invader population dynamics at local landscape scales. Furthermore, We suggested that forest canopy density, inter-specific competition and anthropogenic factors might account for the differences in size and productivity of *C. odorata* populations.

CHAPTER FIVE

Morphological plasticity and age structure of corms in enhancing invasion of *Chromolaena odorata*

This Chapter is based on:
C. Joshi, J. De Leeuw, J. van Andel, A. K. Skidmore, O. Dünisch and I. C. Van Duren. Spatial variation, morphological plasticity and age structure of corms in enhancing vegetative invasion of *C. odorata*: Submitted to the Biologycal Conservation.

5 Morphological plasticity and age structure of corms in enhancing invasion of *Chromolaena odorata*

Abstract

In this paper, we investigate the vegetative growth of *Chromolaena odorata* and the influence of light intensity on the understorey environment of *Shorea robusta* forest at Chitwan on south-central *Terai*, Nepal. *C. odorata* is a clonally growing shrub and typically consists of several clones with an underground "cormous organ" (a modified stem to store food reserve; here after "corm") belonging to the identical genet. In *C. odorata*, the biomass of such corms varied across the light gradient. The number of shoot described a strong logarithmic relation with biomass of corm. Under open forest canopy environment, corm biomass was strongly correlated with the number of shoots and the corm's age. However, under dense forest canopy, there was no significant relationship between corm biomass and its age. This result shows that corms of *C. odorata* were capable of maintaining their viability for a long period even under closed canopy environment. Any disturbances in forest canopy density would ultimately trigger its clonal growth capability. This plasticity of corms appeared to be a key strategy for invasion success of this species. Comparison of these results further indicates the importance of canopy density in determining invasion success of *C. odorata*.

Key words: Corm anatomy, "corm ring", *Chromolaena odorata*, clonal growth, invasion, light intensity, plant age, size plasticity

5.1 Introduction

The establishment of invasive plant species starts with the arrival of propagules, typically in the form of seeds (Ridley 1930). Once established, invaders may attain local dominance through sexual or vegetative regeneration (Cousens and Mortimer 1995). It has been repeatedly acknowledged that vegetative reproduction may be as valuable as seed production, as it makes the species less dependent on germination chance (Bunting 1960). Vegetative growth facilitates the penetration into adjacent vegetation under more stressful conditions (D'Antonio 1993).

Clonal growth is a very common type of vegetative regeneration and dominant growth form in many ecosystems (Williams 1975, Harper 1977, Cook 1983, Callaghan *et al*. 1992, Sachs and Novoplansky 1997). Clonal species penetrate into un-colonized areas through dispersal of stolons, rhizomes, tubers or corms (Sachs and Novoplansky 1997). Clonal growers are typically strong competitors (Auge and Brandl 1997, Pysek 1997) The combined ability of spreading and rising to dominance makes clonal growers invaders with a potentially strong influence on their direct adjacency. However, the role of clonal growth in invasiveness and invasibility has also been little explored (Alpert 1995, Shumway 1995, Pysek 1997).

Environmental conditions influence the production of clonal dispersal organs (De Kroon and Hutchings 1995, Sachs and Novoplansky 1997). Under suitable conditions, above ground parts allocate resources to produce clonal reproductive organs. Individuals growing under less favourable environmental conditions may not be able to allocate sufficient resources. Hence, heterogeneity in environmental conditions might generate spatial variation in the relative growth rate of clonal reproductive organs (Hutchings and De Kroon 1994, Hunt and Cornelissen 1997). Spatial variation in environmental conditions are thus expected to result in spatial variability of the rates of clonal growth of plants, particularly when factors are involved influencing their ecophysiology.

Native forest understorey species live under shaded conditions. Opening of forest canopy, due to logging or natural disturbances results in increased light intensities and change in species composition. Typically ruderal and heliophytic species rise to dominance when light intensity increases (Bayfield 1980, Appleby 1998, Kobayashi and Kamitani 2000). A number of exotic plant species such as *Chromolaena odorata* have successfully established as invaders under conditions of degraded canopy and forest clearing. Several of such clonal growers have been spreading vegetatively after initial establishment through seeds. The increased light intensities are postulated to trigger their rate of clonal growth.

Remote sensing has frequently been used to map invasive species. Applications have been restricted to localizing the distribution of invasive species (Everitt and Nixon 1985, Ustin *et al*. 2002). It has been argued that it would also be useful to a manager to know where invasive species spread into unoccupied areas through sexual or vegetative reproduction. In chapter 4, we demonstrated that it was possible to map seed production of a forest understorey species *Chromolaena odorata*. So far, no attempts have been made to map the rate of clonal growth in invaders.

Chromolaena odorata (L.) RM King & Robinson (King and Robinson 1970) is one of the world’s one hundred worst invasive species (ISSG 2004) Originating from

Central America it now occurs in the humid tropics around the world (Muniappan 1988, Gautier 1992b, Kriticos *et al*. 2005). It has a ruderal strategy and occurs in a wide range of environments including road verges, neglected agricultural fields and as an understorey species in forest. As a heliophyte (Gautier 1992a), it requires sufficient light to grow and produce seeds (Witkowski and Wilson 2001). Because of this it becomes invasive in forest where the canopy has been broken up (De Rouw 1991).

C. odorata plants in Nepal are composed of one or more genetically identical clones (Figure 5-1). Every clone has a root system supporting a corm (an underground swollen stem base specialized for food storage) from which shoots emerge (Figure 5-1, left). Buds located belowground on the corm allow *C. odorata* to escape the dry season with its frequent fires. This strategy to escape the adverse season classifies *C. odorata* as a geophyte (Raunkiaer 1937).

Figure 5-1. Morphology of *C. odorata* consisting of a single clone (left) and a genet of multiple clones (right). Photos taken at start of the rainy season show the corm, previous seasons' and newly emerging shoots.

Single clone *C. odorata* plants develop into genets (Kays and Harper 1974) of multiple clones. After a number of years clones decay and split into two or more new clones spaced typically 5 to 10 cm apart. Older plants because of this consist of a dense crowd of clones together forming one genet that may reach up to several meters in diameter (Figure 5-1, right). Stems of *C. odorata* grow up to 2 m in height.

In this paper, we investigate and model the biomass of corms of *C. odorata* to light intensity and corm age in *Shorea robusta* forests in Southern Nepal. Corm age was established using "corm rings", which we describe in this article. We next use remote sensing image to predict light intensity in the forest understorey. Finally, we predict the spatial variation in clonal growth rate from the light intensity map.

5.2 Methods and materials

5.2.1 Study area

The study was executed in a tropical forest corridor north of Chitwan National Park (27° 31´ to 27° 44´ N and 84° 24´ to 84° 31´ E), 140 km SW of Kathmandu, Nepal. The forest was dominated by *Shorea robusta*, while the understorey was invaded by *C. odorata*.

5.2.2 Data collection

The study area was visited twice. In September and October 2003, 275 plots were visited to measure forest canopy density percentage and light intensity at 1.2 m above the ground using hemispherical photography taken with fisheye lens attached to a Nikon digital camera.

In April-May 2005, we collected 237 corms of *C. odorata* plants. We measured the diameter of the corm and recorded the number of new shoots. The number of rings in the corm was counted on a transverse section. The corms were dried for 40 min in a microwave and dry biomass (g) measured. Morphology and phenology of *C. odorata* were described following several years of intensive study of the species.

Meteorological data on the monthly maximum and minimum temperatures, relative humidity, precipitation and evapotranspiration recorded for an average of 36 years at Rampur weather station were obtained from the Department of Meteorology, HMG, Nepal. Map of cover abundance of *C. odorata* was used from chapter 4.

5.2.3 Anatomical analyses

The anatomy of corms was further investigated at the Federal Research Centre for Forestry and Forest Products, Institute for Wood Biology and Wood Preservation, Hamburg, Germany. The anatomical structure of the xylem of the shoots and of the corms was investigated by light microscopy in order to prove the suitability of tree ring analyses for age determination in *C. odorata*. The shoot/corm was divided into segments of 1 cm length each. The samples were fixed in a FAA solution. The samples were embedded in polyethylen glycol (PEG 1500) with increasing concentration (PEG 1500:H2O, 1:2, 1:1, 2:1, 1:0, 1:0). For light microscopy and for the histometrical analyses of the xylem cells, transverse and radial sections (10 µm) were prepared by a microtome (Reichert, Austria). For light microscopy unstained and stained (safranine/astrablue (1:1)) slides were used. Histometrical measurements were carried out with an image analyser.

5.2.4 Data analysis

Total light intensity was calculated from the digital hemispherical photographs using Gap light analyzer software version 2 (Frazer *et al*. 1999, Frazer 2001). For ANOVA, we classified the light intensity in the forest understorey into three categories: high or full light (above 13 mJ m^{-2} day^{-1}, moderate light or semi-shade (7-13 mJ m^2 day-1) and low light or full shade (below 7 mJ m^{-2} day^{-1}).

We used two way ANOVA to investigate to what extent the biomass of *C. odorata* corm was related to light intensity and its age. We used curve expert (Hyams 2005) to select the best fitting curve describing the relation between number of shoots and biomass of the corm.

5.2.5 Mapping Light Intensity

Field measurements on light intensity were combined with an ETM+ image to produce a map of light intensity in the forest understorey. A three-layer feedforward backpropagation artificial neural network (ANN) consisting of an input, a hidden and an output layer was used for image classification (Skidmore *et al*. 1997). The Landsat ETM+ image of October 24, 2001 was registered with subpixel accuracy (±17m). The first seven bands were used as input to the network. Light intensity calculated from the hemispherical photographs was used to train the ANN. The total data set of 275 samples was randomly divided into two groups. One subset of 138 samples was used for training and the other 137 samples for testing. The best results were obtained with a learning rate of 0.9, a momentum of 0.7 and two hidden nodes. Finally 20 iterations of 7000 epochs were performed (the RMSE stabilized after approximately 7000 epochs) and we selected the best classification based on least root mean square error (RSME). Forest canopy density in percentage was also calculated using the same data set and procedure.

5.3 Results

5.3.1 Climate and *Chromolaena odorata* phenology

The season of heavy rainfall with precipitation higher than evapotranspiration extends in southern central Nepal from July to September (Figure 5-2). Significant rains may start in May and extend until November. The dry season extends from December to May. The season of active growth extends for *C. odorata* from May to October. Flowering starts in December and seeds ripen in February. After seed ripping, the stems die and drop off. Forest fire frequently destroys this decayed above-ground growth. New shoots develop from the resources stored in the corm in April, just before the start of the rainy season. If not destroyed by fire old stems may persist. Old bushes thus form a tangled mass of old and new stems.

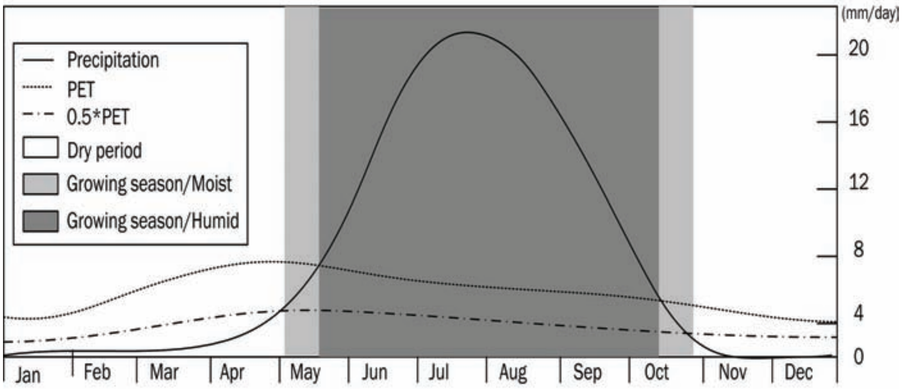

Figure 5-2. Monthly mean (1968-2004) precipitation and temperature for Rampur station (27°37' N, 84°25' E). The numbers on the vertical line denotes degree Centigrade for the Temperature (mean temp) and centimetres for Mean Precipitation (Mean prec), Maximum Precipitation (Max prec), Minimum Precipitation (Min prec) and Evapotranspiration (Evapotrans) for the curves.

5.3.2 Corm anatomy

Distinct increment zones were found in the xylem of the shoot and in the xylem of the corms of *C. odorata*. Increment zones in the xylem of *C. odorata* are labeled by fibres and ray parenchyma with reduced cell lumina as well as by wider rays (Figure 5-3a-d). The shoot/corm samples exhibited continuously developed increment zones (Figure 5-3b, d). Counting the number of increment zones found in the xylem at the shoot-root transition zone of plants of known age indicated the annual formation of "corm rings" in the xylem of *C. odorata*. Corm ring analyses showed that the oldest plants considered in our study had an age of 5 years. However, due to a very inhomogeneous pattern of ring width over the stem cross section and along the corm axis, corm ring analyses was not suitable to quantify the biomass allocation in the corms on an annual scale.

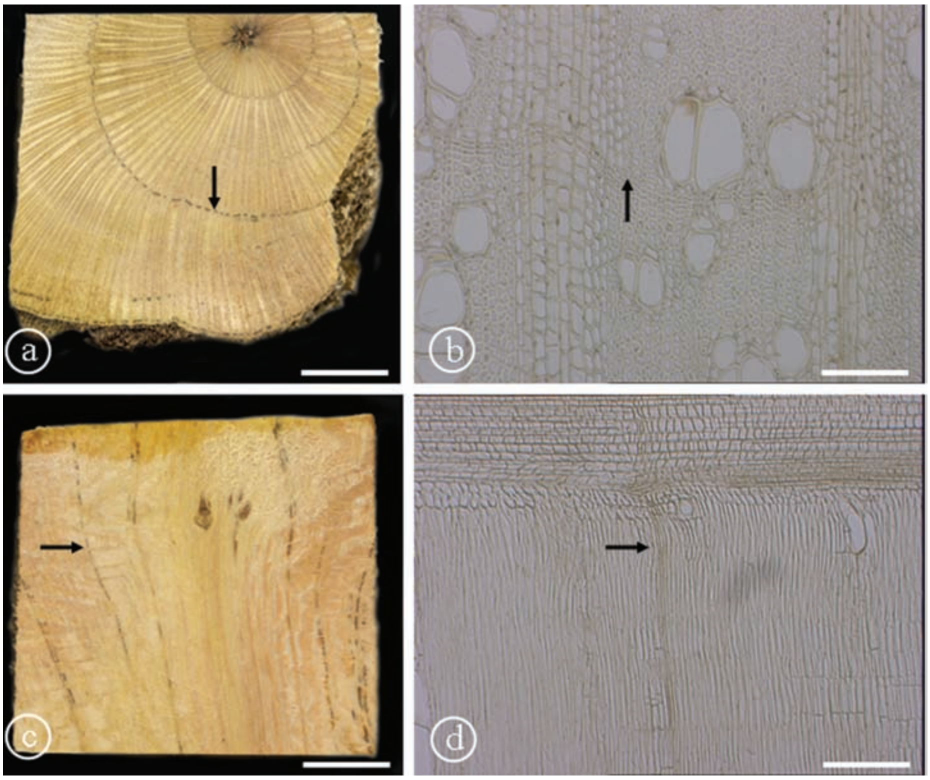

Figure 5-3. Cross sections (a/b) and radial sections (c/d) of the xylem of corm of C. odorata (sample 107). Increment zones (arrows) are labeled fibres and ray parenchyma with reduced cell lumina and dark stained accessory compounds within (ray parenchyma). Scale bars: (a/c) = 500 µm, (b/d) = 100 µm.

5.3.3 Corm and shoot dynamics

The maximum age of corms observed under full light was five years. Parts of these corms had decayed, leaving fragmented living tissue supporting vital shoots. Maximum observed corm age under full shade was also five years. However, these corms looked rotten, with little live tissue remaining.

Figure 5-4 shows corm biomass in relation to age and light intensity. Log transformation was used because it equalized group standard deviations, which originally varied by more than an order of magnitude. The figure suggests that corm biomass increases when plants age under full light or semi shaded conditions, but not under full shade.

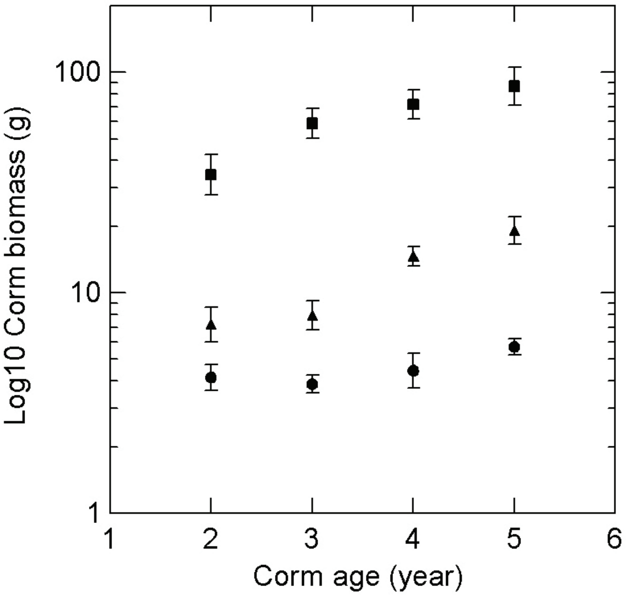

Figure 5-4. Corm biomass in relation to age of *C. odorata* and light intensity: ■ = full light, ▲ = semi-shade, ● = full shade.

This was confirmed by analysis of variance (Table 5-1). Both light and corm age had a highly significantly effect. The significant interaction implied that the increase of corm biomass with age changed with light intensity. While corm biomass differed significantly between two, three, four and five year old plants in the semi-shade (one-way ANOVA; p = 0.000) and in full sun-shine (p = 0.003), it did not in the full shade (p = 0.22). Hence, corms do apparently not grow in full shade while they do under semi shaded and lighter conditions.

Table 5-1. ANOVA of Log_{10} corm biomass in relation to corm age and light intensity

Source	SS	df	MS	F	P
Light	55.08	2	27.90	330.29	0.000
Corm age	3.14	3	1.04	12.40	0.000
Light * corm age interaction	1.17	6	0.19	2.32	0.033
Error	19.05	225	0.08		

One might expect that larger corms support larger plants. Figure 5-5 reveals that the number of shoots (y) produced by of *C. odorata* increased with the biomass of its corm (x). The relation was best described by the following Horel model (Equation 5-1), that explained 82% of variation in shoot number:

$$y = 2.501 * 1.001^{x} * x^{0.266} \qquad \text{Equation 5-1}$$

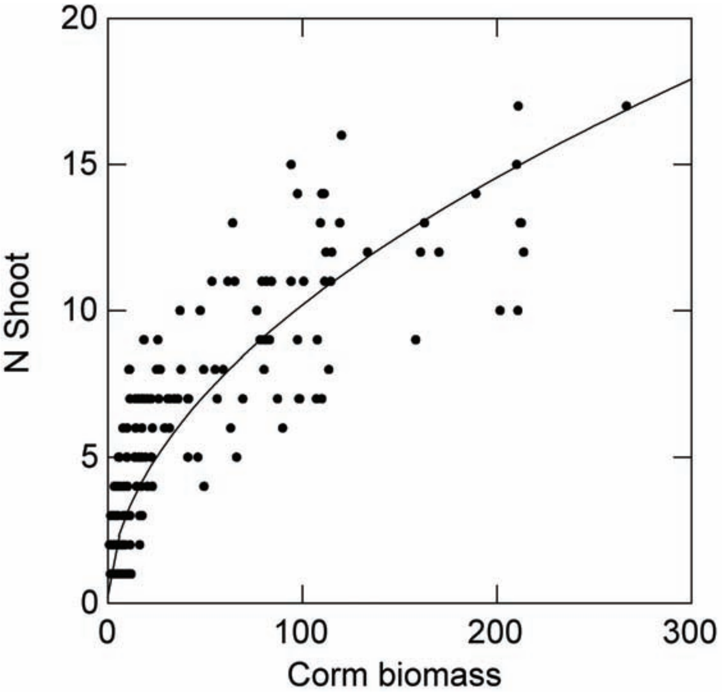

Figure 5-5. Relation between corm biomass (g) and number of shoots produced by *C. odorata*.

If low light constrains the growth of corms and subsequently shoot numbers, one would expect no enlargement of shoot numbers over time under low light conditions. Figure 5-6 reveals that this is the case indeed. The annual rate of increase of shoot numbers was significantly related to light conditions. Number of shoots increase with age only under semi shaded or full light conditions.

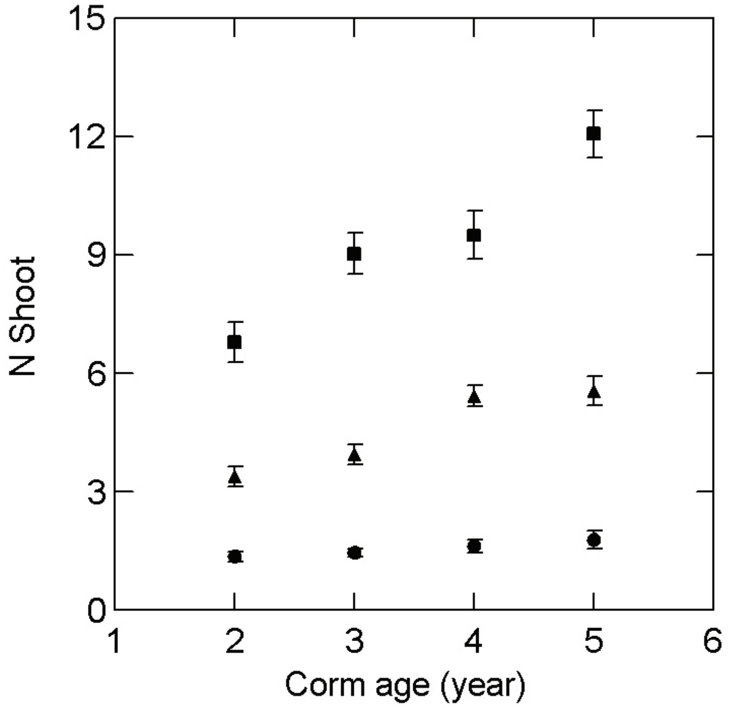

Figure 5-6. Relation between shoot number in relation to plant (corm) age and light intensity: ■ = full light, ▲ = semi-shade, ● = full shade in *C. odorata*.

Table 5-2 conformed that there is no significant difference in shoot numbers between two, three four and five year old shaded plants. While number of shoots differed significantly between two, three, four and five year old plants in the semi-shade (one-way ANOVA; p = 0.000) and in full sun-shine (p = 0.000), it did not in the full shade (p = 0.29). These results indicate that number of shoots almost not increase in plants growing in the shade. It does increase for plants growing under semi-shade and full sunshine.

Table 5-2. ANOVA of number of shoots in relation to corm age and light intensity

Source	SS	df	MS	F	P
Light	2517.13	2	1259	429.76	0.00
Corm age	220.36	3	73.5	25.08	0.00
Light * corm age interaction	148.31	6	24.7	8.44	0.00
Error	658.92	225	2.93		

5.3.4 Mapping forest understorey light intensity

Figure 5-7 and Figure 5-8A presents the performance of the artificial neural networks in scatterplots of observed versus predicted light intensity and the map produced by ANN classifier. It revealed that the prediction of light intensity by the artificial neural network was unbiased. The t-statistic of the slope (b) and intercept (a) reveal that these did not differ significantly from 1 and 0 respectively (a = - 0.002, SE_a = 0.005, t_a = - 0.361, b = 1.003, SE_b = 0.035, t_b = 0.086, r^2 = 0.81).

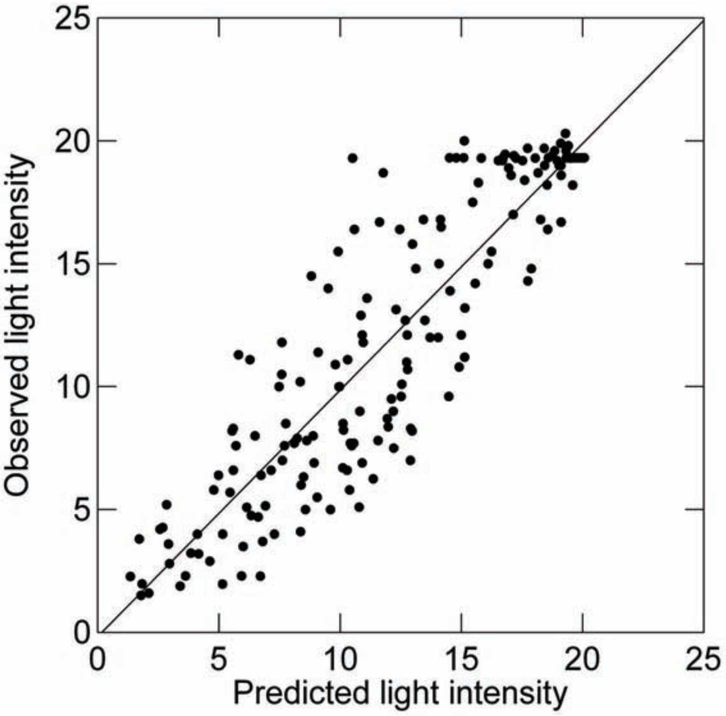

Figure 5-7. Scatterplot of light intensity (mJ m2 day-1) reaching the forest floor observed in the field, against the predicted light intensity derived from a classification of a Landsat ETM+ image.

5.3.5 Spatial prediction of rate of clonal growth of *C. odorata*

Number of shoots per year produced by the plant was utilized for spatial prediction of rate of clonal growth of *C. odorata*. The relation was best described by the following modified power model (Equation 5-2) which explained 73% of variation in clonal growth rate:

$$y = 0.251 * 2.258^{x} \quad \text{Equation 5-2}$$

where, Y is rate of clonal growth and X is light intensity.

We inverted this model to predict rate of clonal growth of *C. odorata* in the Chitwan region of Nepal. Map (Figure 5-8B) revealed that the current clonal growth rate of *C. odorata* was high along the forest edge and road, however, it has a high potential to substantially increase its range towards forest interior. It could potentially invade large areas of the forest interior which are climatically suitable for clonal growth in terms of size and number of corms as well as increase in occupied areas by stems.

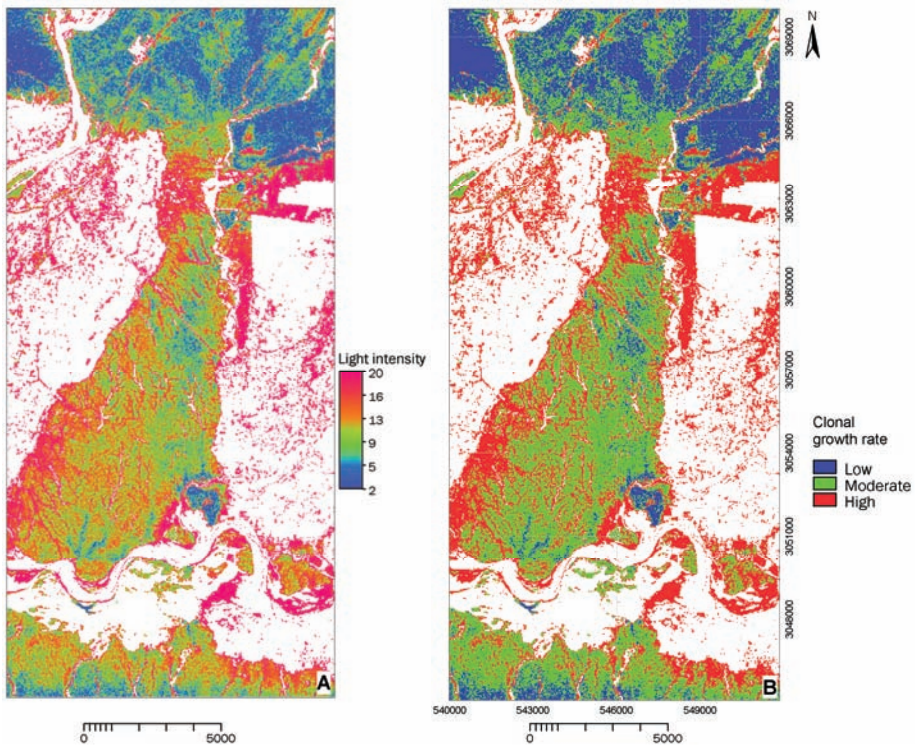

Figure 5-8. Map A) Light intensity (mJ m-2 day-1) reaching to the forest floor and B) displaying the potential clonal growth rate of *C. odorata*.

5.4 Discussion

The results presented in this study indicate that light intensity controls the growth of the corms of *C. odorata*. Our results further reveal that the size of the corms determines the number of next seasons' shoots. Plants growing under full light conditions thus enlarge their canopy, which in turn allows them to allocate more resources to the corm, which next year produces an even larger canopy. This positive feed back loop continues for a maximum of five years, after which the clone splits in two or more offspring starting the cycle anew. Over time, this leads to continuous enlargement of *C. odorata* genets.

Our results revealed by contrast that plants growing under shaded conditions do not enlarge their corm. Consequently, their number of shoots remains stagnant and hence no enlargement takes place. Our observation on the decaying condition of five-year old corms under shaded conditions suggests that they probably succumb within five years. The fact multi-clone genets were never observed under shaded conditions fits into this picture.

C. odorata thus manages to grow vegetatively under medium to full light conditions. These conditions rarely exist in natural forest. In forests, under shaded conditions one expects *C. odorata* populations to consist of young single clone genets, which is indeed the case. Under canopy openings however, *C. odorata* would receive sufficient light for vegetative expansion. Conditions of canopy opening widely occur in *S. robusta* forests in the Nepalese *Terai*. Not surprisingly, *C. odorata* populations here consist of multi-clonal genets, sometimes several meters in diameter. Figure 5-9 summarizes the effect of light intensity at different stages of the development of *C. odorata* linking to clonal growth and its vegetative invasion.

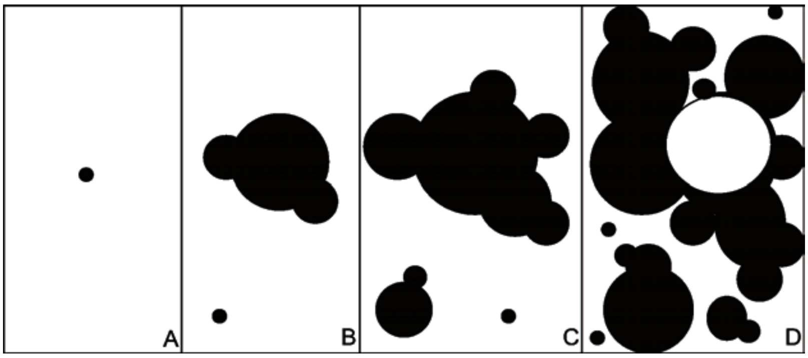

Figure 5-9. Spatial and temporal dynamics of corm of *C. odorata* under high light environment. A) An establishment of a mother corm in a new environment by long or short distance seed dispersal. B) Development and growth of daughter corms. C) Growth and range expansion of daughter corms of the identical genet. D) Death of mother corm and range expansion of new daughter corm populations.

What makes *C. odorata* such a successful invader in the *S. robusta* forests of the Nepalese *Terai*? *C. odorata* invades through its huge production of achenes, and its corm production and vegetative spreading were promoted by human disturbances through altering forest understorey light regime (opening up of forest canopy that benefit its competition and dominance among native plant communities). Furthermore, deliberate fire destroys native vegetation and most of the upper parts of *C. odorata* bush, leaving the basal stem unaffected. These stems regenerate shoots in the rainy season enabling the species to become the first dominant in the next growing season (Liggitt 1983) Grazing on neighbouring plant species further enhances its clonal growth by reducing inter-specific competition as the leaves and young shoots of *C. odorata* are poisonous to livestock (Sajise *et al*. 1972).

Comparisons of the traits of different invaders suggest that some species are more invasive in a particular environment than others. A wide range of species traits may underline these differences, including ruderal nature, wide seed dispersal mechanism, producing large amount of seeds, or allocation of resource during favourable conditions or long survival in the system (Grime 1979, Richardson and Bond 1991, Roche *et al*. 1997). All these characters interact with each other and with the environmental factor making it difficult to identify the cause of invasiveness in individual cases. However, our results show the primary importance of biomass allocation in *C. odorata* in determining spatial patterns of vegetative reproduction along light gradients. *C. odorata*'s ability to store energy enhances the early clonal growth in the next growing season. This competitive ability appears to be an important key to understanding its behaviour as an invader. If openings present in a stand, *C. odorata* becomes established and outcompetes previously established native species.

Mapping the rate of clonal growth could be well applied to other invasive species. Clonal reproduction in many species is determined by physical factors such as light, moisture nutrient availability or distance related seed dispersal mechanisms. Spatial distribution of these factors are increasingly available or can be mapped using remote sensing (Robinson and Valentine 1979, Dennis and Brusven 1993, Austin *et al*. 1996, Stone *et al*. 1997, Baker *et al*. 2000, Corsi *et al*. 2000, Guisan and Zimmermann 2000, Kerr and Ostrovsky 2003). Our study showed how these physical factors, may linked to map the clonal growth rate of an invasive species. Incorporation of remote sensing techniques with species biometry yield instantaneous, useful, cost effective, multi-scale and temporal information on clonal growth dynamics of an invasive species.

Mapping clonal growth rate in *C. odorata* population could significantly reduce control cost by allowing one to concentrate control efforts to manage habitats so as to curtail the spread of invasive species. In this respect, the immediate benefit of this research has been to contribute to the knowledge base of land managers by providing improved information on the rate of spatial and temporal spread of clonal population *C. odorata*, which will support efficient habitat ranking to restore invaded areas and protect non-invaded ecosystems.

CHAPTER SIX

Interrelated causes of invasibility of tropical forest ecosystems of Nepal

This Chapter is based on:
Chudamani Joshi, Jan De Leeuw, Andrew K. Skidmore, Iris C. van Duren, Jelle G. Ferwerda. Submitted to Agriculture, Ecosystems and Environment

6 Interrelated causes of invasibility of tropical forest ecosystems of Nepal

Abstract

The spread of invasive species has been attributed to intrinsic “aggressive” species traits, the extrinsic susceptibility of the recipient environment and the lifting of distribution barriers. People have stimulated invasions while lifting barriers and changing environmental conditions. In this paper, we investigate the hypothesis that environmental change as a result of disease control facilitated invasion. The study was executed in Chitwan district in central Nepal. Malaria control increased human population growth rates to levels exceeding 10% per year in the 1960’s. This resulted in land use change and scarcity of grazing lands outside forests. The increasing demand for grazing and wood resources resulted in an opening up of the forest canopy. It is deduced that the pressure to exploit the forests for grazing increased over time as a result of land use change and increased human population density. Our results indicate that the neo-tropical shrub *Chromolaena odorata* maintains invasive populations at canopy densities below 60%, which prevail in forests with opened canopies. We therefore argue that invasion of *C. odorata* accelerated a result of canopy degradation which was caused by change in land use and demography triggered by malaria control.

Key words: Landuse change, Malaria control, *Chromolaena odorata*, Population migration, Forest canopy

6.1 Introduction

Few exotic species become successful invaders (Veltman 1996). Groves (1991) estimated that while 10% of all alien species get established, only 1% become invasive. Lodge (1993) therefore concluded that exotic species differ in their ability to establish and invade local communities.

What makes an exotic species a successful invader? Some researchers stressed the importance of the intrinsic ability of species to invade (di Castri 1989, Williamson 1996, Kolar and Lodge 2001). Among the traits promoting establishment of populations are a ruderal strategy (Grime 1979), rapid germination, and initial growth rate and high production and efficient dispersal of seeds. Others (Groves and Burdon 1986, Macdonald *et al*. 1986, Mooney and

Drake 1986, Hengeveld 1989, Lonsdale 1999a) stressed the importance of propagule pressure and invasibility or the susceptibility of ecosystems to invasion.

Invasions do occur naturally (Botkin 2001, Schullery 2001), since plants and animals disperse across the globe. The rate of species invasion however has been greatly accelerated by humans. The most frequently cited anthropogenic enhancement of invasions is the moving of species across geographical barriers previously restricting distribution and by modifying the recipient ecosystem (Johnstone 1986, Hobbs 2000b, Vila` and Pujadas 2001). Several studies attributed the invasive success of exotic species to disturbance or degradation imposed by people (Elton 1958, Huston 1994, Hobbs 2000b). Knops (1995) reported a higher frequency of introduced plant species in disturbed than in undisturbed areas. Nowadays mankind influences ecosystems worldwide (Vitousek 1997) through land use change. Land use change in turn is caused by a complex interaction of demographic and socio-economic drivers, the study of which recently gained more attention (Pino 2005). So far, however, few attempts have been made to relate invasion, ecosystem degradation and land use change to these more distant socio-economic drivers. Land use change alters the existing environmental conditions which creates opportunities for invasive species to establish (Hobbs and Huenneke 1992).

In many parts of the world, disease control has resulted in rapid immigration, and penetration of the agricultural frontier into more natural ecosystems. Tsetse control in Ethiopia, for example, increased the rate of human migration and the progression of agriculture and settlements (Reid *et al*. 1997). In Zimbabwe, the rate of immigration was more than doubled by tsetse control (Govereh 1999). Though recent reports have described the relation between land use change following disease control (Muriuki *et al*. 2005), there is so far little evidence to support the hypothesis that land use change induced by disease control triggered the invasion of exotic species.

In this study, we investigate the chain of interrelated causes that made lowland forest ecosystems in Nepal susceptible to invasion. We describe how malaria control and government resettlement policies contributed to increased human population density that triggered changes in land use that resulted in the degradation of the forest environment. We finally outline how the degradation and opening of the canopy of lowland Nepalese forests facilitated invasion of a cryptic weed, *Chromolaena odorata*.

6.2 Material and methods

6.2.1 *Chromolaena odorata* (L.) R M. King & Robinson

Chromolaena odorata is ranked among the world's worst invasive species (ISSG 2004). Native to Central America it is currently found throughout the tropics of Asia and Africa (Muniappan 1988). It dominates in open and degraded forest, but remains virtually absent from closed canopy forest (Chapter 4; (Witkowski and Wilson 2001). *C. odorata* is a heliophyte, capable of producing millions of seeds under optimal light conditions (Erasmus 1985, Gautier 1992b, Witkowski and Wilson 2001). Dispersal over limited distances occurs through wind, while short hooks clinging seeds to animal fur, clothes, etc facilitate zoochoric dispersal over longer distances. *C. odorata* is a short-lived perennial with above ground parts dying off in the dry season and re-sprouting at the start of the rains from reserves of starch stored in its roots.

6.2.2 Study area

The study was conducted in Chitwan district, Nepal. The forests are dominated by *Shorea robusta*, a deciduous tree species shedding its leaves in February - March during the second part of the dry season. The district was chosen because the Nepalese malaria control programme was initiated here in 1955 (Jung 2001). Previously the district was densely forested, but rapid deforestation occurred shortly after the start of the malaria control. A degraded narrow forest passage in the middle of the district (Figure 6-1) is the last remaining corridor linking the Royal Chitwan National Park and Indian Wildlife Sanctuaries to the south to the wider pre Himalayan mountain ecosystem in the north. In recent years, the area has been invaded by *C. odorata*, which makes this forest corridor an ideal site for this study.

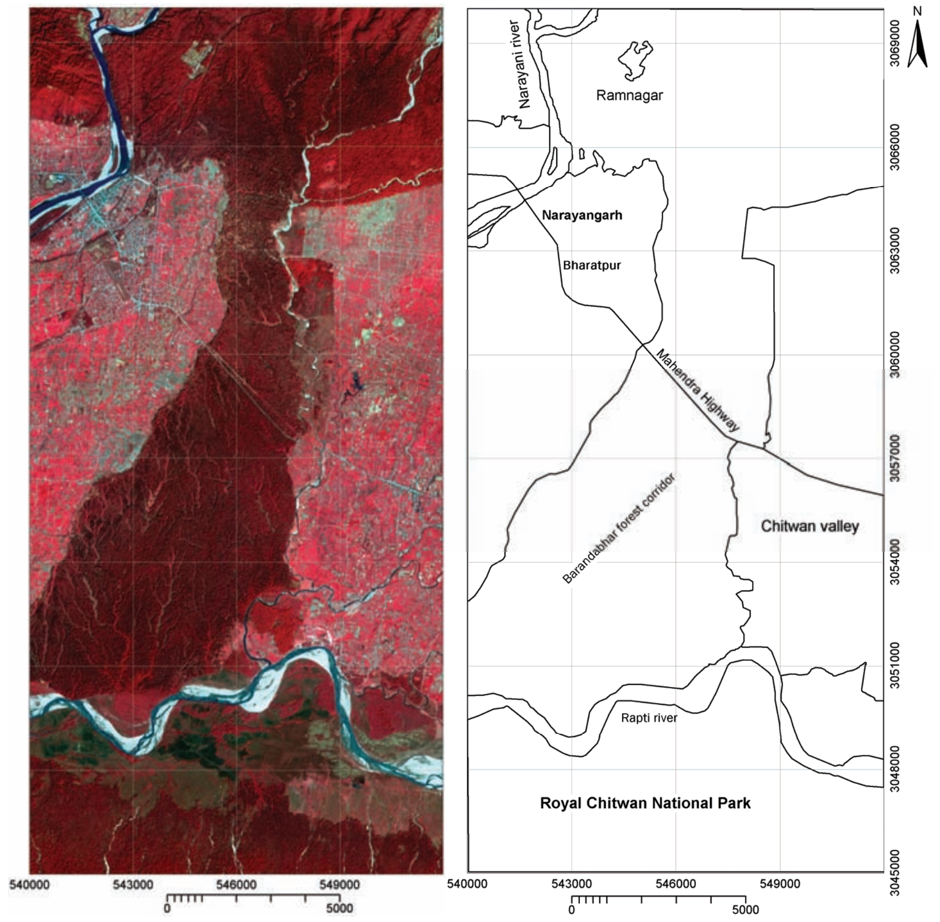

Figure 6-1. False colour composite (band 3, 2, 1) of Aster image of Feb. 25, 2005 (left) and topographic map (right) of the study area.

6.3 Data

6.3.1 Field observations

In September-October 2003 directly after the end of the rainy season when trees and *C. odorata* were in full leaves, we estimated forest canopy density (Chapter 3) and *C. odorata* cover in 153 randomly located plots of 30 by 30m. Furthermore, in 2005, a circular plot with an area of 500 m^2 was used to record the number of trees >10 cm diameter at breast height (DBH). From the centre of this larger plot, a smaller plot of 200 m^2 was constructed to record trees between 2-10 cm DBH. We also recorded number of cut tree stumps for sign of logging and looped tree branches for sign of lopping.

6.3.2 Maps and Imagery

Aerial photographs, land utilization maps, topographic maps, TM, ETM+, Aster satellite images and Corona satellite photographs (Table 6-1) were used to reconstruct the spatial and temporal change in land use and land cover in the study area.

Table 6-1. Satellite images, maps and aerial photographs used for the study and their date and source of acquisition and spatial scale or resolution

SN	Data type	Date	Scale/resolution	Rescale	Source*
1	ASTER 14 bands	25/2/2005	15m, 30m, 90m	15m	ITC
2	ETM+ 7 bands	24/10/2001	28.5m, 57m	30m	GLCF
3	Topographic maps	1994	25000	25000	DoS
4	TM 7 bands	12/10/1988	30m, 90m	30m	ITC
5	Land utilization map	1982	50000	25000	LRMP
7	Aerial photographs	1978	50000	12000	DoS
8	MSS 4 bands	28/10/1976	60m, 90m	30m	GLCF
9	Aerial photographs	1964	12500	12000	DoFR
11	Corona photograph	1962	1.04 m	15m	USGS
10	Topographic maps	1958	63000	25000	DoS

*Data source: ITC = International Institute for Geo-Information and Earth Observation, Enschede, the Netherlands; GLCF = Global Land Cover Facility site; DoS = Department of Survey, HMG Nepal; DoFR = Department of Forest research, HMG, Nepal; LRMP = Land Resource Utilization Project, DoS/Kenting Earth Sciences, Canada; USGS = U.S. Geological Survey, The National Map, EROS Data Centre

6.3.3 Demography and epidemiology

Data from population censuses executed between 1930 to 2001 and migration data from 1952 to 2001 were obtained from the Central Bureau of Statistics (CBS 1958, 1961, 1971, 1984, 1985, 2001) of Nepal. Data related to malaria were derived from the annual reports of Department of Health Services, Ministry of Health, HMG Nepal and from Jung (2001).

6.3.4 Interviews

During the field visit in February-April 2005, 5 retired forestry personnel and 29 old farmers who herded livestock in the 1950's and 1960's were interviewed. The following questions were asked to the farmers:

- Did you graze your animals in the forest prior to malaria control (1955) campaign? If not, why not?
- Where did your animals graze before they started grazing the forest?
- How many years after opening up the forest canopy did the composition of the vegetation change?
- What was the type and grazing quality of the vegetation when you first started grazing?
- Did you start to graze your animals in the forest because of shortages of grazing resources outside? If yes, what caused these shortages?

6.3.5 Map production

Various maps were produced to reconstruct the land use change in Chitwan district. Satellite images (Table 6-1) were geometrically corrected to the Nepalese National Grid (Projection: Transverse Mercator, Ellipsoid: Everest India 1830, False easting: 500000, False northing: 0, Central meridian: 84°E, Central parallel: 0°N, Datum: India-Bangladesh). They were re-sampled to 30 (Landsat) and 15 m (Aster) pixel size respectively. Radiometric and atmospheric corrections were applied (NASA 2005).

Stereo-pairs of aerial photographs of 1964 were used to estimate canopy density using a canopy closure template. The photographs covering the study area were than scanned, geo-referenced and glued into an orthophoto mosaic. Similarly, training samples of canopy density were generated from aerial photographs of 1976 and 1989 to classify MSS and TM images of 1976 and 1988 respectively. Forest canopy density maps for 2001 and 2005 were generated from an ETM+ image of 2001 and an Aster image of 2005 respectively using field observation data from 2003 and 2005 (detail in Chapter 3). All these various images were classified into seven land cover classes using eCognition 4.0 software that we found to be an appropriate tool for this study. The software is based on the concept that important semantic information necessary to interpret an image is not represented in single pixels but in meaningful image objects and their mutual relations for land cover mapping (eCognition 2003). First, image segments were defined and calculated. For each image, parameters were defined for the scale, shape and spectral properties. These image segments were calculated on several hierarchical levels in a “trial and error” process to result in final image segments to represent single objects of interest. The organization of the workflow is as follows: 1) Input images, 2) Multiresolution segmentation, 3) Image object hierarchy, 4) Creation of class hierarchy, 5) Classification using training samples and standard nearest neighbour, 6) Classification base segmentation, 7) Repeat steps for best result, and 8) Final merge classification (Laliberte *et al*. 2004).

Additionally, 36 Herbarium specimens of *C. odorata* housed at National Herbarium (KATH), Kathmandu, Nepal and Herbarium of the Natural History Museum (BM), London were examined. The date and place of collection recorded with each specimen was converted to a point map. The point map was subsequently converted to a raster map using point interpolation method (ILWIS 2005) which showed a chronology of the invasion of *C. odorata* in tropical regions of Nepal.

6.3.6 Data analysis

From the five measured biometric properties of a plant, the average seed production per plant (SPP) was calculated. Regression technique was used to analyze the relation between light intensity, forest canopy density and seed productivity of *C. odorata* (detail in Chapter 4). Two-sample t-test was used to analyse logging and looping variables, and Mann-Whitney U test for comparing stem densities within and outside the National Park.

6.4 Results and discussion

Figure 6-2 shows a strong relation between forest canopy density and the number of seeds produced per *C. odorata* plant as (Equation 6-1):

$$\log_{10} y = a - \frac{b}{1 + c * \exp^{d*x}} \qquad \text{Equation 6-1}$$

where, x = forest canopy density; y = seed production per plant, a = 6.47, b = 7.21, c = 648.51, d = 0.11; r2 = 0.86, SE = 0.79.

The seed production per plant differed six orders of magnitude between highly and poorly reproductive populations, a contrast which was also reflected by the size and architecture of plants. Up to one million seeds per plant were produced in areas with open forest canopy, while no seeds were produced in forest with a canopy density over 60% (Figure 6-2). We indicated (Chapter 4) that forest canopy density was strongly correlated (r= 0.98) to light intensity penetrating into the forest understorey. They also reported that 93% of the variation in seed productivity could be explained by light intensity alone. We further demonstrated that sterile *C. odorata* plants existed at light intensity below 6.5 mJ m^{-2} day^{-1}. These sterile populations can only be sustained through continuous immigration of seeds from elsewhere. Livestock and wildlife may well be the agents responsible for this seed dispersal. In chapter 4, we suggested that *C. odorata* is structured by forest canopy density into source and sink populations.

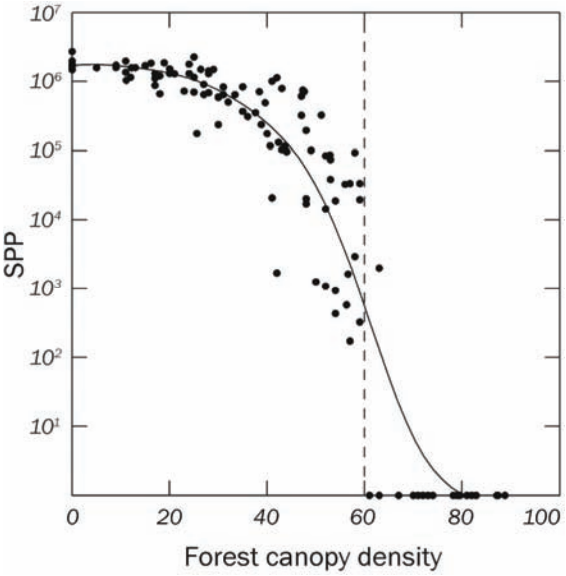

Figure 6-2. Relation between the number of seeds produced per plant (SPP) and forest canopy density (%). The dotted line indicates the canopy density threshold of 60% segregating reproductive and non reproductive populations of *C. odorata.*

The foregoing indicates that *C. odorata* maintains invasive populations at canopy densities below 60%. Figure 6-3 shows that now a days, canopy densities below 60% are very common in areas with open access to inhabitants such as the forests in the Barandabhar forest corridor. Analysis of the historic maps reveals that the canopy closer of the forests in this corridor was above 60% in the 1960’s (Figure 6-3A).

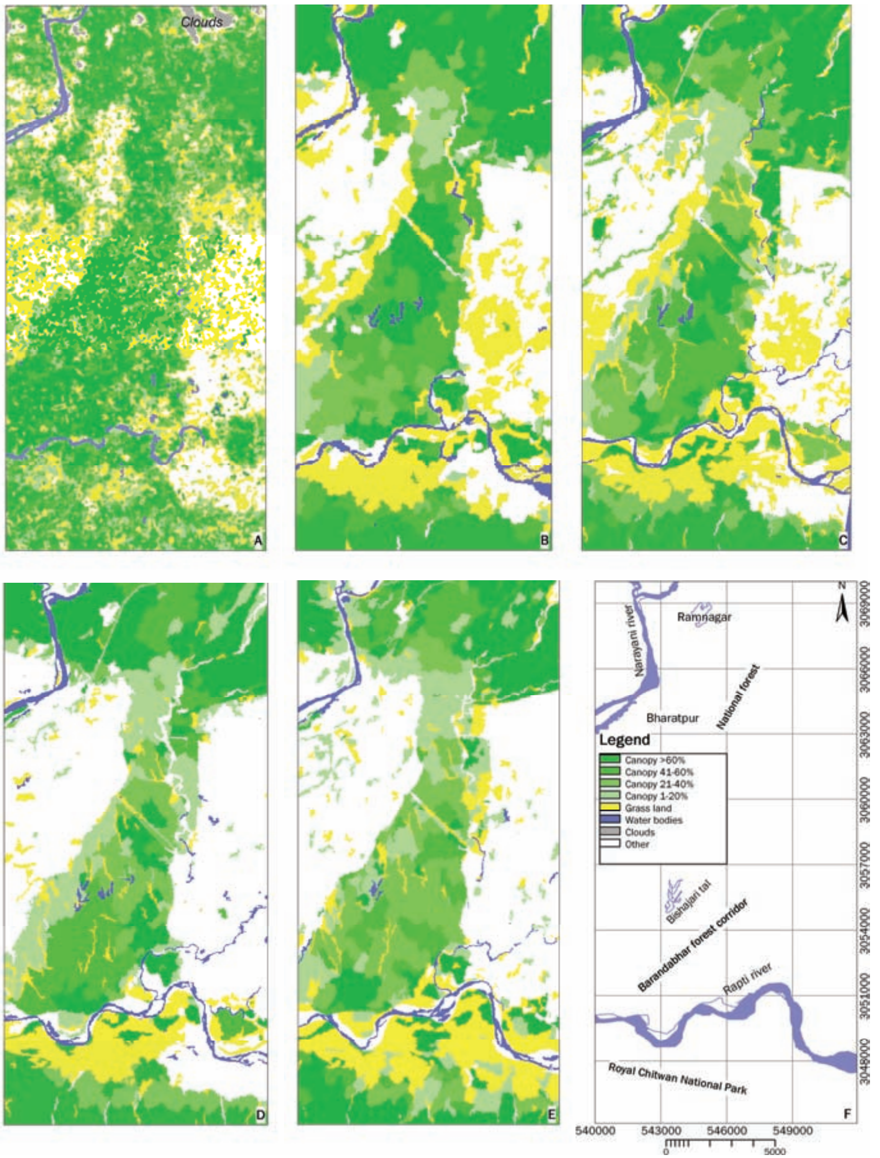

Figure 6-3. Land cover change in the Chitwan region between 1962 and 2005. Source data: A) Corona satellite photography 1962; B) Landsat MSS 1976; C) = Landsat TM 1988; D) Lansdat ETM+ 2001; E) = Aster 2005 and F) represents legends of all the maps.

Comparison of the later maps (Figure 6-3B, C, D and E) indicates that the canopy opened up since the early 1960's. This was confirmed through interviews with retired forestry staff employed in the area in the 1950 and 60's who reported that they witnessed the opening up of the *S. robusta* forests. They furthermore informed us that the legal protection of the dominant tree species *S. robusta* in 1961 had an un-anticipated side effect. The legal protection of *S. robusta* implied all other non-protected tree species were free to be logged, and thus caused the opening of the canopy.

Logging resulted in a decline of stem density per unit area. Hence, one would expect a relation between canopy density and stem density per unit area. Figure 6-4 confirms there is a positive relation between stem and canopy density. The regression equation is (Equation 6-2):

$$y = 62.24 * \exp *^{-0.0024 * x} \qquad \text{Equation 6-2}$$

where X = Number of stems per ha and Y = Forest canopy density, SE = 10.8, $r^2 = 0.73$.

The figure also reveals that forests inside Chitwan National Park had canopy and stem density of over 40% and 500 individuals per ha respectively while forest outside were more open and had lower stem densities.

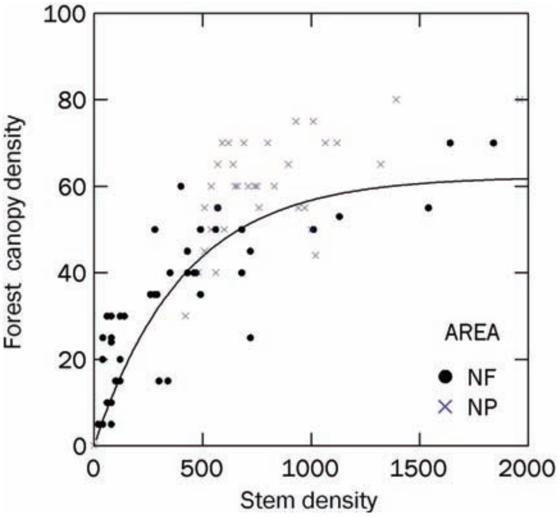

Figure 6-4. Relation between forest canopy density and stem density per hectare (DBH >2 cm) for *S. robusta* forests in Chitwan region (NP = Royal Chitwan National Park, and NF = National forest/Bharandabhar forest corridor area) of Chitwan, Nepal.

What exactly happened next in the 1960's and 1970's? Figure 6-3 reveals that the amount of arable land increased at the expense of forest and particularly grasslands. Interviews with farmers herding livestock in the 1950's and 1960's revealed that the recently opened forests attracted them because of the good quality of tall grasses. Farmers also indicated that, dry season grazing resources became more difficult to obtain because natural grasslands gradually disappeared (Figure 6-3). In response, they converted forest to grassland south of the Rapti river. The closing of access to these grasslands when they became part of Chitwan National Park in 1976 further reduced the area of available grazing land. This resulted in a severe shortage of grazing resources in the encroaching agricultural landscape and pushed the herdsmen to turn towards the recently opened forests.

Not surprisingly, 24 out of 29 interviewed herdsmen confirmed that they reduced the number of cattle because of these forage shortages. They also reported that the tall grasses, which initially dominated the forest understorey replaced by shorter species of lower grazing quality.

Under natural conditions, open forest closes through recruitment of trees. A comparison of tree sapling (DBH = 2-5cm) density in grazed (outside the National Park) and non-grazed (inside the National Park) areas revealed that the sapling density was significantly higher inside the National park (Mann-Whitney U test, test statistic = 404, P <0.001). We therefore conclude that very limited number of tree seedlings attend mature in grazed areas. The natural regeneration and growth of tree species in these sylvo-pastoral systems is effectively suppressed (Joshi, pers. obs.), possibly as a result of grazing of the understorey vegetation.

Canopy gaps could also be closed through enlargement of existing crowns. However, lopping of the trees for fodder or fuel wood prevented this. The percentage of trees with signs of lopping was significantly higher (14.2% versus 0%) in open access than in protected areas (two sample t-test, t = 5.64, p<0.01). The interviewed herdsmen confirmed that this practice was introduced in the 1950's and continues outside the National park to this day. Lopping provides another benefit, while the young leaves of *S. robusta* are highly nutritious fodder in a period at the end of the dry season when little other good quality forage is available.

Together, logging, lopping and suppression of regeneration due to grazing (Figure 6-5) thus well explain why the open canopy structure of the *S. robusta* forests has been maintained. It is not surprising that people started to graze their animals in the forest. Animals play an important role in the agro-ecosystems of the *Terai*. Bovine species are used as labour force and for the provision of milk. These animals grazed in the natural grasslands which according to the 1962 map (Figure 6-3A) fringed the edges of the forest. Today these grasslands have been transformed into rice paddies, while the number of animals increased along with the number of households. While paddies provide forage during harvest in early dry season, they fail to do so during the height of the dry season. Interviewed herdsman further stated that at the middle of the dry season most cattle graze inside the forest. The remnant forest patches in this intensifying paddy agro-ecosystem thus became increasingly attractive alternative grazing lands.

Figure 6-5. Logging (left), lopping (centre) and grazing (right) in the study area.

How was this canopy opening related to socio-economic drivers and why did it occur in this particular period? Until the 1950's, the *Terai* was densely forested and sparsely populated because of hyperendemic malaria (Peters *et al*. 1955, Brydon 1961, Darsi Jr and Pradhan 1990). The indigenous Tharu, noted for their resistance to malaria, occupied pockets of clearings in a matrix of malaria bearing forest. A national campaign aiming at control of malaria was started in 1955 in Chitwan district (Isaacson *et al*. 2001). The infection rate in the Nepalese *Terai* reduced from almost 90% in the early 1950's to much lower levels in the 1970's and thereafter (Figure 6-6).

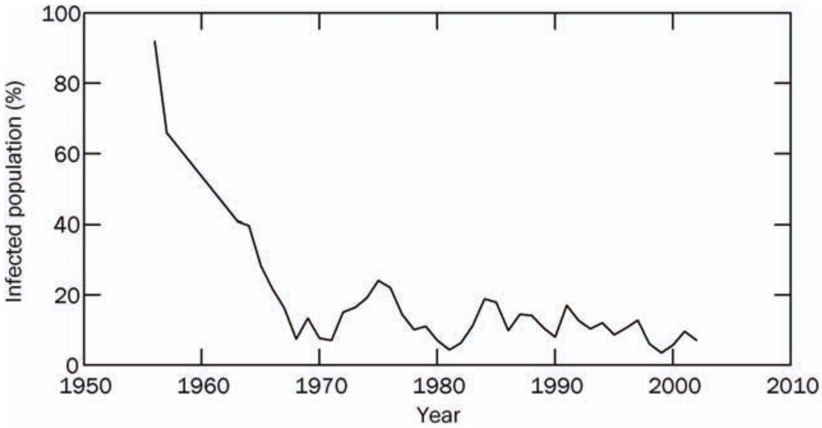

Figure 6-6. Change in percentage of the population infected by malaria parasites in the Nepalese *Terai* between 1955 and 2003.

The control of malaria in the lowlands was followed by rapid immigration from the higher parts of Nepal (Ghimire 1992). The construction of the East–West Highway in the early 1970s and the promise of new land prompted further waves of in-migration. This immigration was extremely pronounced in Chitwan district. As a result overall population growth rate increased from 1.17% before 1941 to 6 and 10% during the 1950's and 1960's respectively (Figure 6-7). Furthermore, malaria control doubtlessly reduced infant mortality rates from 70% before 1957 towards zero in the early 1970's (Jung 2001, Guyatt and Snow 2004). This contributed to increase the autonomous population growth rate.

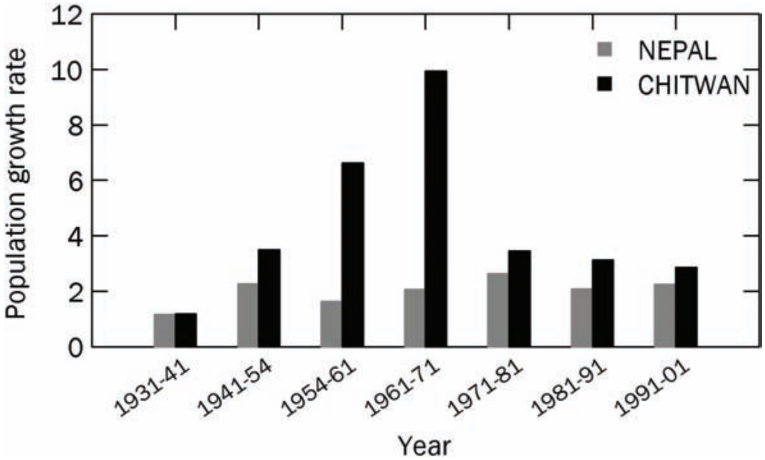

Figure 6-7. Average annual rate of population growth (%) in Chitwan district and Nepal for seven inter-census periods between 1931 and 2001.

The population grew slowly before the malaria control. Growth rates accelerated in the 1950's and 1960's and population increased more than 10 fold within forty years since 1941 (Figure 6-8). We also simulated what would have been the population size without immigration and the growth rate maintained at pre-malaria control level of 1.17% annually. The figure shows that the population of Chitwan district would have been much lower today when the pre 1950's growth rates of 1.17% annually would have been maintained.

The rapid immigration was not only triggered by malaria control alone. By 1960, the Rapti Valley Development Project (1955 - 1964) had distributed 27,759 ha forest land to 5,233 families (Müller-Böker 1991, Guneratne 1994). Isaacson *et al.* (2001) confirmed that both "malaria control and Rapti Valley rural development encouraged the opening of the *Terai*, while supporting increased agricultural production and the formation of industrial towns". As a result, within three decades Chitwan district was transformed from a poorly inhabited jungle into one of the most heavily settled regions of the country.

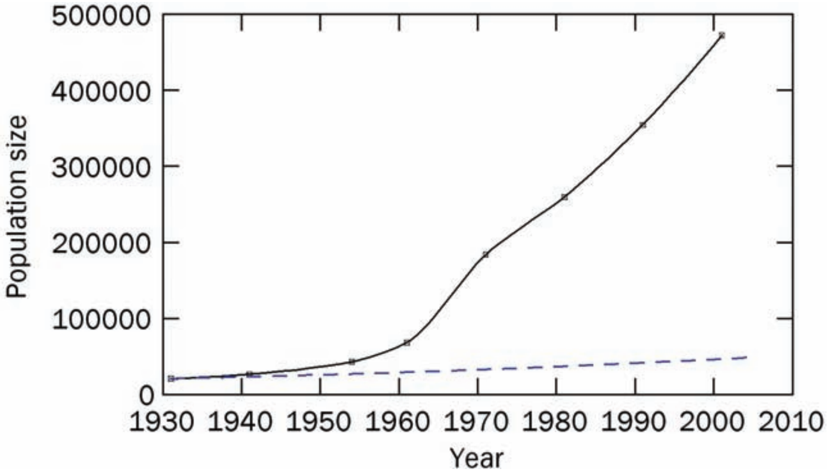

Figure 6-8. Population size of Chitwan district: solid line = actually observed; interrupted = predicted when continued to grow at 1.17% annually.

When did *C. odorata* invade Nepal? The flora of Nepal has been systematically surveyed since 1803 (Rajbhandari 1976). Herbarium records show that species went unnoticed till 1964 (Wallich 1824-1826, Don 1825, Wallich 1829-32, Hooker 1882). John David Adam Stainton recorded the first specimen from Nepal (Stainton 1964) in 1956 (No. 14, BM; Figure 6-9A). Nowadays, *C. odorata* is common in tropical regions of central and east Nepal. Based on the 36 herbarium records (KATH and BM), a chronology of the spread of *C. odorata* in Nepal is presented in Figure 6-9B.

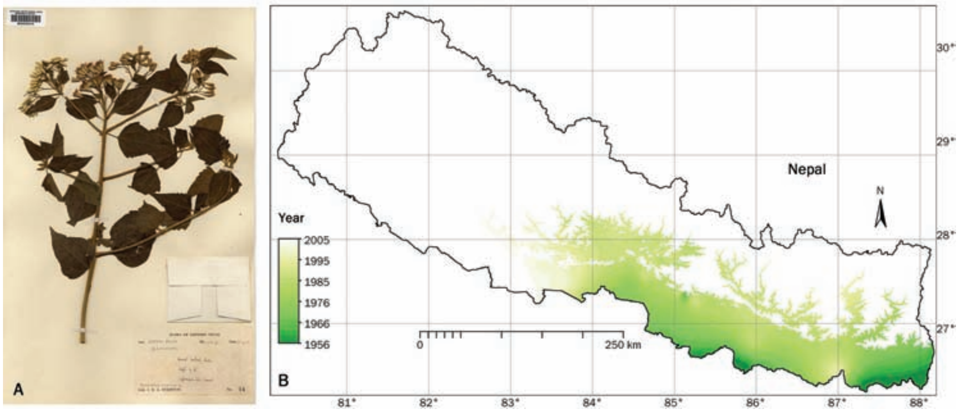

Figure 6-9. A) The earliest collected *C. odorata* specimen from Nepal housed in the Herbarium of the Natural History Museum (BM), London collected by J.D.A. Stainton in 1956, B) Chronology of the invasion of *C. odorata* in Nepal.

Our historical land cover change analysis revealed that forests were opened up somewhere between the 1960's and 1970's. Hence, the period when the canopy of forests opened coincided with the spread of *C. odorata* into the region. We argue that this is not a mere coincidence. Canopy opening created the light conditions in the forest environment required for establishment and reproduction of *C. odorata*. We therefore argue that the degradation of the canopy triggered the invasion success of *C. odorata* into the lowland forests of Nepal. In other words, *C. odorata* did not become a successful invader because of its traits alone, but rather because of the combination of these traits and the degradation of the forest environment. We further demonstrated that the control of malaria increased human population density, which according to Guneratne (1994), Müller-Böker (1991) and Soussan *et al.* (1995) triggered land use changes and forest degradation. We finally argue that this sequence of events leading to forest degradation contributed to the invasion of *C. odorata* in the lowland forests of Nepal. We summarizes the hypothesized cause and effect relationships linking malaria control, population growth and forest degradation to the invasion of *C. odorata* in Figure 6-10.

We are not aware of other cases where a claim has been made that the invasive success of exotic species was facilitated by the control of infectious diseases and the related changes in human demography. Control of infectious diseases has opened up large previously uninhabitable areas of the tropics. Tsetse control in East and Southern Africa for instance, triggered immigration into previously scarcely populated areas resulting in rapid change in land use (Murwira 2003). It would be interesting to further investigate the invasive success of exotics in disease control areas.

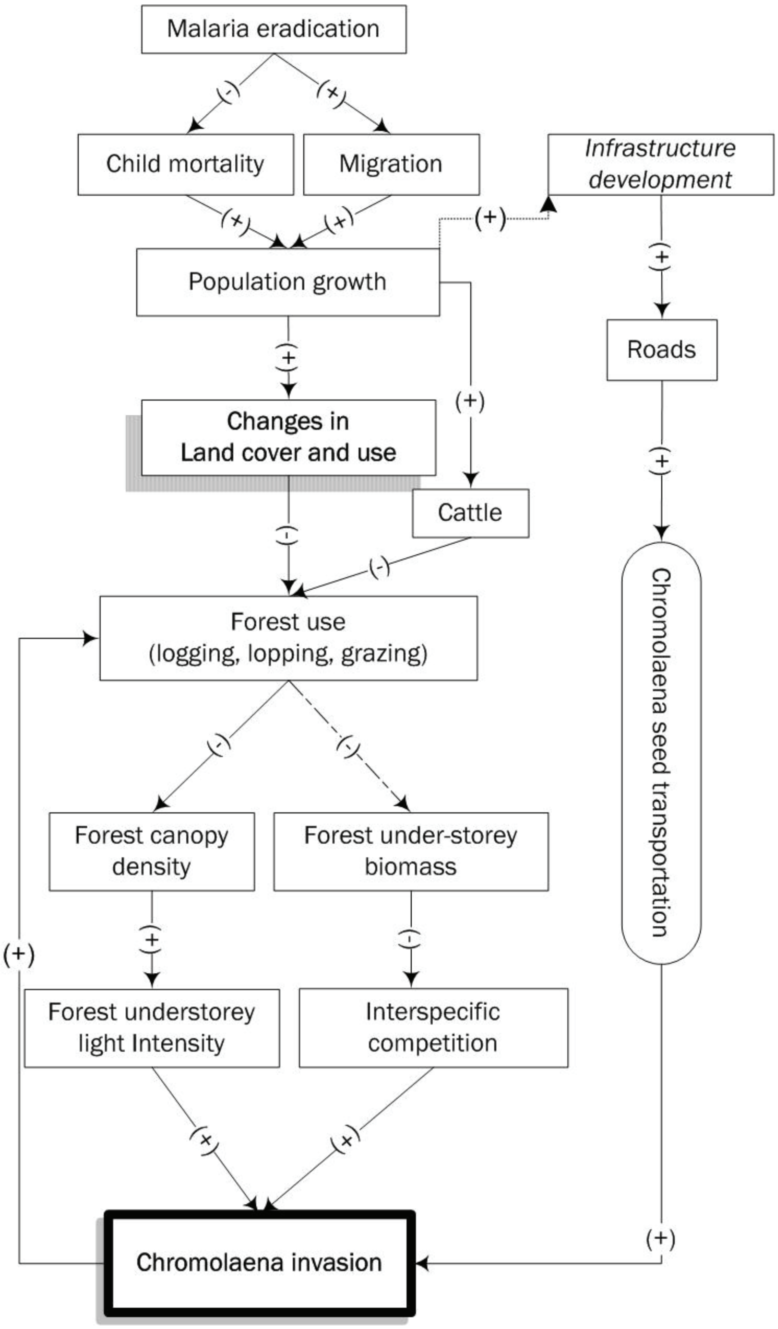

Figure 6-10. Scheme showing hypothesized cause and effect relationships linking malaria control, population growth and forest degradation to the invasion of *C. odorata.*

CHAPTER SEVEN

Modeling the spatial distribution of *Chromolaena odorata* in Nepal

This Chapter is based on:
Joshi, C., De Leeuw, J., and Skidmore, A. K. 2006. upscaling species invasion patterns from local to regional for forest ecosystem management. Proceedings of the ISPRS mid-term Symposium 2006 “Remote sensing: From pixel to processes”. International Archives of the Photogrammetry, Remote Sensing and Spatial Informational Sciences. The Netherlands 8-11 May 2006.

And

Chudamani Joshi, Jan De Leeuw and Skidmore, A. K. 2006. Comparison of bioclimatic and agro-ecological approach in mapping *Chromolaena odorata* in Nepal. Submitted to Diversity and Distribution

7 Modeling the spatial distribution of *Chromolaena odorata* in Nepal

Abstract

Remote sensing and GIS driven bioclimatic variables and their application in predicting presence and absence of invasive species are increasing, however, only few studies have explored the effect of multi-collinearity in model outputs. We developed three models: an expert model, an empirical expert model and a fully empirical model using the first ten components derived from the PCA to predict presence and absence of *C. odorata* in Nepal. Selection of input variables was largely based on our understanding of species auticology from the literature and empirical observation. High resolution interpolated climate surfaces derived from the SRTM and interpolated climate surfaces and a combination of altitude and length of growing season for *C. odorata* allowed us to predict the distribution of *C. odorata* distribution with 80% accuracy. The principal component analysis could predict its distribution with 85% accuracy. This suggests that the use of very few climatic variables could potentially predict the presence and absence of *C. odorata* with high accuracy. Furthermore, very high resolution interpolated surfaces data are more robust in delineating climatic variability within which *C. odorata* could thrive. Coarse scale for species prediction is not appropriate for country similar to Nepal where topographic and climatic variation is high because course scale can not deal with all response shape. The worldwide distribution nature of *C. odorata* populations indicates that bioclimatic variables and their respective resolutions are carefully chosen for model production. Our results indicate that, for modeling the spatial distribution of *C. odorata* having worldwide distribution, one should employ a modeling method whose results are more contributing to ecological explanation than models produced by a 'black box' algorithm.

Key words: Bioclimatic models, Predicting species presence absence, *Chromolaena odorata*, Nepal

7.1 Introduction

Mapping the potential distribution of invasive species has recently gained interest. This interest emerged from the concern that exotic species might, when left uncontrolled, infest areas so far not invaded. Invasion risk maps predict areas at risk of invasion. Typically these are based on a model describing the environmental conditions under which the invasive species has been observed to occur. This model is then used to search for other areas with similar conditions.

Climatic variables have frequently been used to predict potential invasive species distribution (Pysek *et al*. 1998, Higgins *et al*. 1999, Baker *et al*. 2000). Climate based invasion risk assessment has typically been established at small geographical (e.g. (sub) -continental) scale. This predominance of small scale studies is the result of a number of factors. Firstly, studies assessing the potential distribution of invasive species have been called for by national authorities having responsibility over large areas of land. CSIRO for instance developed invasive species risk assessment for the whole of Australia (Sindel 2000, Groves *et al*. 2001, CSIRO 2005). A second reason is that climate impacts on species distribution become in general apparent only over longer distances. A final reason might be that it is easier to obtain maps displaying species distribution and climatic variables at small than at large geographical scale.

Recently, climate databases have been developed with a resolution of 1 km^2. This WORLDCLIM dataset offers the opportunity to model species climate relationship at finer resolution (Hijmans *et al*. 2005). This would be attractive, particularly in countries with complicated topography where climatic conditions change over short distances.

Nepal is such a country. It's complicated topography causes strong climatic variation over short distance. Climate conditions change in relation to elevation, as well as from the humid southern slopes exposed to the monsoon to the arid trans-Himalayan plains in the north. The monsoonal influence also changes from the east to the west of the country. This extreme climatic variation is one of the factors explaining Nepal's high biodiversity. The country hosts floral and faunal elements from a wide variety of bio-geographical provinces in Asia. The wide range of eco-climatological zones also implies that the country could potentially host a large part of the world's flora and fauna. So far however, no attempts have been made to assess the invasion risk at country level.

Invasion risk assessments have been made for the world or individual continents such as Asia. Recently an invasion risk model was established for instance for *Chromolaena odorata* RM King & Robinson in Asia (Kriticos *et al*. 2005). This CLIMEX model predicts the worldwide distribution of *C. odorata* with a 0.5 degree resolution, which is by far too course to account for the topoclimatic variability of Nepal.

It would be tempting to use the 1 km^2 WORLDCLIM database to develop a fine resolution invasion risk models. The WORLDCLIM dataset however, has been developed through interpolation climate stations using elevation as a co-variate. This implies that climate variables might be highly correlated among each other. Multi-collinearity or correlation among predictor variables (Buckland and Elston 1993) inflates the estimation of regression coefficients and may bias models. The WORLDCLIM database also offers tens of variables. Exploring for significance in a large sample of dependent variables increases the danger of type II error e.g. deciding there is a relationship while in reality there is not.

In this paper we investigated the correlation between high resolution climate surfaces for Nepal and developed and compared climate based distribution models for *C. odorata* in Nepal.

7.2 Material and Methods

7.2.1 Nepal and *Chromolaena odorata*

Nepal has a complex topography and climate varies accordingly from tropical to alpine and humid to arid. This high climatic variation makes the country highly attractive not only for tourists but also for species originating from other parts of the world. Many exotic species have found their way into the Nepalese flora and fauna, and an increasing number of these become invasive (HMGN/MFSC 2002).

One of the established invasive species is *C. odorata* commonly known as Banmara or killer of the forest (Figure 7-1). This shrub invaded the tropical forests around the world. Originating from South America it is now considered to be one of the world's 100 worst invaders. *C. odorata* entered the Indian sub-continent around Calcutta prior to 1840 as an ornamental garden plant (Hooker 1882, Prain 1906, McFadyen and Skarratt 1996). From there it spread into lower Burma and Malaysia, and north India (Gautier 1992c, McFadyen and Skarratt 1996). The plant naturalized in the

Ganges flood plain by the 1870s (Clarke 1876). Nowadays, it is common in central and east tropical regions of Nepal (Joshi, pers. obs).

Figure 7-1. Life cycle of *Chromolaena odorata* recorded in the tropical region in the Chitwan region of Nepal 1) A one year old corm and growth of a new shoots in Mar-Apr, 2 and 3) Vegetative growth (Mar-Nov) 4) Flowering (Nov-Feb) 5) Fruiting and seed shedding (Feb-Mar) 6) Death of aerial part (Mar-Apr).

7.2.2 Distribution of *Chromolaena odorata*

We compiled a dataset of 773 sites located throughout Nepal for which we recorded the presence and absence of *C. odorata*. Part of these site descriptions were obtained during botanical surveys undertaken by the first author prior to 2001. Also included were the sites of herbarium records from the National Herbarium in Kathmandu (KATH) and the British Museum (BM). A further sample of 594 site observations was captured during various surveys in 2002, 2003 and 2004. Geographical locations were recorded using Garmen GPS. This sample was used to develop and validate the distribution models for *C. odorata*.

7.2.3 Elevation

An elevation model of the study area was created from 26 scenes of SRTM (Shuttle Radar Topography Mission) images downloaded from Global Land Cover Facility site (GLCF 2004). The 26 scenes were mosaiced and an elevation model of Nepal was derived using ArcGIS 9.0 (ESRI 2004).

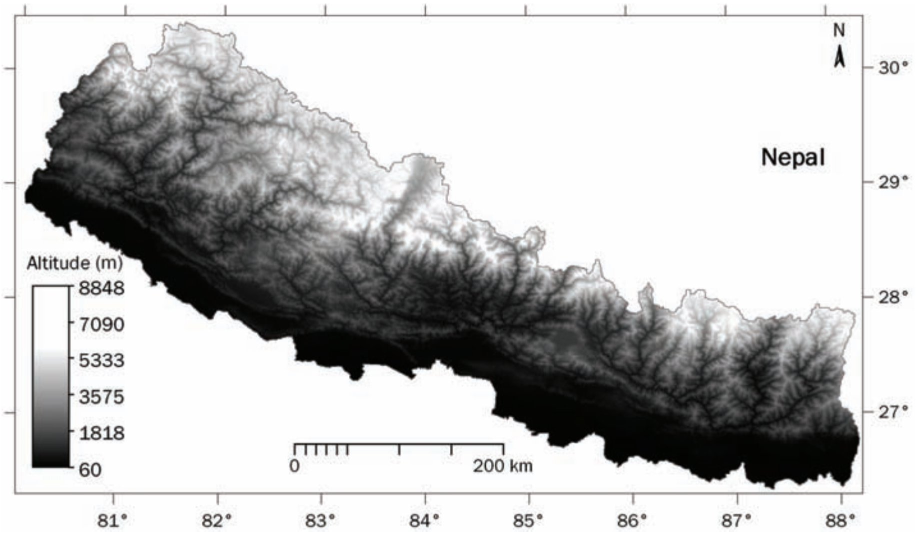

Figure 7-2. A mosaic of 26 scenes of Shuttle Radar Topography Mission images showing elevation and the boundary of study area.

7.2.4 Climate data

We compiled a dataset with ~1 km^2 resolution of fifty years average of monthly precipitation, potential evapotranspiration, minimum, mean and maximum daily temperature. Temperature and precipitation surfaces with 1 km^2 resolution were downloaded from the WORLDCLIM database (www.worldclim.org/), described by (Hijmans *et al*. 2005). The New LOCCLIM software from FAO site (www.fao.org/sd/2002/en1203a_en.htm) was used to generate local climate data of mean monthly potential evpotranspiration and length of growing season for 4550 geographical locations systematically distributed across the country. These point data were converted into point maps. Co-Kriging point interpolation method was applied in a GIS environment using elevation as a co-variable. All raster maps were then resampled to a 1 km^2 grid. The details of the data type, date of acquisition and the source is presented in Table 7-1.

Table 7-1. Description of the data used in this study

SN	Data type	Date	Source*
1	SRTM images (Elevation)	2000	USGS / GLCF
2	Monthly Precipitation	50 years average	(Hijmans *et al*. 2005)
3	Monthly Temperatures	50 years average	(Hijmans *et al*. 2005)
4	Monthly Evapotranspiration	50 years average	New LocCLIM (FAO)
5	Length of growing season	50 years average	New LocCLIM (FAO)
6	Topographic maps	1996	DoS
7	*Chromolaena odorata*	Field observation	2002-2005
8	Herbarium records	1954-2005	KATH, BM

*DoS = Department of Survey, Nepal; KATH = National Herbarium, Kathmandu, Nepal; BM = British Museum, London, England; USGF/GLCF = the Earth Science Data Interface (ESDI) at the Global Land Cover Facility website (http://glcfapp.umiacs.umd.edu:8080/esdi/index.jsp).

7.2.5 Length of growing season (LGS)

The length of growing season (number of days) for each spatial location was computed (Figure 7-3) using Local Climate Estimator (New LocClim software) developed by FAO.

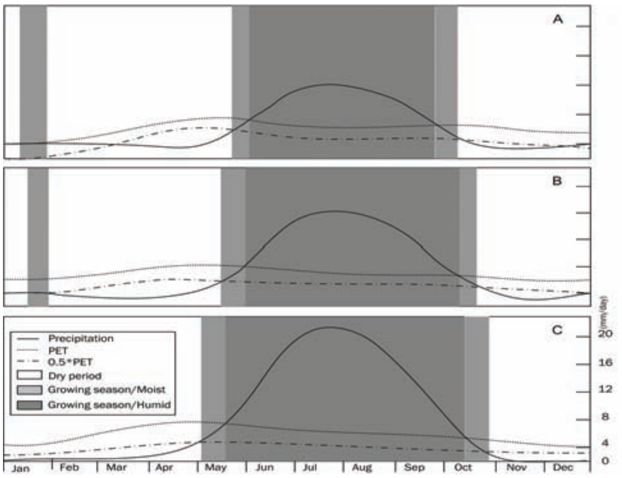

Figure 7-3. Average growing season, precipitation and potential evapotranspiration pattern in the tropical region of A: west Nepal (81 degree Longitude), B) Central Nepal (85 degree Longitude) and C) East Nepal (87 degree Longitude). The numbers on the vertical line denotes degree Centigrade for the Temperature (mean temp) and centimetres for Mean Precipitation (Mean prec), Maximum Precipitation (Max prec), Minimum Precipitation (Min prec) and Evapotranspiration (Evapotrans) for the curves.

7.2.6 Principal component analysis

We calculated 10 principal components (PC) using all environmental variables (monthly mean temperatures (mean, maximum and minimum), monthly mean evapotranspiration, mean monthly precipitation) to reduce data dimensionality. Principal components analysis (PCA) is a multivariate technique that produces a set of components (variables) called principal components which are weighted linear combinations of the original variables (Chatfield and Collins 1980, James and McCulloch 1990).

7.2.7 Model development

We developed three different models: an expert model (Model 1), an empirical expert model (Model 2) and a fully empirical model (Model 3) using the first ten components derived from the PCA.

What factors are considered to limit the distribution of *C. odorata*? The distribution of *C. odorata* has been related to average annual precipitation. Reported precipitation thresholds vary from 1500mm in Nigeria (Ivens 1974), 1200 mm worldwide reported by MacFayden (1989), 1100 mm in Ivory Coast (Gautier 1992a) and 700 mm for Corpus Christi in Texas, reported in Gauthier (Gautier 1992a). The distribution of *C. odorata* has also been related to elevation. It occurs in the Himalayas up to an elevation of 1100 m (Moni & George 1959, cited by (Kriticos *et al*. 2005)). They attributed this to sensitivity to frost. Gauthier (1992a) referred to a third factor when mentioning that *C. odorata* occurs in West Africa in areas with a dry season not exceeding five months.

We combined these variables as inputs in an expert model to predict the distribution of *C. odorata* in Nepal. We considered an area suitable when mean annual precipitation was >1200mm, elevation below 1100m and the length of the growing season at least 7 months (210 days). The length of the growing season was excluded from the expert model, because it remained subjective how to relate the suggestion of Gauthier (1992a) about the importance of the duration of the dry season to the LGS estimates, because duration of the dry season was not defined.

The empirical expert model used annual precipitation, elevation and LGS. These were used as inputs in a logistic regression, which was then used to estimate the coefficients. The fully empirical model was a logistic regression model using the first ten components of the PCA as inputs. The data set had a sample size of 773. A subsample of 387 records (50%) was used to develop the models described above. The remaining 50% of the sample was used to validate the models (see below).

7.2.8 Model evaluation

The accuracy of the predictions obtained with the three models was evaluated using overall accuracy and Kappa. A Kappa based z test has traditionally been used to assess whether the alternative methods differ in accuracy. De Leeuw *et al.* (2006) recently indicated that the Kappa z test is appropriate when comparing independent error matrices. The three confusion matrices used in this study used the same sub set of observations. The data are therefore likely to be dependent. We used a chi square test to test for dependence among the error matrices. We followed De Leeuw *et al.* (2006) who recommended utilizing the more sensitive McNemar test in case of dependent error matrices.

7.3 Results

7.3.1 Correlation between climatic factors

Figure 7-4 shows a matrix revealing the pairwise correlation between the fifty-year averages of monthly potential evapotranspiration, precipitation and the monthly mean of minimum, maximum and average daily temperatures. Also included are altitude, length of the growing season and latitude/longitude in Nepal. The matrix shows all monthly temperature values were extremely highly correlated. Correlation between temperature and precipitation and potential evapotranspiration was lower.

The length of the growing season showed low to intermediate correlation to the above-mentioned climatic variables. Elevation, latitude and longitude generally showed lower correlations to climate variables. This figure shows that multi-collinearity would be a real problem when entering these climate variables as independent variables in a statistical analysis.

The figure also reveals the correlation of the first ten principal components derived from the climate data. PC 8 and PC 10 were highly correlated to thermoclimate. The other components were showing medium correlations to precipitation in various months. The figure reveals that the principal components were not or poorly correlated among each other.

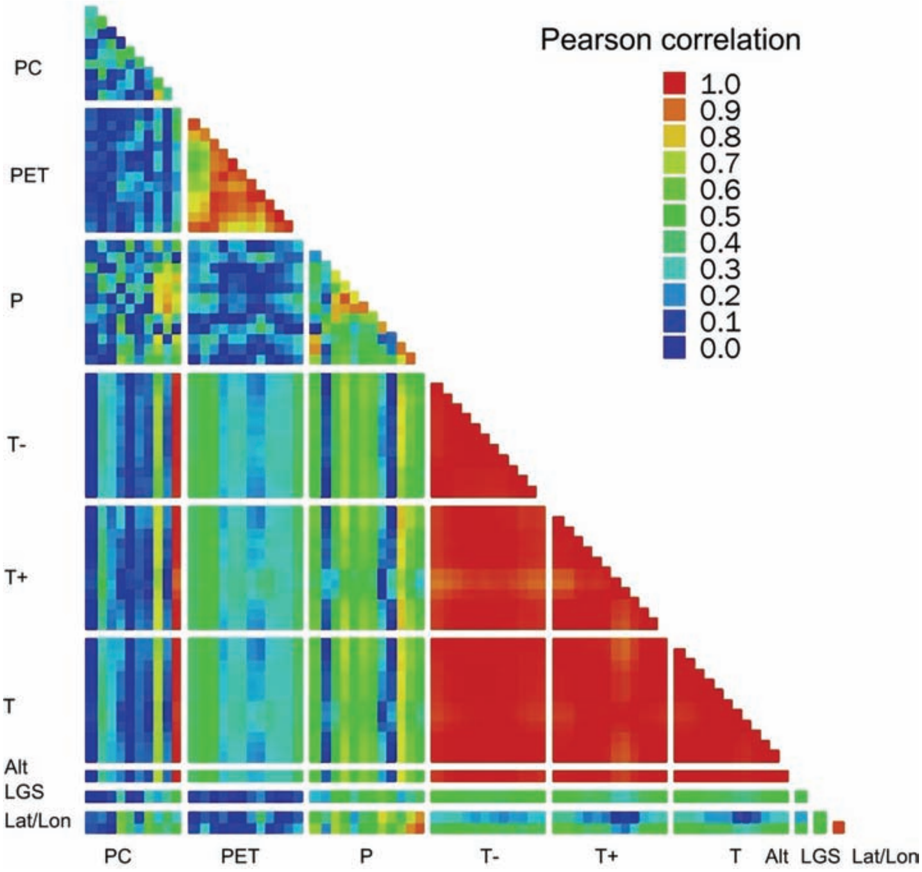

Figure 7-4. Matrix revealing the Pearson correlation between pairs of A) climate variables, length of the growing season, elevation and latitude and longitude. Abbreviations: PET = mean monthly potential evapotranspiration; P = mean monthly precipitation; PC = Principal components (1-10).

Figure 7-5A shows the distribution of *C. odorata* in Nepal. The figure suggests a relation with longitude and elevation, which is confirmed in Figure 7-5B.

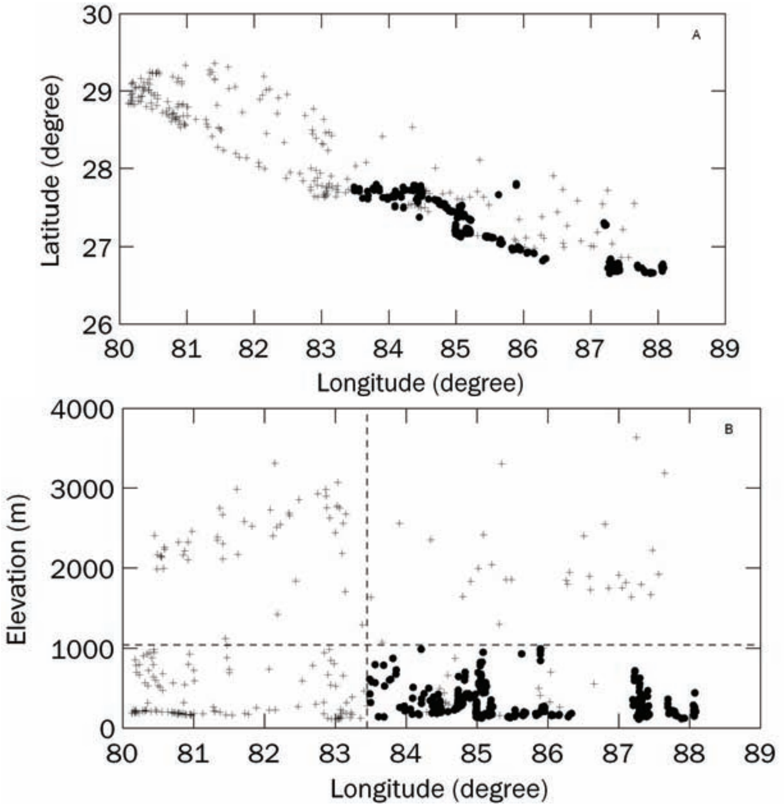

Figure 7-5 A) Distribution of *C. odorata* in Nepal according to 773 records throughout the country. B) Relation of the observed distribution with longitude and elevation.

The expert system (Model 1) predicted the distribution of *C. odorata* using an environmental envelope with elevation <1100 m and annual precipitation >1200 mm/yr. Figure 7-6 shows the distribution of *C. odorata* predicted by this model.

The prediction had a low kappa accuracy of 36%. Figure 7-7 reveals the reason for this poor performance: *C. odorata* distribution was poorly related to annual precipitation. Elevation was a better predictor.

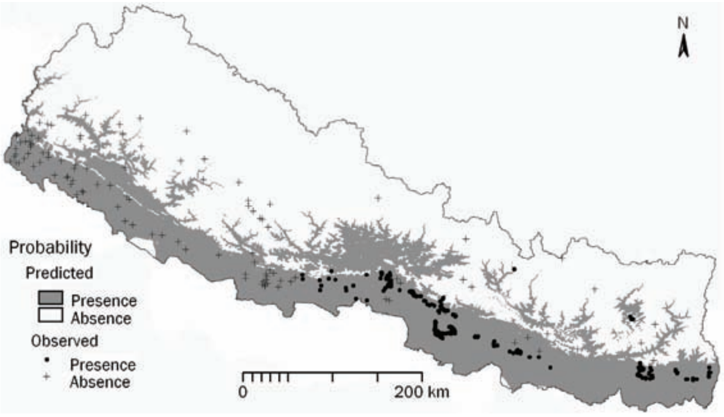

Figure 7-6. Observed (test sample) and empirical model based predicted distribution of *C. odorata* in Nepal at pixel resolution of 100 m.

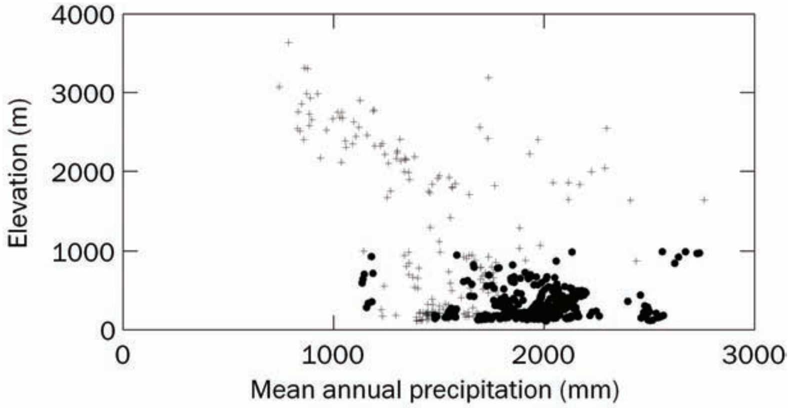

Figure 7-7. Distribution of *C. odorata* presence in a biplot of annual precipitation versus altitude.

We next investigated empirically whether the three variables proposed by experts were related to the distribution of *C. odorata*. Stepwise forward logistic regression confirmed a significant contribution of all three variables (Table 7-2). The model was significantly better than the null model ($-2*(LL_3\text{-}LL0) = 294.42$, df = 3, $p<0.000$) and had a McFadden's $Rho^2 = 0.61$. Figure 7-8 shows the distribution of *C. odorata* predicted by this model.

Table 7-2. Results of logistic regression between *C. odorata* presence absence and three explanatory variables

Variable	Coefficient	SE	t	P
Intercept	-29.71	4.14	-7.16	0.000
LGP	0.1399	0.0214	6.52	0.000
Annual rainfall	0.0054	0.0010	5.28	0.000
Altitude	-0.00512	0.00074	-6.90	0.000

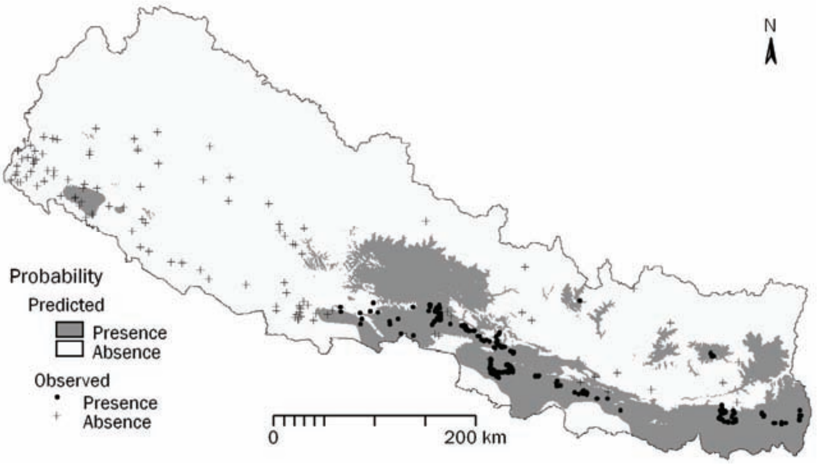

Figure 7-8. Observed (test sample) and empirical expert model based predicted distribution of *C. odorata* in Nepal at pixel resolution of 100 m.

Six components were selected when running a logistic regression model with the first ten principal components as dependent variables (Table 7-3). This model was significantly better than the null model (-2*(LL_6-LL0) = 357.98, df = 6, p<0.000) and had a McFadden's Rho^2 = 0.74. The distribution resulting from this model is displayed in Figure 7-9. The Figure reveals a close match between observed and predicted distribution.

Table 7-3. Results of logistic regression between *C. odorata* presence absence and principal components

Variable	Coefficient	SE	t	P
Intercept	-13.52	7.630	-1.77	0.071
PCA 1	0.023	0.003	6.69	0.000
PCA 5	0.045	0.010	4.48	0.000
PCA 6	-0.113	0.032	-3.45	0.000
PCA 7	-0.117	0.019	-5.90	0.000
PCA 8	-0.264	0.058	-4.53	0.000
PCA 10	0.054	0.025	2.15	0.031

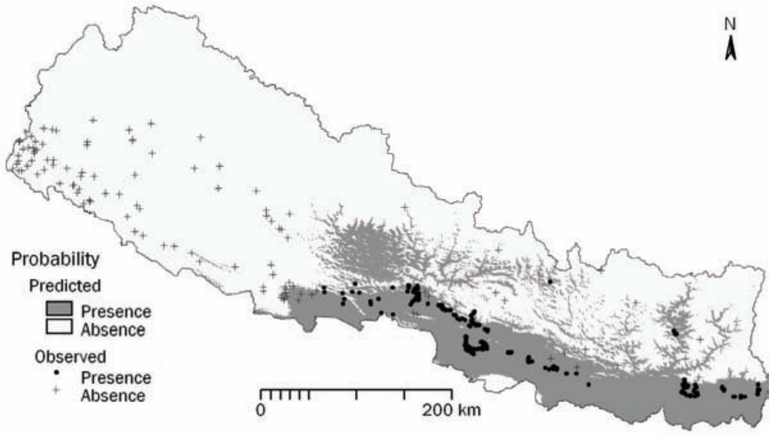

Figure 7-9. Observed (test sample) and principal components based predicted distribution of *C. odorata* in Nepal at pixel resolution of 100 m.

Assessment of the accuracy of the predictions made by the three models revealed that the empirical model based on PCA had the highest accuracy, followed by the empirical expert model (Table 7-4).

Table 7-4. Overall and kappa accuracy and its standard error (SE) for maps classified by three models

Model	Overall Accuracy	Kappa	SE
Expert	78%	0.36	0.0495
Empirical expert	90%	0.74	0.0386
PCA	94%	0.86	0.0289

McNemar's test inferred a significant difference between models 1 and 2, 1 and 3 and 2 and 3.

Table 7-5 shows that model 1 and 2 disagreed for 51 observations. Model 1 classified 3 observations correctly which had been misclassified by model 2, while model 2 had 48 observations correctly classified which had been misclassified by model 1. McNemar's test revealed that there was a significant difference between the two models (χ^2 McNemar symmetry = 39.7; df = 1, $p<0.000$). Similarly model 1 and 3 showed a significant difference in models performance (χ^2 McNemar symmetry = 52.4; df = 1, $p<0.000$). The test also revealed a significant difference between the performance of model 2 and 3 (χ^2 McNemar symmetry = 8.53; df = 1, $p<0.0034$).

Table 7-5. Cross table displaying the frequency of correctly and wrongly classified observed (test sample) points according to the three classification method

		Expert		Empirical expert	
		Wrong	Correct	Wrong	Correct
Empirical expert	Wrong	35	48		
	Correct	3	300		
PCA	Wrong	17	66	15	7
	Correct	5	298	23	341

Figure 7-10 compares the distribution of *C. odorata* as predicted by CLIMEX (Kriticos *et al.* 2005) at coarse resolution of 0.5 degree with the high spatial resolution model based on principle component analysis described above. The figure shows a remarkable difference between the outputs of the two models, both in terms of spatial resolution as well as predicted distribution.

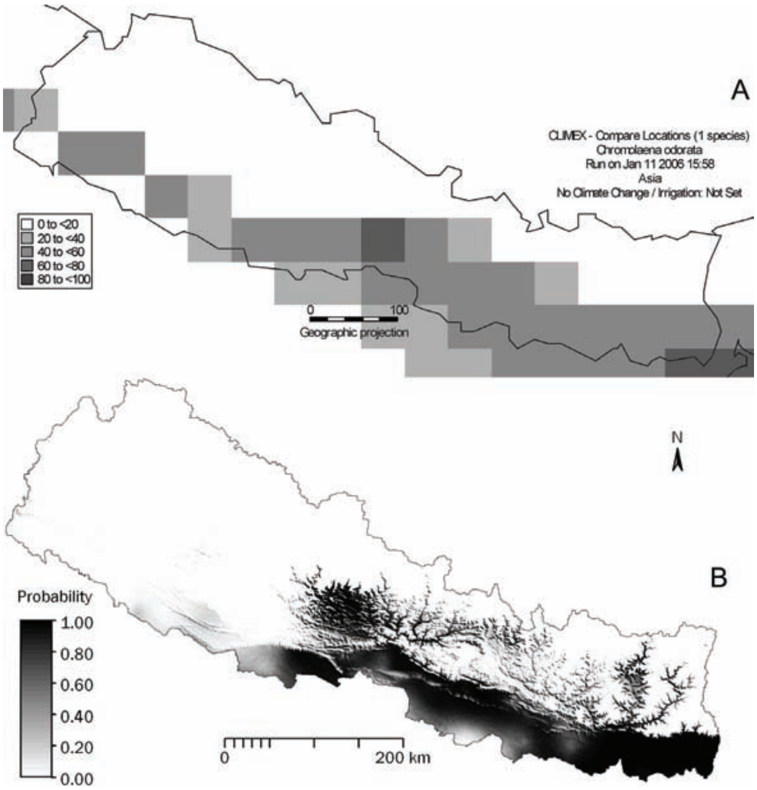

Figure 7-10. Probability of presence and absence of *C. odorata* in Nepal predicted by Climex model (Kriticos *et al.* 2005) at a scale of 0.5 degree (A) and by PCA model at fine scale of 100 m (B).

7.4 Discussion

Our results demonstrated that identifying spatial extent of potential *C. odorata* invasion over large areas can be successfully accomplished using a very few climatic variables. In this study, we also demonstrated that very high resolution interpolated climate surfaces derived from the SRTM and weather stations allowed us to predict the distribution of *C. odorata* with 86% accuracy. Furthermore, higher resolution interpolated climate surfaces have potential to map climate boundaries with higher precision in terrain with complicated topography such as Nepal. As a measure of the proportion of presence or absence that is predicted by a model, McNemar's statistics offer a meaningful numerical variable for intercomparison between models. The evaluation of map accuracy revealed clear differences in performance between different modeling approaches. Models derived from empirical expert knowledge and computer simulated data model performed better than models derived from expert knowledge only.

The expert model is computationally simplistic approach to analyse minimum number of predictive variables, however it failed to produce high accuracy predictive maps of presence and absence of *C. odorata* when compared with empirical expert or PCA based models. The PCA based model yielded highest accuracy predictive map, however their input is a "black-box approach".

We attribute the differences in model performance to a number of issues related to the worldwide distribution of *C. odorata* and other issues related to the input variable to the algorithm. Certainly, image resolution, and predictor variables are two components that must be carefully chosen when constructing any distributional model. Species, for instance *C. odorata* having worldwide distribution may not allows us to draw a climatic envelop. Based on the expert knowledge on the ecology of *C. odorata* we could not draw a fine line of climatic envelop. The problem in mapping of the presence and absence of a species at national or regional scale could be solved by using finer resolution interpolated climate surfaces or remote sensing images which usually provides better predictive ability in models (Guisan and Thuiller 2005). Thus, an increase in spatial resolution of climate data is one of the primary factors necessary to increase model prediction accuracy, particularly for areas with microtopographic variation (Guisan and Zimmermann 2000). The climate data layers used for modeling are available at various resolutions, but even the highest resolution, multi-collinearity among the predictive variables could easily mislead our judgment.

High bioclimatic variation and altitudinal gradient in the Nepalese mountains would suggest that ecological effects on organisms should be strong. This means the potential for effective modeling should be high. However, spatial scale certainly plays a significant role since climate could change within a distance of few hundred meters. The high-altitude distribution limits of a species can be constrained by the low winter temperatures or the amount of frost below which level a plant species could stops its photosynthetic activity or simply could not survive. *C. odorata* could not survive above frost line. Our model assumes that lower lethal temperature could be the strongest influential climatic variable, which limits *C. odorata*'s vertical distribution i.e. northern boundary. This is reasonable because many literatures mentioned that *C. odorata* couldn't survive at this temperature (Kriticos *et al*. 2005). Our empirical observation also showed that *C. odorata* was not observed beyond 1100 m altitude. Our analysis showed that altitude and temperature variables were highly correlated among each other. Hence, using any one of these variable could equally predict the presence and absence of *C. odorata*, which would reinforce our conclusions regarding the reduction of effect of multi-collinearity among climatic variables.

Temperatures or altitude limits did not predicted the horizontal distribution of *C. odorata* because altitude does not change horizontally in Nepal. However, length of growing season for *C. odorata* significantly differs. *C. odorata* needs length of main growing season of at least 7 months time to complete its life cycle. The summer monsoon (summer rainy season), a strong flow of moist air from the southwest, follows the pre-monsoon season. The arrival of the summer monsoon can vary by as much as a month, in Nepal, it generally arrives in early June starting from Eastern Nepal and lasts through September, when it begins to recede. The plains and lower Himalayas receive more than 70% percent of their annual precipitation during the summer monsoon. The amount of summer monsoon rain generally declines from southeast to northwest as the maritime wedge of air gradually becomes thinner and dryer. Although west Nepal receives sufficient pre and post monsoon rain, the main growing season for most of the areas are below 150 days, which is not long enough where *C. odorata* could complete its life cycle. A long dry season between winter and summer monsoon season could potentially prevent *C. odorata* invasion in west Nepal.

Our study revealed that a box classification approach focusing on generalized climatic stress factors only, did not successfully forecast the distribution of *C. odorata* in Nepal. Consideration of length of the growing period significantly improved the predictions. Kriticos *et al.* (2005) predicted that west Nepal was partly suitable. Our first two models do also predict this, with higher spatial resolution. An empirical expert model is better than the expert model, which is attributable to the inclusion of length of growing season. Principal component based model does not predict suitability in west Nepal. The accuracy assessment assesses how well the three models mimic the current distribution. The PCA model does this better than the two other models. It does not assess which model assesses climatic suitability. Climate based potential distribution models assume that we are able to define the climatic boundaries, which restrict the distribution of a species. This is a reasonable assumption in case distribution is indeed determined by climatic factors. This assumption is not necessarily true. It may be false in case of a recent invader which has not yet reached the limits of its envelope. It may also not be true for species which are absent from parts of its climatic envelope for other reasons. In conclusion, computer simulated models using enormous number of climatic variables serve as valuable tools in the invasion ecology, researchers must increase their emphasis on the applicability of predictions generated from these models. The failure to address proper applicability issues will only weaken the potential utility of such models and ultimately will result in models of limited usefulness. All models we presented or discussed may have some importance in scientific understanding, however, there real test of their accuracy will depend on the usefulness and applicability of theses predictive models in invasive species management.

C. odorata is a problem worldwide. It invades natural ecosystems and poses a serious threat to the maintenance and enhancement of biodiversity values of native vegetation. Land managers currently control the spread of invasive non-native plant species after they have already become established instead of trying to prevent the establishment of the species, but its management requires preventative approaches, both at the local and national level. The early detection of new species invasions and the development of rapid response plans are vital for successful eradication programmes. We argue that predictive maps as a result of “box classification” and without considering useful spatial scale, could potentially influence management programme. Because if a species presence absence map is used for management purposes, both failure to correctly predict locations of observed presence and the prediction of false presence will be ‘expensive’ for planning, management and implementation of eradication/control programme. In our view, this is a simplistic approach of detecting actual and potential area under invasion with high precision. This is highly crucial for invasive species management, since limitation of resources forces invasive species managers to carefully plan and prioritize interventions in areas more severely affected by invaders or localize areas requiring interventions most urgently. And they could focus their precious funds and efforts to control the spread of this species and so that the control methods are cost effective and practical. The approach we presented can be well applied for mapping of other species if there biometry and environmental requirement is known.

CHAPTER EIGHT

An integrated assessment of deforestation and forest degradation for biological conservation in the *Terai* of Nepal

This Chapter is partly based on:

Joshi, C., De Leeuw, J., and Skidmore, A. K. 2006. upscaling species invasion patterns from local to regional for forest ecosystem management. Proceedings of the ISPRS mid-term Symposium 2006 “Remote sensing: From pixel to processes”. International Archives of the Photogrammetry, Remote Sensing and Spatial Informational Sciences. The Netherlands 8-11 May 2006.

8 An integrated assessment of deforestation and forest degradation for biological conservation in the *Terai* of Nepal

Abstract

Deforestation and forest degradation are associated processes, which frequently have been studied in isolation. The objective of this study was to assess the spatial extent and severity of both of these processes in the twenty districts in the *Terai* of Nepal using field survey records, existing vegetation maps, and remote sensing data. Forest canopy density was recorded at 329 sites from west to east Nepal using hemispherical photography. An artificial neural network was used to predict forest canopy density in four classes using a mosaic of 16 Landsat ETM+ images captured around the year 2000. Forest canopy density class was predicted with 82% over all accuracy. Data analysis revealed that the forested area reduced from 21774 km^2 in 1958 to 12649 km^2 in 2000 corresponding to an annual rate of decline of 1.38%. Our analysis further revealed that 70% of the forested area outside conservation areas had a canopies with density below 60%, thus confirming widespread degradation. More surprisingly fifty percent of the forested area inside protected areas had similar open canopies. Our analysis revealed that canopy opening prevailed as well in these communities. We argue that from a biodiversity point of view conservation effort should focus on the preservation and restoration of these forests. The forest degradation maps presented in this article could serve as a start to prioritize such interventions.

Key words: Forest degradation, Deforestation, Remote sensing, Terai Nepal, Biological conservation

8.1 Introduction

Deforestation and forest degradation are two different processes, which are frequently linked together. Deforestation is the conversion of forest to another land cover types (Allen and Barnes 1985). It is widely recognized a critical environmental problem with serious long-term economic and social consequences (Siebert 1987, Saunders *et al*. 1991, Ojima *et al*. 1994). More recently, concern has been expressed about forest degradation, a more gradual process, which negatively affects many forests. Both processes occur worldwide but are most prominent in the tropics, where land is cleared to supply resources and free space for a rapidly growing population.

Concerns over the impacts of deforestation and forest degradation creates demand for accurate statistics describing the distribution and rates of these processes (Geist and Lambin 2002). Achard *et al.* (2002) reported a rate of decline of humid tropical forest of 4.9 million ha per year, a statistic 23% lower than the FAO (2001) estimate of 6.4 million ha per year. Percentage of the total forest area annually disappearing was higher in Asia (0.71%) than Africa (0.36%) or Latin America (0.33%).

Forest degradation, is a more subtle process. It may involve opening of the canopy, modification of the vertical structure of the forest or change of other attributes. The forest however remains intact and somehow discernable from the surrounding environment. As such, forest degradation is less straightforward to assess from satellite imagery and rates of forest degradation are harder to obtain. Remote sensing has however been used to assess forest canopy density as an indicator of forest degradation. Intensity of canopy degradation makes the remote sensing measurement sensitive to the state of the underlying ground. There exists a variety of empirical approaches to map forest canopy density. An object based classification (Dorren *et al*. 2003), artificial neural network , fuzzy logic (Pierce *et al*. 1994), linear regression (Iverson *et al*. 1989, Zhu and Evans 1994, DeFries *et al*. 1997, Levesque and King 2003), decision tree classification (Souza Jr. *et al*. 2003), radar backscattering (Saich *et al*. 2001), geometric-optical model (Scarth and Phinn 2000), digital surface models (Gong *et al*. 2002) and a biophysical model based deductive approach (Rikimaru and Miyatake 1997) are some of the methods which were used in mapping forest canopy density. Achard *et al.* (2002) estimated an annual rate of forest degradation in the humid tropics of 2.3 million ha or 0.20% of the total forest area. Rates differed between continents, with Asia (0.42%) having higher rates than Africa (0.21%) and Latin America (0.13%).

Confronted with these trends national and international communities called for interventions to stop both processes. Legislation aiming to conserve forest has been introduced in many countries and increasingly large stretches of forested land are being protected. Management may vary from production systems to forests protected for strict biological conservation. While restoration may be achievable in degraded production forest, this may not be the case for biological conservation areas, once species are extinct.

While deforestation and forest degradation are a point of concern for management of both conservation categories, there would be a need to more closely monitor biological conservation areas, because of the potentially irreversible impacts of deforestation and forest degradation. So far, remotely sensed deforestation and forest degradation investigations have rarely been combined with conservation status, in order to assess the impacts on biodiversity resources.

In this study, we assess the rate of deforestation and the current degree of forest degradation in Nepal. We investigate whether the historic rate of deforestation and current forest degradation status differs between areas protected for biological conservation (e.g. National Parks and wildlife reserves) and those demarcated as production forests.

8.2 Material and methods

8.2.1 Study area

Nepal extends approximately 885 km from east to west and covers an area of 147180 km^{2}. The Himalayas and the trans-Himalayan plains form the northern two-thirds of the country. The study was executed in twenty districts in the *Terai* area to the south (Figure 8-1), which consist of pre-Himalayan footslopes extending into the Indo-Gangetic plain in India.

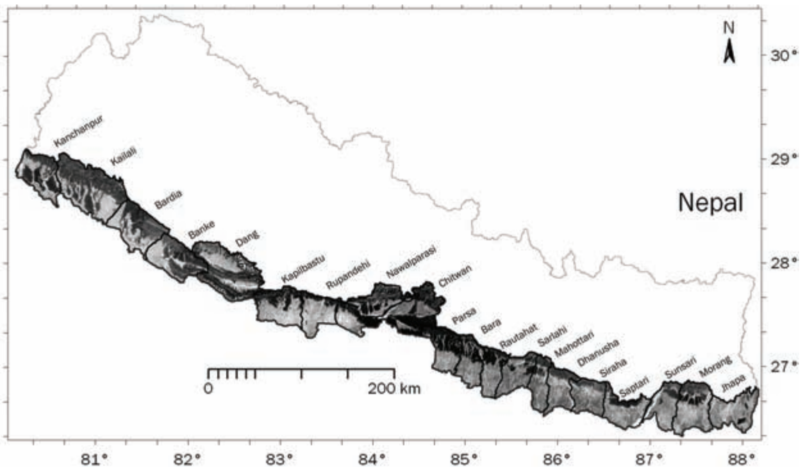

Figure 8-1. Map of Nepal and Landsat ETM+ mosaic image of band 5 showing the 20 *Terai* districts selected for the present study.

The *Terai* has a monsoonal climate and evergreen and semi-deciduous tropical forest as climax vegetation (Stainton 1964, Dobremez 1976). Historically, the *Terai* had little population, because of hyperendemic malaria (Peters *et al*. 1955, Brydon 1961, Darsi Jr and Pradhan 1990). Malaria control initiated in 1956 and a subsequent resettlement policy started in the 1960's drew immigrants from the higher parts of Nepal (Kansakar 1980, Ghimire 1992, Isaacson *et al*. 2001). Rapid population increase resulted in conversion of forest to agriculture (Metz 1991, Soussan *et al*. 1995).

The Nepalese government confronted with this deforestation, legally protected the remaining forests in the 1961. Initially this was motivated from a perspective of resource conservation, while the lowland *Terai* forests deliver commercially valuable timber such as Sal (*Shorea robusta*). Later on biodiversity values further supported the urgency for conservation. The National Parks and Wildlife Conservation Act of 1973 provided the legal basis for the establishment of a number of National Parks aimed at biological conservation. Since then the remaining forest has been placed under two broad management categories – approximately one third of the forest area is conserved as National Parks and wildlife reserves while most of the remaining area is protected as national forest.

Widespread poverty in the *Terai*, with many landless families, puts a high pressure on these remnant forest ecosystems. Many forests have been degraded as a result. We used remote sensing imagery (chapter 3) to assess the degree of forest degradation in Eastern Chitwan district. So far, forest degradation has not been assessed for the whole of the Nepalese *Terai*.

8.2.2 Data

8.2.2.1 Historic forest maps

A digital map displaying the distribution of forest in 1958, derived from 1: 63.360 topographic maps produced by the Survey of India was obtained from the Worldwide Fund for Nature (WWF), Nepal. Dobremez *et al.* (1975), Dobremez (1976), Dobremez and Shakya (1977), Dobremez and Shrestha (1980), Dobremez (1984) and Dobremez *et al.* (1984) produced maps of the actual and potential distribution of forests in Nepal. A digital version of these maps updated by IUCN (Nepal), ICIMOD and HMG, Nepal (unpublished) is referred to in this article as the Dobremez forest community map.

8.2.2.2 Satellite imagery

We selected 16 Landsat ETM+ images captured around the year 2000 (Table 8-1) to study the recent extent and degradation status of the *Terai* forests. Some forest types show considerable phenological variation over the year, due to shedding of leaves in March (Singh and Kushwaha 2006). This may complicate the consistency of image classification between scenes. We therefore selected images from the same season (October to December) shortly after cessation of the monsoon but before leaf fall. The first seven bands of all images were atmospherically corrected, georeferenced, resampled to 30m pixel size and combined into one mosaic using Erdas 8.7.

Table 8-1. Landsat ETM+ Images used in this study, including path and row number, acquisition date and source

SN	Path/row	Acquisition date	Producer
1	139/041	12/26/2000	EarthSat
2	139/042	11/6/1999	USGS / GLCF
3	140/041	10/30/2000	EarthSat
4	140/042	10/28/1999	EarthSat
5	141/040	11/22/2000	EarthSat
6	141/042	10/24/2001	EarthSat
7	142/040	12/13/1999	EarthSat
8	142/041	12/13/1999	EarthSat
9	143/039	10/3/2000	EarthSat
10	143/040	12/25/2001	EarthSat
11	143/041	10/17/1999	EarthSat
12	144/039	10/13/2001	EarthSat
13	144/040	11/9/1999	USGS
14	144/041	11/11/2000	EarthSat
15	145/039	10/15/1999	EarthSat
16	145/040	10/15/1999	USGS

8.2.2.3 Field observations

Forest canopy density was recorded at 329 sites from west to east Nepal in Sept-Oct 2003 and 2004. A digital camera (Nikon coolpix 995) with fisheye lens attached was used to take skyward photographs of the canopy. The camera, positioned at 1.2 m height, produced hemispherical circular images recording gaps in the forest canopy. Canopy density (%) was computed from the digital photographs using Gap Light Analyzer (GLA) software version 2 (Frazer *et al*. 1999). A view angle of 114° was used for canopy density calculation. This corresponded to estimating canopy density for a 900m^2 canopy area for an average tree height of 22.5m. These canopy density estimates were subsequently classified in four classes of <20%, 21-40%, 41-60% and >60% density. From this larger sample of 329 sites one sub-sample of

50% (n=165) was randomly selected to train an artificial neural network to predict canopy density. The remaining sub-set of 164 sites was used to validate the predictions of the neural net.

8.2.2.4 Image classification and data analysis

Artificial neural networks are neurologically inspired statistical mechanisms which has been employed in classification of forest cover using various sensors (Barrow *et al*. 1993, Boyd *et al*. 2002). Here, a three-layer feedforward error-backpropagation artificial neural network implemented in Interactive Data Language (IDL) was used in order to predict forest canopy density in the four classes mentioned above. The algorithm minimizes the root mean square error between the actual output of the multi-layered feed forward perceptron and the desired output (Skidmore *et al*. 1997). The field observations were used to train a neural network. We followed (Atkinson and Tatnall 1997) to search for system parameters to increase the accuracy of the method and avoid overtraining of the neural network. The neural network was trained with canopy density and the reflectance values for the seven ETM+ bands for the sub sample of 165 sites. Another subsample of 164 observations was used in post-classification cross-validation. The best combination of optimum learning rate and momentum to minimize the root mean square error (RSME) was empirically established. The best results were obtained with a learning rate of 0.7, a momentum of 0.7 and four hidden nodes. The RSME stabilized after approximately 3000 epochs. Twenty iterations of 3000 epochs were performed and the best one was selected based on root mean square error.

Finally, a standard majority filter was run through the classified image for smoothing. It works in a 3x3 pixel matrix environment and selects the predominant (most frequently occurring) value or class name of a pixel to the centre pixel in the output map (ILWIS 2005). The validation data set was used to compile a confusion matrix. Accuracy and associated errors of commission and omission and the Kappa statistic was calculated as a measure of actual agreement and chance agreement between training data and classified according to Cohen (1960).

8.3 Results

Figure 8-2 reveals a considerable forest decline in the Nepalese *Terai* between 1958 and 2000. The forested area reduced from 21774 km^2 in 1958 to 12649 km^2 in 2000. This corresponds to a 41.9% decline in 42 years, or an annual rate of decline of 1%.

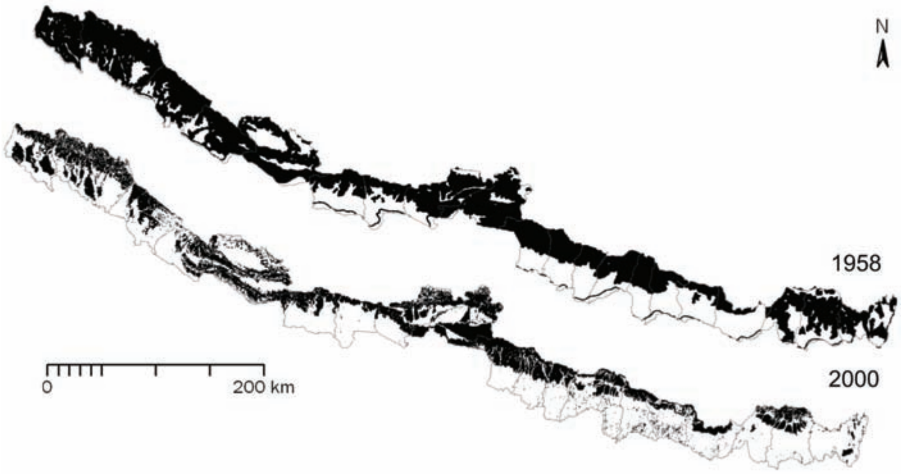

Figure 8-2. Forest distribution (black) in the Nepalese *Terai* in 1958 (derived from 1:63,360 topographic maps) and 2000 (Landsat ETM+ imagery).

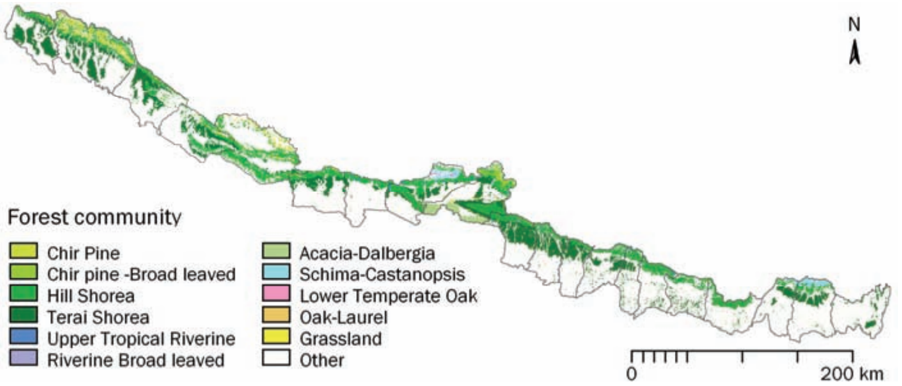

Figure 8-3. Distribution of the forest communities of *Terai* districts of Nepal in 2000, derived while combining figure 2B with the Dobremez forest community map.

Figure 8-3 shows the actual distribution of ten forest communities in 2000. This map was used to produce Table 8-2, which shows the aerial extent of the ten forest communities in 1958 and 2000 and their annual rate of change. The table reveals a decline of forest area for eight out of the ten forest communities.

Table 8-2. Extent (Km^2) in 1958 and 2000 and annual rate of change for ten forest types in the *Terai* districts of Nepal

Forest community	1958	2000	Decline	Decline%	% Yr-1	CV*
Chir Pine	863.9	664.4	-199.5	-23.09	-0.63	2
Chir Pine-Broad Leaved	324.6	321.3	-3.3	-1.01	-0.02	3
Hill *Shorea*	6607.1	5447.7	-1159.4	-17.55	-0.46	1
Terai Shorea	12866.4	5354	-7512.4	-58.39	-2.09	1
Riverine Broad Leaved	68.4	35.3	-33.1	-48.41	-1.57	3
Riverine Khair-Sissoo	681.1	469.9	-211.2	-31.01	-0.88	1
Upper Tropical Riverine	66.6	48.3	-18.3	-27.51	-0.76	2
Schima-Castanopsis	260.4	285.2	24.8	9.54	0.22	4
Lower Temperate Oak	30.9	18.2	-12.7	-41.16	-1.26	3
East Himalayan Oak-Laurel	4.2	5.3	1.1	26.79	0.55	4
Total	21773.6	12649.6	-9124.0	-41.90	-1.29	

*Commercial value: 1= very high, 2 = High, 3 = Moderate, 4 = Low

Terai S. *robusta* forest community was affected most with annual transformation rate of 2%. The Oak-Laurel and *Schima- Castanopsis* forest types increased in size during this period with an average annual rate of 0.55 %. The rate of forest decline was close to significantly related to the commercial value of the forests (Spearman correlation, r = -0.62, n = 11, p = 0.054).

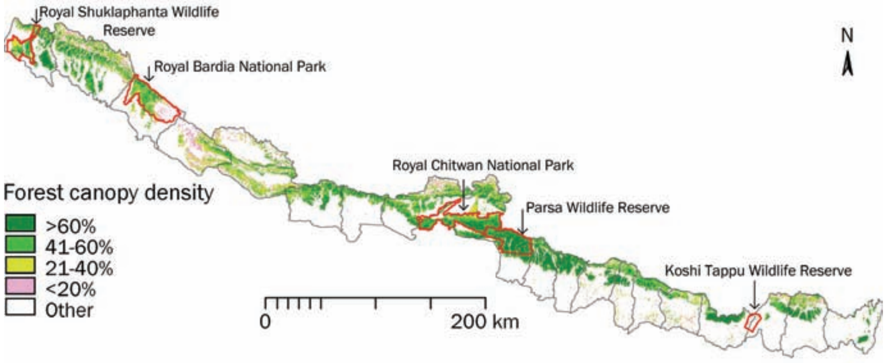

Figure 8-4. Forest canopy density in the *Terai* in 2000. Red lines represent boundaries of National Parks and wildlife reserves.

Figure 8-4 presents the spatial distribution of forest canopy density within and outside the protected areas. Accuracy assessment (Table 8-3) revealed that the canopy classes were mapped with high accuracy. 135 out of 164 observations had been correctly classified. The confusion matrix analysis yielded an overall accuracy of 82.3%.

Table 8-3. Error matrix for observed versus predicted canopy density class by an artificial neural network

Forest canopy density	Other	<20%	21-40%	41-60%	>60%	Total	Ommission error	Producer's accuracy
Other	26	4	2	0	0	32	18.75	81.25
<20%	2	14	2	0	0	18	22.22	77.78
21-40%	1	2	21	7	0	31	32.26	67.74
41-60%	0	1	1	23	6	31	25.81	74.19
>60%	0	0	0	1	51	52	1.92	98.08
Total	29	21	26	31	57	164		
Commission error	10.34	33.33	19.23	25.81	10.53		135	
User's accuracy	89.66	66.67	80.77	74.19	89.47	Overall Accuracy		82.3%

Table 8-3 further shows that the <20% and 21-40% canopy density class were classified with producer's accuracy of over 77% and 67% respectively with much of the error attributed to confusion with the other and 40-60% canopy density class. The high canopy density class (>60%) had the larger producer's accuracy of 98%.

Table 8-4. Frequency distribution (%) of ten forest communities over four canopy closure classes

Forest community	<20%	21-40%	41-60%	>60%	Total
Chir Pine	27.96	35.29	18.31	18.45	100
Chir pine -Broad leaved	21.37	35.86	18.87	23.90	100
Oak-Laurel	0.00	11.20	27.13	61.67	100
Hill *Shorea*	12.65	29.10	26.56	31.68	100
Lower Temperate Oak	29.74	36.12	21.50	12.64	100
Terai Shorea	13.71	22.82	24.73	38.74	100
Riverine Broad leaved	8.63	16.84	26.49	48.03	100
Acacia-Dalbergia	4.44	11.04	36.36	48.15	100
Schima-Castanopsis	29.13	42.20	18.87	9.80	100
Upper Tropical Riverine	6.79	26.30	35.01	31.91	100
Total	14.23	26.58	25.37	33.82	100

Table 8-4 shows forest canopy of all seven communities of *Terai* districts of Nepal was open (except Oak –Laurel and Riverine forest). Visual analysis (Figure 8-4) further indicated that closed canopy forest can exists only inside the protected areas. The canopy closing outside protected forests was significantly lower than the protected ones.

Figure 8-5 revealed that 70 % of tropical forest outside the protected areas had a canopy density below 60%. A larger proportion of the forest inside conservation areas had canopy density above 60%. However, within protected areas almost 50% of the forest area had a canopy density below 60%.

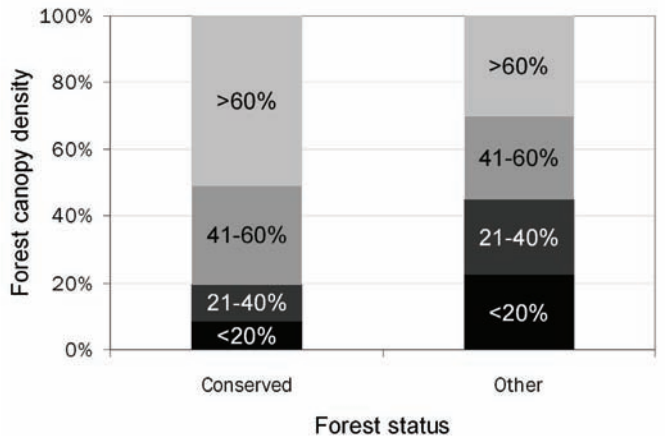

Figure 8-5. Proportion of four forest canopy density classes in and outside National parks and wildlife reserves in the *Terai* districts of Nepal in 2000.

8.3.1 Discussion

This study demonstrated that the forested area in the *Terai* reduced by 41% between 1958 and 2000. Deforestation in the Terai has been reported before by Myers (1986) and Isaacson *et al.* (Isaacson *et al*. 2001). Estimates of deforestation rates and the area of remaining forest differ however between studies, most likely because of different definitions of "forest area" and "deforestation". The impact of definitions is exemplified by the discrepancy between forest cover estimates at a national scale. The FAO and the Land Resources Mapping Project (LRMP) using a 50% and 40% crown cover criterion estimated 29.1 and 28.1% forest cover in Nepal in 1976 and 1985 (Metz 1991). The government of Nepal using a threshold of 10% crown cover estimated that 38.1% of Nepal was forested in 1985/86. The National Planning Commission of Nepal estimated the forest cover of Nepal declined from 60% to 19% between 1961 and 1980, corresponding to an annual rate of decline of 5.7%. The claim of Karan and Iijhna (1985) that one-fourth of the forests had been cut in the preceding decade corresponds to 2.9% annual decline. A slower rate of 1.4% was reported by the World Bank stating that 25% of the forest has been lost in the last 20 years (IBRD 1979). A higher estimate of 4.6% was reported by Allan (1986) who reported that "'over 50% of the forests have been destroyed in 15 years".

The observed deforestation may be attributed to a number of factors. The *Terai* had lower population until the 1950's due to the prevalence of malaria (Peters *et al*. 1955, Brydon 1961, Darsi Jr and Pradhan 1990). Only the Tharu, a tribe resistant to malaria practiced shifting cultivation (Guneratne 1994). Malaria control, started in 1956 in the Rapti valley (Jung 2001) was soon extended to the whole of the *Terai*. The control was followed during the 1960s and 1970s by resettlement programs encouraging immigration to bring additional lands under cultivation. Together these two policies, malaria control and creation of settlement opportunities drew immigrants from the hilly parts of Nepal. The population of the *Terai* increased five fold in forty years time (Isaacson *et al*. 2001). Lands were cleared for agriculture, resulting in the deforestation reported in this study.

The forests of Nepal became national property in 1957. The first national Forest Act came into force in 1961. These were general acts, which did not foresee demarcation of land to be protected or conserved. Realizing the rapid deforestation, the Nepalese government responded by declaring a National Parks and Wildlife Conservation Act, 1973, and first National Park (Royal Chitwan) was established in the same year. The Act established and demarcated protected areas within forest. In 1993, forest rules were amended (Forest Act, 1993 and its Regulations, 1995) to legally protect the remaining forests. Deforestation in the *Terai* is at present effectively halted, as all the remaining forest has been demarcated and protected either as nature conservation areas or as a national forest.

Natural undisturbed forest in the *Terai* typically has canopy closure above 60% (Joshi, personal observation). Our results revealed that 66% of the forests in 2000 had canopies with a closure below 60%. Canopy openings occurring naturally due to storms or dying of trees could bring the canopy density of individual pixels of the satellite image below 60%. However, such canopy gaps of natural origin are a local phenomenon, affecting a small portion of the area at a time. It is unlikely that the 66% of forest area with canopy closure below 60% could be of natural origin. Hence we consider the opening of the canopy of these forests as a sign of degradation. As such our results indicate that forest degradation appears to be a widespread phenomenon in the Nepalese *Terai*.

Malaria control and an active resettlement policy drew immigrants to the *Terai*. As a result, the *Terai* was transformed within three decades from a jungle into one of the most heavily settled regions of the country. Many people are landless, while many those who own land have small holdings only. Confronted with extreme poverty people turned to the forests to complement their livelihood. Although formally forbidden, such utilization by the local poor people was tolerated in most of the national forests. Utilization included collection of secondary forest products, grazing livestock on understorey biomass and logging and lopping of the forest canopy. The forests degraded as a result, a situation continuing until this day. The poverty of the surrounding populations forms the root cause maintaining the *Terai* forests in this degraded state.

Our study reveals that forests outside national parks were more degraded than inside. National parks have far more staff than other forests to ensure effective control. The vulnerability of the national forests led the government to a strategic plan to control the degradation under the Forest Act of 1993. The act is clear: it prohibits deforestation, grazing, and illegal removal of products from government forest land. It did however not stop the degradation, most likely because it did not address the underlying driver, the poverty of the people.

Our results also revealed that 50% of the forest area inside the national parks had an opened canopy. This is alarming. It indicates that protection within parks, notwithstanding better control, falls short. The degradation of the canopy of these forests negatively affects their productivity and potential for wood supply. Primarily however these forests have been protected as National Parks for biological conservation purposes. Degradation of their canopy may have negative implications for their biodiversity value. Johns (1992) and Kellman and Tackaberry (1999) reported that low density extraction of trees may have severe declines of biodiversity. Malcolm and Ray (Malcolm and Ray 2000) suggested that canopy degradation is the most appropriate measurement of anthropogenic disturbance to forests. No specific research has been done so far on this issue in the *Terai* but we (Joshi, pers. obs.) observed that forests with opened canopies have strongly reduced richness of the epiphytic flora. At the level of ecosystem diversity the Nepalese *Terai* is of international importance for three forest ecosystems mentioned in this study: the tropical deciduous riverine forest, tropical monsoon forest and tropical evergreen forest (HMGN/MFSC 2002).

A number of heliophytic invasive species are commonly found under opened canopies, e.g. *Chromolaena odorata*, *Lantana camara* L. and *Mikania micrantha Kunth.* They are unpalatable and rise together with unpalatable native species (e.g. *Clerodendrum viscosum* Vent.) to dominance only in heavily grazed areas. The associated reduction of the forage quality reduces the potential of the understorey to provide benefits to the local communities. Deprived of palatable forage for their cattle, local people started to lop trees to collect leaves to supply to their cattle. This lopping avoids closure of the opened canopy.

The current study thus reveals the ubiquitous presence of forest degradation in the *Terai* of Nepal. At present, with protected areas and national forests being demarcated, deforestation may no longer be the issue of primary concern. Control of forest degradation deserves to be moved to priority number one of the administration. Our study revealed that remote sensing could be applied to efficiently collect information on degradation status and could thus be used as a tool to demarcate areas deserving intervention and to monitor the success of such interventions.

What could be done to reverse this trend? Law enforcement apparently does not work to achieve the policy and objectives to put a halt to degradation. Community based management starts with the assumption that poverty can be alleviated while exploiting natural resources, while at the same time maintaining the ecological integrity of the ecosystem. A panacea to this problem remain exclusive, but balancing strong protection with recognition of human rights and cultural differences in approaches to forest management, must remain the starting point.

CHAPTER NINE

Mapping cryptic invaders and invasibility of tropical forest ecosystems: Synthesis

9 Synthesis

9.1 The challenge of mapping cryptic invaders

Invasive species management initially aimed at eradication. One century of experience has learned however, that except for very few cases this is not achievable. Currently, invasive species management aims at control and mitigation rather than eradication. Limitation of resources forces invasive species managers to carefully plan and prioritize interventions. Ideally, priority should be given to interventions optimizing benefits in terms of damage reduction, durability and cost effectiveness. This could typically more easily be realized in areas most severely affected by invaders. Maps predicting the distribution of invaders and severity of the impact and damage could thus be used to localize areas requiring interventions.

The distribution of invasive species has been mapped from large (local) to small (global) geographical scale. Small-scale invasive species maps either generalize the known geographical range across a continent or the world or indicate areas at risk of invasion. Invasion risk is generally predicted while establishing the climatic envelope within which a species occurs in its realized range. This climatic envelope is then used to delineate areas with similar climatic conditions, which if not yet invaded would be at risk of invasion. Remote sensing has been applied to map the actual distribution of exotic species, generally at relatively large geographical scales.

Applications of remote sensing to invasive species mapping so far focused on detection of species dominating the upper layer of the invaded community. In this thesis, however, we observed that 67% of the world's 100 worst invasive species (ISSG 2004) do not dominate the ecosystem canopy. Most of invaders are cryptic, either because they are small or because they are hidden from detection by remote sensing devices (Pysek and Prach 1995, Gerlach 1996, Chittibabu and Parthasarathy 2000).

We introduced a classification of invasive species based on their remotely sensed canopy reflectance response. Class I includes species dominating the canopy and forming homogeneous single species stands. Class II includes *mixed canopy dominant* species that are members of a multi-species canopy and still directly contribute to the reflected electro-magnetic radiation. Class III includes species not reflecting, but influencing the reflective properties of canopy members, while Class IV finally includes *non-canopy dominant* species that neither "reflect light" nor influence the reflective properties of other canopy dominant species.

Remote sensing for invasive species mapping thus focused on canopy-dominant species, while most of the IUCN's worst invaders appear to be cryptic species, beyond detection for straightforward application of remote sensing. Mapping the distribution of cryptic invaders would thus be a challenge, a challenge we have taken up in this thesis while exploring alternative methods to map invaders at local scale.

What alternative possibilities do we have to map invasive species distribution at local scale? A first important observation is to note that invasive species are not distributed at random. Similar to other species they occur in certain environments while not in others. Local variation in environmental conditions thus relates to and sometimes causes their distribution across landscapes. If this assumption would be true, one could use environmental variables to predict the distribution of species across landscapes.

This idea, attractive as it appears, has thus far little been explored in large-scale invasive species mapping. This is probably because the invasive species community has been struggling with a gene or environment debate, while focusing on the question "What makes exotic species successful invaders?". Some researchers stressed the importance of the intrinsic traits and the inherited ability of species to invade (di Castri 1989, Williamson 1996, Kolar and Lodge 2001). Others maintain that the recipient environment determines whether an exotic species will be established or not (Lonsdale 1999b, Alpert *et al*. 2000, Vila` and Pujadas 2001).

9.2 Species-environment relationships across scale

Above we already noted that different environmental factors determine the distribution of species at different scales. Climatic factors determine species distribution at continental level, while environmental variables varying across landscapes such as the availability of nutrients or other resources may determine species distribution at local level. Figure 9-1 schematically represents the factors determining invasive species distribution at various scales. The global species pool is split up in a number of regional species pools, which evolutionary speaking developed because of their isolation and remained segregated from each other through geographical barriers. Movement of species across such barriers occurred but at low rates, which maintained the segregation of floras and faunas.

Mankind greatly speeded up the movement of species across barriers. Exotic species have been entering and continue to enter regional species pools. Once a bridgehead population establishes the new territory lays open and the exotic may disperse as far as climatic constraints allow. The speed of dispersal will depend on the invaders dispersal mechanisms. Climatic conditions within the local area may act as a sieve. It allows those species to invade the local area which manage to establish and reproduce under these climatic conditions. We used an environmental sieve model (van der Valk 1981, Zobel 1997, Van Andel and Aronson 2005a) to illustrate this; some invaders may find the climatic conditions of a local area suitable, others not.

To invasive species, the local area may still be a highly heterogeneous environment, notwithstanding its relatively homogeneous climate. Environmental factors such as water level, soil aeration, nutrient levels etc. may vary across the landscape. Such environmental conditions make a site within such a landscape suitable for certain species while it remains hostile and uninhabitable to others. Hence, landscape-related environmental variation may act as a second level sieve allowing certain species from the local species pool to enter the species assemblage of a site.

At the level of a site, species compete for resources and the better competitors would come to dominance if undisturbed. Disturbance such as grazing and fire modify competitive relationships, and the actual species composition will be the outcome of both these processes.

Below we review the outcomes of our thesis research in the perspective of the scheme presented in Figure 9-1.

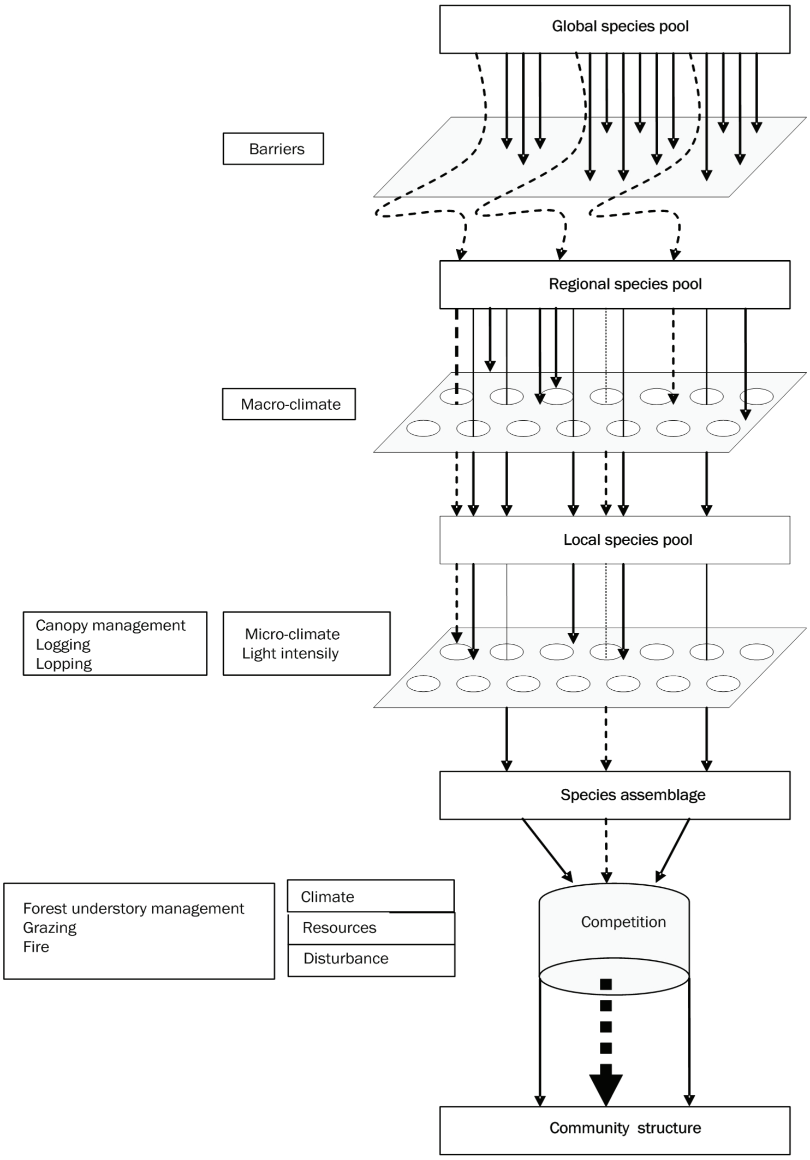

Figure 9-1. Scheme showing factors controlling the distribution of invasive species at various geographical scales.

9.3 The distribution of *Chromolaena odorata*

Our study dealt with *C. odorata*, one of the world's worst invasive species which has invaded tropical forests understorey around the world (Goodall and Erasmus 1996, McWilliam 2000). Our research empirically established two environmental sieves, which constrain the distribution of *C. odorata* in the forests of Nepal. Climatic conditions and heterogeneity in light intensity at landscape level formed two distinct environmental sieves.

A number of authors investigated the climatic constraints to the distribution of *C. odorata*. Its worldwide distribution has been related total annual rainfall (Muniappan 1988, Goodall and Erasmus 1996), frost incidence (Goodall and Erasmus 1996). Kriticos (2005) predicted the worldwide distribution of *C. odorata* using a number of empirically derived climatic variables. The predictions had a spatial resolution of 30 min, which is rather course for a country with a complex topography like Nepal. In chapter 8, we developed 1 km resolution models predicting the climatic boundaries of the distribution of *C. odorata* in Nepal. *C. odorata* disappears at elevations above 1100 m, most likely because of frost sensitivity. The species is also absent from West Nepal. The length of growing season for *C. odorata* in west Nepal is shorter than in the east. The model had an accuracy of 80% and the 1 km climate data thus allowed to model *C. odorata* distribution realistically (Figure 9-2).

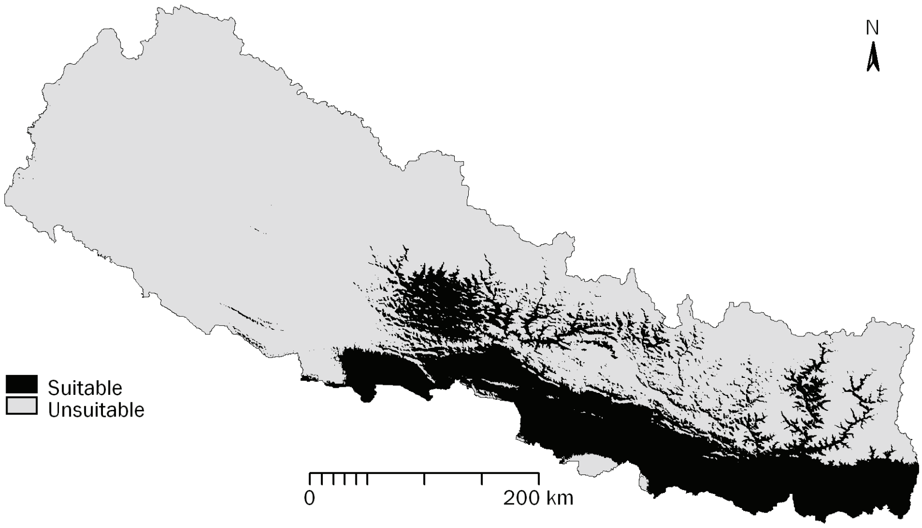

Figure 9-2. Climatic suitability of Nepal for *Chromolaena odorata*.

C. odorata occurs in the lowland forests of Nepal, where it shows a remarkable response to light (Chapter 4). It remains virtually absent from dense forest, but dominates and vigorously produces seed in forest clearings and ruderal environments with more open canopies. This suggests that light intensity determines the competitive and reproductive traits of *C. odorata*. We demonstrated a very strong relationship between light intensity and the sexual and vegetative reproduction of *C. odorata* and concluded that this factor controls the invasion of the species in the forest environment. Light intensity was strongly related to the number of seeds produced per plant (Figure 9-3). *C. odorata* is a clonal grower and we demonstrated that the rate of clonal growth was positively related to light intensity as well.

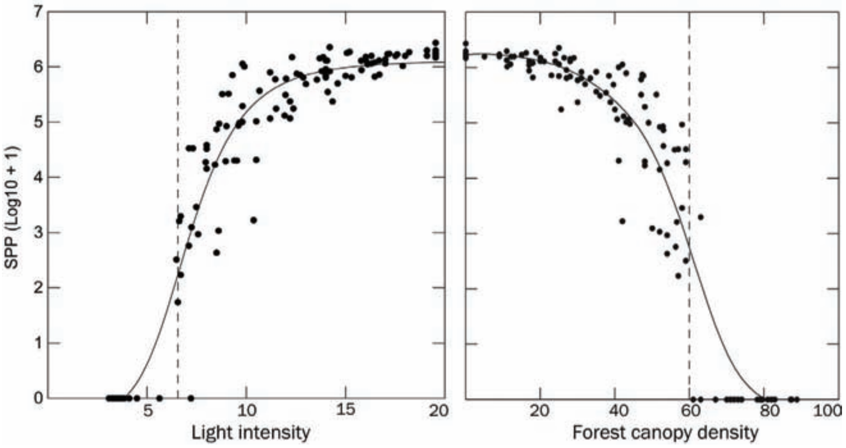

Figure 9-3. Relation between the number of seeds produced per plant (SPP) and light intensity and forest canopy density. The dotted line indicates the light intensity threshold of 6.5 mJ m^{-2} day^{-1} and forest canopy density threshold of 60% separating reproductive and non reproductive populations of *C. odorata*.

Light intensity in the forest is determined by canopy density. Canopies of most forest had been opened up, due to logging and other utilization. We predicted forest canopy density and light intensity reaching the understorey while classifying Landsat imagery using a neural network. These forest canopy density and light intensity maps could then be used to predict the distribution of *C. odorata* populations at local level.

We next mapped the forest in the whole of the Nepalese *Terai* and used Landsat images to assess the degree of canopy degradation (Chapter 8). We further upscale this map up to the climatic suitability limit in Nepal for *C. odorata*.

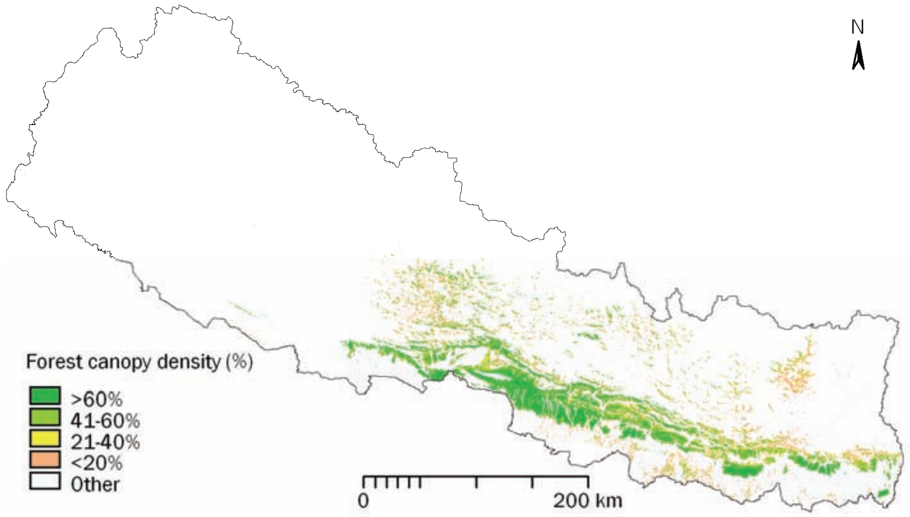

Figure 9-4. Forests canopy density map of Nepal within the suitability range of *C. odorata.*

Derived from this we produced a map showing the suitability of forest to maintain seed producing populations of *C. odorata* (Figure 9-4). This thus corresponds to the second environmental sieve constraining the distribution of *C. odorata*.

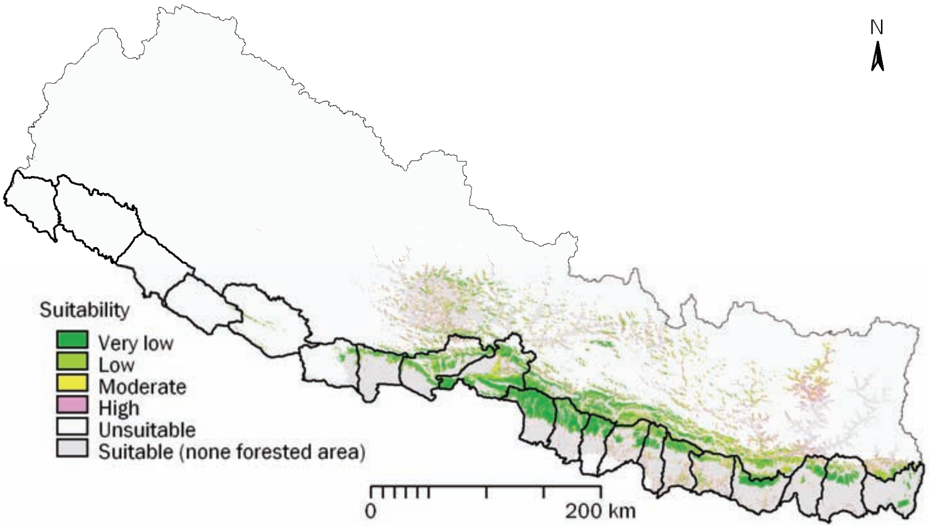

Figure 9-5. Suitability of forests of Nepal for invasion by *C. odorata* considering forest canopy density in *Terai* districts.

Finally, we combined these two environmental sieves into a map showing the potential distribution of *C. odorata* in the forests of Nepal. It predicts the distribution of *C. odorata* while considering climatic constraints and the influence of light intensity due to the degradation of the forests (Figure 9-5).

The map displays the suitability of the forest understorey for *C. odorata*. Its actual presence and abundance depends on the interaction with other species. We noted that *C. odorata* was absent from suitable sites within well-protected areas with little human interference. Here native grass species dominated the forest understorey. People who collect forest products and allow their animals to graze however heavily utilize most of the forests. *C. odorata* is an unpalatable species. Intensive grazing causes the disappearance of the replacement of the original vegetation by unpalatable species like *C. odorata*. Frequent fire used to manage the sylvo-pastoral system further favours *C. odorata*, which is well adapted to survive fire in the dry season.

We argue that opening up of the canopy alone is not enough to explain the widespread invasion of *C. odorata* in the forests in the *Terai*. We hypothesize that overgrazing and management of the forest understorey enabled *C. odorata* to outcompete the native flora and rise to dominance. Or in terms of Figure 9-1, management of the understorey modified the competitive relationships in favour of *C. odorata*. The above assumptions were inferred from field observations. It would be interesting to confirm these supposed causal links through experimentation.

The findings of our study illustrate how remote sensing and GIS technologies can provide ecologists and land managers with an innovative perspective with which to study the factors influencing the patterns of invader population dynamics at local landscape scales. Furthermore, we suggest that forest canopy density, inter-specific competition and anthropogenic factors might account for the differences in size and productivity of *C. odorata* populations. This study shows how these physical factors linked to map the clonal growth rate of an invasive species. Incorporation of remote sensing techniques with species biometry yields instantaneous, useful, cost-effective, multi-scale and temporal information on clonal growth dynamics of an invasive species. Mapping clonal growth rate in *C. odorata* populations could significantly reduce control costs by allowing one to concentrate control efforts to manage habitats so as to curtail the spread of invasive species. In this respect, the immediate benefit of this research has been to contribute to the knowledge base of land managers by providing improved information on the rate of spatial and temporal spread of clonal populations of *C. odorata*, which will support efficient habitat ranking to restore invaded areas and protect non-invaded ecosystems.

9.4 Invasibility of the *Terai* forests

Above we indicated that *C. odorata* invaded the *Terai* forests where the canopy had been broken up. This allows light to penetrate the under story, thus enabling *C. odorata* to reproduce seeds and become invasive.

Analysis of historic maps reveals that the canopy of the forests in *Terai* region of Nepal was more closed (>60% closure) in the 1960s. Today canopy densities below 60% are very common in areas with open access to inhabitants such as the forests in the Barandabhar forest corridor in Chitwan district of Nepal. This suggests that *C. odorata* invaded the forests in the *Terai* because they were degraded.

Our historical land cover change analysis reveals that forests were opened up somewhere between the 1960s and 1970s. Hence, the period when the canopy of forests opened coincided with the spread of *C. odorata* into the region. We suggest that this is not a mere coincidence. Canopy opening created the light conditions in the forest environment required for establishment and reproduction of *C. odorata*. We therefore argue that the degradation of the canopy triggered the invasion success of *C. odorata* into the lowland forests of Nepal. In other words, *C. odorata* did not become a successful invader because of its traits alone, but rather because of the combination of these traits and the degradation of the forest environment. We further demonstrated that the eradication of malaria increased human population density, which - according to Guneratne (1994), Müller-Böker (1991) and Soussan *et al.* (1995) - triggered land use changes and forest degradation. We finally argue that this sequence of events leading to forest degradation contributed to the invasion of *C. odorata* in the lowland forests of Nepal. Hence, human plays a significant role in invasions by exotic species than previously thought (Blumenthal 2005).

9.5 Ecological restoration

Repeated disturbance, typically grazing, logging and lopping will continue to provide suitable light intensity for colonization by *C. odorata* in the forest understorey. Well-established genets of *C. odorata* will eventually be present in the disturbed neighbourhood of forest understorey, because the high light intensity and high levels of continuous disturbance provide long-term habitat. Despite growing awareness and the extensive amount of research carried out on *C. odorata* worldwide, the spread of this species has not been curbed by any chemical, mechanical or cultural control means (Goodall and Erasmus 1996). Even though biological agents - for instance the arctiid moths *Pareuchaetes pseudoinsulata*, *Acalitus adoratus* and *Actinote anteas* - have been introduced for reducing aboveground vegetative biomass and seed production of *C. odorata* with variable success (Kluge 1991, Kluge and Caldwell 1993, McFadyen *et al*. 1996), invasive corms of *C. odorata* pose a continuous threat under light conditions sufficient for growth. Therefore, in order to understand the invasion success of *C. odorata*, the plasticity in corm morphology and the long-term impact of biological agents on resource allocation dynamics must be taken into consideration. We conclude from our study, that both vegetative propagation of corms and aboveground growth and seed production of established *C. odorata* populations decline after recovery or restoration of the disturbed forest until a canopy density of over 60% has become re-established.

This work is one-step further to succeed in efforts to answer the question of how to mitigate the effects of invasive species on tropical forest ecosystems and how to make those ecosystems less prone to invasion. Deliberate introduction of biological agents, as occurs now in biological control efforts will surely be less needed (if they are well effective) if we understand how control of species can be enhanced through the local knowledge and management measures. Similarly, this approach could help in controlling the excessive use of toxic bio-chemicals in suppressing species invasion.

Successful restoration of sustainable communities and *de novo* creation of persistent complex systems that provide essential ecosystem services in novel environments are considered the true tests of our understanding of forest dynamics. That, in short, as Van Andel and Aronson (2005b) explained, is the shared frontier of community ecology and ecological restoration. We need to know which aspects of community structure are restorable once dissembled, and which are not. In our view, after this study, we have that understanding.

9.6 Perspective: science and society

Many of the following recommendations are specific to the tropical region of Nepal. Species and invasion dynamics may differ depending on climatic conditions or the disturbance history. Nonetheless, the conceptual framework for developing *C. odorata* management strategies demonstrated in this study will be appropriate in many other settings. Experience from other tropical countries shows that chemical and mechanical or biological eradication of *C. odorata* seems to be impossible. We propose a feasible strategy in which active control efforts are directed toward infestations that pose the greatest threat to the biological values of the Nepalese tropics. In addition, more passive control methods can be employed to manage less critical but more widespread infestations.

With the support of remotely sensed images, GIS and modeling tools, we developed and supplied the technical information for cryptic invaders to design and implement a comprehensive system that enables the local, regional and national managers to predict, prevent, and rapidly respond to new incursions of invasive species, thus minimizing their spread and impact.

This research gives a new perspective of the capabilities of indirect use of remote sensing tools in mapping and management of invasive species. We believe that this insight will allow the remote sensing community, ecologists and land managers to enhance their skills in understanding invasibility of ecosystems and mapping other cryptic invaders that pose an immediate threat to the recipient ecosystems. This strategy engenders the following recommendations.

- Any disturbances in forest canopy promote invasion of *C. odorata* by increasing the invasibility of such forest communities. Tropical ecosystems of Nepal are highly productive. Effective shading is one of the most effective ways of suppressing the cover abundance of *C. odorata*. Mapping the presence of reproductive populations of this species could significantly reduce the size of area infested by *C. odorata*, which could help in formulating a management strategy and in focusing on areas most requiring interventions. Hundreds of governments and non-government organizations focus on forest management issues independently. These efforts can easily be redirected for the dual purpose, i.e. sustainable management of forests and control of invasive species.
- Control measures will not be necessary for sink populations. Forest with above 60 % canopy cover does not support reproductive populations of *C. odorata* probably because the light intensity is too far below the level of its reproductive growth.
- *C. odorata* is well established abundantly at highly grazed areas throughout the entire tropical region of east and central Nepal. However, our observations suggest that *C. odorata* is incapable of establishing its abundance under dense grassland, hence long-term exclusion of grazing and fire, which kills most of the understorey vegetation, could significantly reduce its abundance.

- *C. odorata* is also abundant along roadsides, forest edges and most of the neglected areas with no agriculture. These populations may not be controlled easily, so targets for intensive control efforts must be prioritized. Fast-growing trees or development of fruit orchards could help in controlling its further abundance and spread.
- The presence of *C. odorata* to the forest interior suggests that the species may invade these areas in the future in case of disturbance. Disturbance associated with livestock grazing may provide *C. odorata* with a bare-soil niche for generative regeneration and for corm invasion and vegetative reproduction. *C. odorata* invasion into such areas can be curtailed by active management by leaving sufficient forest understorey vegetation for more inter-specific competition.

We tried to resolve debates (at least in case of *C. odorata*) about what makes an area susceptible for invasion. As the human population persistently grows, its cumulative impact on the Earth's resources goes up as well. Multiple disturbances have a complex effect on both physical and biotic processes as they interact. Because disturbance alters resource levels, effective management requires understanding of the ecology of disturbance. Tropical forests are experiencing an unprecedented level of impact in both ways from land use change and exotic species invasion. Both natural and anthropogenic disturbance regimes accelerate invasion, which physically influence the ground.

Bio-invasion is a by-product of globalization (Wilson 1995). Prevention becomes a more formidable challenge (Meyerson and Reaser 2002). Eradication of invasive species is an illusion. Hence, the continued persistence of potentially harmful species at tolerable levels could be an ultimate strategy. However, the ability of land managers to combat invasive plants is persistently limited by funding constraints. Information to suggest low-cost control measures and a system to prioritize labour-intensive site-specific eradication could be vital. Through this thesis, we offered an understanding why *C. odorata* is so successful an invader in tropical forest ecosystems of Nepal. This understanding is essential to controlling its further spread. Furthermore, this technique can be equally useful in understanding, detecting and managing other cryptic invasive species.

This thesis highlights the needs and challenges involved in local to regional-scale modeling of plant invasion as a response to land use change. Output from analysis of remotely sensed images and models should be a primary tool for researchers and decision makers to evaluate the question of sustainable resource utilization and safe rates of land use change, and associated impacts on biodiversity, protection of species, and other resource issues and their management.

The ability of the Earth to support more humans is dependent on the ability of humans to manage natural resources wisely. Hence, there is a fundamental need to bring together existing information on invasion processes and vulnerability of ecosystems. Perhaps more urgent at this time is the need to provide information on the management of these ecologically pivotal resources.

References:

Achard, F., H. D. Eva, S. Hans-Jürgen, P. Mayaux, J. Gallego, T. Richards, and M. Jean-Paul. 2002. Determination of deforestation rates of the world's humid tropical forests. Science **297**:999 - 1002.

Akpagana, K., K. A. Guelly, and Y. M. Gumedzoe. 1993. An adventitious plant currently invading the territory of Togo: *Eupatorium odoratum* L. (syn. *Chromolaena odorata* (L.) R.M. King & Robinson), Compositae. Acta Botanica Gallica **140**:535-543.

Allan, N. J. R. 1986. Accessibility and altitudinal zonation models of mountains. Mountain Research and Development **6**:185-194.

Allen, J. C., and D. F. Barnes. 1985. The causes of deforestation in developing countries. Ann. Assoc. Am. Geogr. **75**:163–184.

Allen, T. R., and J. A. Kupfer. 2000. Application of spherical statistics to change vector analysis of Landsat data: Southern appalachian spruce - Fir forests. Remote Sensing of Environment **74**:482-493.

Alpert, P. 1995. Does clonal growth increase plant performance in natural communities? Abstracta Botanica Helvetica **19**:11–16.

Alpert, P., E. Bone, and C. Holzapfel. 2000. Invasiveness, invasibility and the role of environmental stress in the spread of non-native plants. Perspectives in Plant Ecology, Evolution and Systematics **3**:52-66.

Anderson, G. L., J. H. Everitt, A. J. Richardson, and D. E. Escobar. 1993. Using satellite data to map false broomweed (Ericamera austrotexana) infestations on south Texas rangelends. Weed Technology **7**:865-871.

Appleby, M. W. A. 1998. The incidence of exotic species following clear felling of Eucalyptus regnans forest in the Central Highlands, Victoria. Australian Journal of Ecology **23**:457-465.

Atkinson, P. M., and A. R. L. Tatnall. 1997. Introduction neural networks in remote sensing. Int. J. remote sensing **18**:699-709.

Auge, H., and R. Brandl. 1997. Seedling recruitment in the invasive clonal shrub, Mahonia aquifolium. Oecologia **110**:205-211.

Austin, G. E., J. T. Chris, D. C. Houston, and D. B. A. Thompson. 1996. Predicting the distribution of buzzard *Buteo buteo* nesting area using a geographical information system and remote sensing. The Journal of Applied Ecology **33**:1541-1550.

Baker, H. G. 1965. Characteristics and modes of origin of weeds. Pages 147–169 *in* H. G. Baker and G. L. Stebbins, editors. The genetics of colonizing species. Academic Press, New York.

Baker, R. H. A., C. E. Sansford, C. H. Jarvis, R. J. C. Cannon, A. MacLeod, and K. F. A. Walters. 2000. The role of climatic mapping in predicting the potential geographical distribution of non-indigenous pests under current and future climates. Agriculture, Ecosystems & Environment **82**:57-71.

Barrow, P., G. Duff, D. Liddle, and J. Russell Smith. 1993. Threats to monsoon rainforest habitat in northern Australia: the case of Ptychosperma bleeseri Burret (Arecaceae). Australian Journal of Ecology **18**:463-471.

Bayfield, N. G. 1980. Replacement of vegetation on disturbed ground near ski lifts in the Cairngorm mountains, Scotland. Journal of Biogeography **7**:249-260.

Bell, C. 1995. GRAPES project prepares California vineyard managers for insect infestation. Geo Info Systems **5**:44-47.

Benediktsson, J. A., P. H. Swain, and O. K. Esroy. 1993. Conjugate-gradient neural networks in classification of multisource and very-high-dimensional remote sensing data. Int. Journal of Remote Sensing **14**:2883-2903.

Benton, A. R., Jr., and R. M. Newnam. 1976. Color aerial photography for aquatic plant monitoring. Journal of Aquatic Plant Management **14**:14-16.

Berdichevsky, D., I. G. Richardson, B. J. Thompson, D. V. Reames, R. MacDowall, S. P. Plunkett, D. J. Michels, M. L. Kaiser, R. P. Lepping, K. W. Ogilvie, and R. G. Stone. 2000. Examples of fast solar wind transients, their sources and the forecast of possible geomagnetic impact. Geofisica Internacional **39**:5-11.

Binggeli, P. 1996. A taxonomic, biogeographical and ecological overview of invasive woody plants. Journal of Vegetation Science **7**:121-124.

Bischof, H., W. Schneider, and A. J. Pinz. 1992. Multi-spectral classification of Landsat images using neural network. I.E.E.E. Transactions on Geoscience and Remote Sensing 30:482-490.

Blackmore, A. C. 1994. Seed dispersal of *Chromolaena odorata* reconsidered. *in* Proceedings: Third International Workshop on biological control and management of *C. odorata*, Cote d'Ivoire. (access: Dec 12, 2004), available at: http://www.ehs.cdu.edu.au/Chromolaena/fourth/black.htm,

Blumenthal, D. 2005. Interrelated causes of plant invasion. Science 310:243-244.

Botkin, D. B. 2001. The naturalness of biological invasions. Western North American Naturalist 61:261-266.

Boyd, D. S., G. M. Foody, and W. J. Ripple. 2002. Evaluation of approaches for forest cover estimation in the Pacific Northwest, USA, using remote sensing. Applied Geography 22:375-292.

Brabec, J., and P. Pysek. 2000. Establishment and survival of three invasive taxa of the Genus Reynoutria (Polygonaceae) in mesic mown meadows: A field experimental study. Folia Geobotanica 35:27-42.

Bren, L. J. 1992. Tree invasion of an intermittent wetland in relation to changes in the flooding frequency of the River Murray, Australia. Australian Journal of Ecology 17:395-408.

Brydon, H. W., Joshi, G. and Pradhan, S. P. 1961. Anopheles distribution and its relation to the malaria eradication programme in central Nepal. Mosq. News 21:102-105.

Buckland, S. T., and D. A. Elston. 1993. Empirical models for the spatial distribution of wildlife. J. Applied Ecology 30:478–495.

Budd, J. W., T. D. Drummer, T. F. Nalepa, and G. L. Fahnenstiel. 2001. Remote sensing of biotic effects: Zebra mussels (Dreissena polymorpha) influence on water clarity in Saginaw Bay, Lake Huron. Limnology and Oceanography 46:213-223.

Bunting, A. H. 1960. Some reflections on the ecology of weeds. Pages 11-26 *in* J. L. Harper, editor. The Biology of Weeds. Blackwell Scientific Publications, Oxford, UK.

Burke, M. J. W., and J. P. Grime. 1996. An experimental study of plant community invasibility. Ecology 77:776–790.

Bush, J. K., and O. W. Van Auken. 1986. Light requirements of Acacia smallii and Celtis laevigata in relation to secondary succession on floodplains of South Texas. American Midland Naturalist 115:118-122.

Callaghan, T. V., I. S. Jonsdottir, S. Carlsson, B. M. Svensson, and S. Jonasson. 1992. Clonal plants and environmental change: introduction to the proceeding and summaries. Oikos 63:341–347.

Carson, H. W., L. W. Lass, and R. H. Callihan. 1995. Detection of yellow hawkweed with high resolution digital images. Weed Technology 9:477-483.

CBS. 1958. Population census 1952/54. Central Bureau of Statistics, Kathmandu, Nepal.

CBS. 1961. Population census 1961. Central Bureau of Statistics, Kathmandu, Nepal.

CBS. 1971. Population census 1971. Central Bureau of Statistics, Kathmandu, Nepal.

CBS. 1984. Population census 1981. Central Bureau of Statistics, Kathmandu, Nepal.

CBS. 1985. Intercensal changes of some key census variables Nepal 1952/54-81. Central Bureau of Statistics, Kathmandu, Nepal.

CBS. 2001. Statistical year book of Nepal 2001. Central Bureau of Statistics, Kathmandu, Nepal.

Chatfield, C., and A. J. Collins. 1980. Introduction to multivariate analysis. Chapman & Hall, London.

Chazdon, R. L., and C. B. Field. 1987. Photographic estimation of photosynthetically active radiation: evaluation of a computerized technique. Oecologia (Berlin) 73:525–532.

Chittibabu, C. V., and N. Parthasarathy. 2000. Understory plant diversity in a tropical evergreen forest in the Kolli hills, Eastern Ghats, India. Ecotropica 6:129-140.

Civco, D. L. 1993. Artificial neural networks for land cover classification and mapping. International Journal of Geographical Information systems 7:173-186.

Clarke, C. B. 1876. Compositae Indicae.1-30.

Cogan, C. B. 2002. Biodiversity conflict analysis at multiple spatial scales. *in* J. M. Scott, P. J. Heglund, and M. L. M. e. al, editors. Predicting species occurrences, issues of accuracy and scale. Island Press, Washington.

Cohen, J. 1960. A coefficient of agreement for nominal scales. Educational and Psychological Measurement 20:37-46.

Colautti, R. I., and H. J. MacIsaac. 2004. A neutral terminology to define 'invasive' species. Diversity and Distributions 10:135–141.

Congalton, R. G., and K. Green. 1999. Assessing the accuracy of remotely sensed data: Principles and practices. Lewis Publishers, Washington.

Cook, R. E. 1983. Clonal plant populations. American Scientist 71:244-253.

Corsi, F., J. de Leeuw, and A. K. Skidmore. 2000. Modeling species distribution with GIS. Pages 389-434 *in* L. Boitani and T. K. Fuller, editors. Research Techniques in Animal Ecology. Colombia University Press, New York.

Coulter, L., D. Stow, A. Hope, O. L. J, D. Turner, P. Longmire, S. Peterson, and J. Kaiser. 2000. Comparison of high spatial resolution imagery for efficient generation of GIS vegetation layers. Photogrammetric Engineering and Remote Sensing 66:1329-1335.

Cousens, R., and M. Mortimer. 1995. Dynamics of Weed Populations, Cambridge, UK.

Cowling, R. M., R. L. Pressey, M. Rouget, and A. T. Lombard. 2003. A conservation plan for a global biodiversity hotspot--the Cape Floristic Region, South Africa. Biological Conservation 112:191-216.

CSIRO. 2005. CSIRO (Commonwealth Scientific and Industrial Research Organisation) Annual report 2004-05. Australia.

Czech, B., and P. R. Krausman. 1997. Distribution and causation of species endangerment in the United States. Science **277**:116- 117.

D'Antonio, C. M. 1993. Mechanisms controlling invasion of coastal plant communities by the alien succulent *Carpobrotus edulis*. Ecology **74**:83-95.

Daehler, C. C. 2001. Two ways to be an invader, but one is more suitable for ecology. ESA Bulletin **82**:101–102.

Darsi Jr, R. F., and S. P. Pradhan. 1990. The mosquitoes of Nepal: Their identification, distribution and biology. Mosquito Systematics **22**:69-130.

Davis, M. A., J. P. Grime, and K. Thompson. 2000. Fluctuating resources in plant communities: A general theory of invasibility. Journal of Ecology **88**:528-534.

Davis, M. A., and K. Thompson. 2000. Eight ways to be a colonizer; two ways to be an invader: a proposed nomenclature scheme for invasion ecology. ESA Bulletin **81**:226–230.

Davis, M. A., and K. Thompson. 2001. Invasion terminology:should ecologists define their terms differently than others? No, not if we want to be of any help. ESA Bulletin **82**:206.

De Kroon, H., and M. J. Hutchings. 1995. Morphological plasticity in clonal plants: the foraging concept reconsidered. Journal of Ecology **83**:143-152.

de Leeuw, J., H. Jia, L. Yang, X. Liu, K. S. Schmidt, and A. K. Skidmore. 2006. Comparing accuracy assessments to infer superiority of image classification methods. International Journal of Remote Sensing **27**:223-232.

De Rouw, A. 1991. The invasion of *Chromolaena odorata* (L.) King & Robinson (ex *Eupatorium odoratum*), and competition with the native flora, in a rain forest zone, south-west Cote d'Ivoire. Journal of Biogeography **18**:13-23.

DeFries, R., M. Hansen, M. Steininger, R. Dubayah, R. Sohlberg, and J. Townshend. 1997. Subpixel forest cover in central Africa from multisensor, multitemporal data. Remote Sensing of Environment **60**:228-246.

Dennis, J. F., and M. A. Brusven. 1993. Spatial analysis of grasshopper density and ecological disturbance on southern Idaho rangeland. Agriculture, Ecosystems and Environment **43**:31-47.

di Castri, F. 1989. History of biological invasions with special emphasis on the old world. John Wiley, Chichester, UK.

di Castri, F. 1990. On invading species and invaded ecosystems: the interplay of historical chance and biological necessity. Pages 3-16 *in* F. d. Castri, A. J. Hansen, and M. Debussche, editors. Biological invasions in Europe and the Mediterranean Basin. Kluwer Academic Press, Dordrecht, The Netherlands.

Dobremez, J. F. 1976. Le Nepal, Ecologie et Biogeographie + 1 map. CNRS, Paris.

Dobremez, J. F. 1984. Carte ecologique du Nepal. Dhangarhi - Api 1:250.000. *in* Cahiers Nepalais 10. CNRS, Paris.

Dobremez, J. F., D. P. Joshi, P. Bottner, C. Jest, and F. Vigny. 1984. Carte ecologique du Nepal: Butwal - Mustang 1:250.000. *in* Cahiers Nepalais 11. CNRS, Paris.

Dobremez, J. F., A. Maire, and B. Yon. 1975. Carte ecologique du Nepal. Ankhu Khola - Trisuli 1:50.000. Doc. Cartograph. Ecologique **15**:1-20.

Dobremez, J. F., and P. R. Shakya. 1977. Carte ecologique du Nepal. Biratnagar - Kanchenjunga 1:250.000. *in* Cahiers Nepalais 7. CNRS, Paris.

Dobremez, J. F., and T. B. Shrestha. 1980. Carte ecologique du Nepal. Region Jumla - Saipal 1:250.000. *in* Cahiers Nepalais 9. CNRS, Paris.

Don, D. 1825. Prodromus Florae Nepalensis. J. Gale, London.

Dorren, L. K., A. B. Maier, and A. C. Seijmonsbergen. 2003. Improved Landsat-based forest mapping in steep mountainous terrain using object-based classification. Forest Ecology and Management **183**:31-46.

Downey, I. D., C. H. Power, I. Kanellopoulos, and G. G.Wilkinson. 1992. A performance comparison of Landsat Thematic Mapper land cover classification based on neural network techniques and traditional maximum likelihood algorithms and minimum distance algorithms. Pages 518- 528 *in* Proceedings of the Annual Conference of the Remote Sensing Society. Nottingham: Remote Sensing Society.

Drake, J. A., H. A. Mooney, F. di Castri, R. H. Groves, F. J. Kruger, M. Rejmanek, and M. Williamson. 1989. Biological invasions. A global perspective. John Wiley, Chichester.

Dukes, J. S., and H. A. Mooney. 1999. Does global change increase the success of biological invaders? Trends in Ecology and Evolution 14:135-139.

Eav, B. B., R. D. Dillman, and W. B. White. 1984. High-altitude photography for inventories of mountain pine beetle damage. Journal of Forestry 82:175-177.

eCognition. 2003. Definiens Imaging GmbH, Munich, Germany (access: Mar 15, 2003), available at: http://www.definiens-imaging.com.

Elton, C. S. 1958. The ecology of invasions by plants and animals. Meuthuen and Company, London.

ENVI. 2003. The Environment for Visualizing Images (ENVI). *in* Version 4.0. Research Systems, Inc., Boulder, USA.

Epp, G. A. 1987. The seed bank of *Eupatorium odoratum* along a successional gradient in a tropical rain forest in Ghana. Journal of Tropical Ecology 3:139-149.

Erasmus, D. J. 1985. Achene biology and the chemical control of *Chromolaena odorata*. PhD thesis. University of Natal, South Africa.

ESRI. 2004. ArcGIS. *in*. Environmental Systems Research Institute, Redlands, California.

Everitt, J. H., D. E. Escobar, and M. R. Davis. 2001a. Reflectance and image characteristics of selected noxious rangeland species. Remote Sensing GIS Symposium 54:A106-A120.

Everitt, J. H., and P. R. Nixon. 1985. Using color aerial photography to detect camphorweed infestations on South Texas rangelands. Photogrammetric Engineering and Remote Sensing **51**:1793-1798.

Everitt, J. H., C. Yang, B. J. Racher, C. M. Britton, and M. R. Davis. 2001b. Remote sensing of redberry juniper in the Texas rolling plains. Journal of Range Management **54**:254-259.

Ewel, J. J. 1986. Invasibility: lessons from south Florida. Pages 214-230 *in* H. A. Mooney and J. A. Drake, editors. Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York.

FAO. 2001. Global forest resources assessment 2000 main report - FAO Forestry Paper 140. FAO, Rome.

Feyaerts, F., and L. van Gool. 2001. Multi-spectral vision system for weed detection. Pattern Recognition Letters **22**:667-674.

Ford, J. P., and D. J. Casey. 1988. Shuttle radar mapping with diverse incidence angles in the rainforest of Borneo. International Journal of Remote Sensing **9**:927-943.

Fouche, P. S. 1995. The use of low-altitude infrared remote sensing for estimating stress conditions in tree crops. South African Journal of Science **91**:500-502.

Franklin, S. E. 1994. Discrimination of subalpine forest species and canopy density using digital CASI, SPOT PLA, and Landsat TM data. Photogrammetric Engineering and Remote Sensing **60**:1233-1241.

Frazer, G. W., C. D. Canham, and K. P. Lertzman. 1999. Gap Light Analyzer (GLA) Version 2.0. *in* Imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs, users manual and program documentation. Simon Fraser University, Burnaby, British Columbia and the Institute of Ecosystem Studies, Millbrook, New York.

Frazer, G. W., Fournier, R. A., Trofymow J. A., Hall, R. J. 2001. A comparison of digital and film fisheye photography for analysis of forest canopy structure and gap light transmission. Agricultural and Forest Meteorology **109**:249-263.

Frenot, Y., J. C. Gloaguen, L. Massé, and M. Lebouvier. 2001. Human activities, ecosystem disturbance and plant invasions in subantarctic Crozet, Kerguelen and Amsterdam Islands. Biological Conservation **101**:33-50.

Gale, P. 2001. A review: Developments in microbiological risk assessment for drinking water. Journal of Applied Microbiology **91**:191-205.

Gautier, L. 1992a. Contact foret-savane en Cote D'Ivoire centrale: Role de *Chromolaena odorata* dans la dynamique de la vegetation. Ph D. Universite de Geneve, Geneve.

Gautier, L. 1992b. Taxonomy and distribution of a tropical weed: *Chromolaena odorata* (L.) R. King & H. Robinson. Candollea **47**:645-662.

Gautier, L. 1992c. Taxonomy and distribution of a tropical weed: *Chromolaena odorata* (L.) R. King & H. Robinson. Candollea **47**.

Gebhardt, A. 1986. Use of thermography for stand differentiation. Tagungsbericht, Akademie der Landwirtschaftswissenschaften der Deutschen Demokratischen Republik **245**:229-233.

Geist, H. J., and E. F. Lambin. 2002. Proximate causes and underlying driving forces of tropical deforestation. BioScience **52**:143-150.

Gerlach, J. 1996. The effects of habitat domination by invasive Melastomataceae. Phelsuma **4**:19-26.

Ghimire, K. 1992. Forest or farm? The politics of poverty and land hunger in Nepal. Oxford University Press, New Delhi.

GLCF. 2004. Global land cover facility. (access: Oct 19, 2004), available at: http://glcfapp.umiacs.umd.edu:8080/esdi/index.jsp.

Gong, P., M. Xueliang, S. B. Gregory, and Z. Zuxun. 2002. Improvement of an Oak canopy model extracted from digital photogrammetry. Photogrammetric Engineering & Remote Sensing **68**:919-924.

Goodall, J. M., and D. J. Erasmus. 1996. Review of the status and integrated control of the invasive alien weed, *Chromolaena odorata*, in South Africa. Agriculture, Ecosystems & Environment **56**:151-164.

Govereh, J. 1999. Impacts of Tsetse control on migration and capital accumulation: Zambezi Valley, Zimbabwe. Ph.D. Dissertation. PhD Thesis, Michigan State University.

Grime, J. P. 1979. Plant strategies and vegetation processes. John Wiley & Sons, Chichester.

Gross, M. F., V. Klemas, and J. E. Levasseur. 1988. Remote sensing of biomass of salt marsh vegetation in France. International Journal of Remote Sensing **9**:397-408.

Groves, R. 1991. A short history of biological invasions of Australia. Cambridge University Press, Cambridge.

Groves, R. H., and J. J. Burdon. 1986. Ecology of biological invasions. Cambridge University Press, Cambridge.

Groves, R. H., F. D. Panetta, and J. G. Virtue. 2001. Weed Risk Assessment. CSIRO Publishing, Australia.

Guisan, A., and W. Thuiller. 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters **8**:993–1009.

Guisan, A., and N. E. Zimmermann. 2000. Predictive habitat distribution models in ecology. Ecological Modelling **135**:147-186.

Guneratne, A. 1994. The Tharus of Chitwan: ethnicity, class and the state in Nepal. PhD thesis. PhD Thesis, University of Chicago, Chicago.

Guyatt, H. L., and R. W. Snow. 2004. Impact of malaria during pregnancy on low birth weight in Sub-Saharan Africa. Clinical Microbiology Reviews **17**:760-769.

Haltuch, M. A., P. A. Berkman, and D. W. Garton. 2000. Geographic information system (GIS) analysis of ecosystem invasion: Exotic mussels in Lake Erie. Limnology and Oceanography **45**:1778-1787.

Harding, G. B., and G. C. Bate. 1991. The occurrence of invasive Prosopis species in the north-western Cape, South Africa. South African Journal of Science **87**:188-192.

Harper, J. L. 1977. Population Biology of plants. Academic Press, London.

Hengeveld, R. 1989. Dynamics of biological invasions. Chapman and Hall, London.

Hepner, G. F., T. Logan, N. Ritter, and N. Bryant. 1990a. Artificial neural network classification using a minimal training set: comparison to conventional supervised classification. Photogrammetric Engineering and Remote Sensing **56**:469- 473.

Hepner, G. F., T. Logan, N. Ritter, and N. Bryant. 1990b. Artiificial neural network classification using a minimal training set: comparison to conventional supervised classification. Photogrammetric Engineering and Remote Sensing **56**:469- 473.

Hessburg, P. F., B. G. Smith, R. B. Salter, R. D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management **136**:53-83.

Heywood, V. H. 1989. Patterns, extents and modes of Invasions by treeestrial plants. John Wiley & Sons, Chichester.

Higgins, S. I., D. M. Richardson, R. M. Cowling, and T. H. Trinder-Smith. 1999. Predicting the landscape-scale distribution of alien plants and their threat to plant diversity. Conservation Biology **13**:303-313.

Hijmans, R. J., S. E. Cameron, J. L. Parra, P. G. Jones, and A. Jarvis. 2005. Very high resolution interpolated climate surfaces for global land areas. International Journal of Climatology **25**:1965-1978.

Hill, D., P. Coquillard, J. De Vaugelas, and A. Meinesz. 1998. An algorithmic model for invasive species: Application to Caulerpa taxifolia (Vahl) C. Agardh development in the North-Western Mediterranean Sea. Ecological Modelling **109**:251-265.

HMGN/MFSC. 2002. Nepal biodiversity strategy. Ministry of Forests and Soil Conservation, HMG, Nepal.

Hobbs, R. J. 2000a. Land-use changes and invasions. Island Press, Covelo.

Hobbs, R. J. 2000b. Land-use changes and invasions. Island Press, Washington, D.C.

Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology **6**:324-337.

Hong, S. K., I. J. Song, H. O. Kim, and E. K. Lee. 2003. Landscape pattern and its effect on ecosystem functions in Seoul Metropolitan area: Urban ecology on distribution of the naturalized plant species. Journal of Environmental Sciences. **15**:199-204.

Honu, Y. A. K., and Q. L. Dang. 2000. Responses of tree seedlings to the removal of *Chromolaena odorata* Linn. in a degraded forest in Ghana. Forest Ecology & Management **137**:75-82.

Hooker, J. D. 1882. Flora of British India. Spottiswoode, London.

Hu, S. 1999. Integrated multimedia approach to the utilization of an everglades vegetation database. Photogrammetric Engineering and Remote Sensing **65**:193-198.

Hunt, R., and J. H. C. Cornelissen. 1997. Components of relative growth rate and their interrelations in 59 temperate plant species. New Phytologist **135**:395-417.

Huston, M. A. 1994. Biological diversity: the coexistence of species on changing landscapes. Cambridge University Press, Cambridge England.

Hutchings, M. J., and H. De Kroon. 1994. Foraging in plants: the role of morphological plasticity in resource acquisition. Advances in Ecological Research **25**:159-238.

Hyams, D. G. 2005. Curve expert, Hixson, USA. (access: Mar 05, 2005), available at: http://curveexpert.webhop.net/.

IBRD. 1979. Nepal Development Performance and Prospects. The World Bank, Washington, DC.

ILWIS. 2005. Integrated Land and Water Information System. *in*. International Institute for Geoinformation Science and Earth Observation (ITC), Enschede, the Netherlands.

Isaacson, J. M., C. A. Skerry, K. Moran, and K. M. Kalavan. 2001. Half-a-Century of Development: The History of U.S. Assistance to Nepal 1951-2001. the

United States Agency for International Development (USAID) in Kathmandu, Nepal, Kathmandu.

ISSG. 2004. Global Invasive Species Database, Auckland, New Zealand. (access: Jan 27, 2004), available at: http://www.issg.org/database.

Ivens, J. W. 1974. The problem of E. odoratum L. in Nigeria. PANS **20**:76-82.

Iverson, L. R., E.A.Cook, and R.L.Graham. 1989. A technique for extrapolating and validating forest cover across large regions: calibrating AVHRR data with TM data. International Journal of Remote Sensing **10**:1805-1812.

Jakubauskas, M. E., and K. P. Price. 1997. Empirical relationships between structural and spectral factors of Yellowstone lodgepole pine forests. Photogrammetric Engineering and Remote Sensing **63**:1375-1381.

James, F. C., and C. E. McCulloch. 1990. Multivariate analysis in ecology and systematics: panacea or Pandora's box? Annual Review of Ecology and Systematics **21**.

Jarvis, P. J. 1979. The ecology of plant and animal introductions. Progress in Physical Geography **3**:187–214.

Johns, A. D. 1992. Species conservation in managed tropical forests. *in* T. C. Whitmore and J. A. Sayer, editors. Tropical deforestation and species extinction. Chapman & Hall, London.

Johnstone, I. M. 1986. Plant invasion windows: a time based classification of invasion potential. Biological Reviews **61**:369-394.

Jung, R. K. 2001. History of malaria and malaria control in Nepal. Aravali Printers and Publishers, New Delhi, India.

Kanellopoulos, I., A. Varfis, G. G. Wilkinson, and J. Me'gier. 1992. Land-cover discrimination in SPOTHRV imagery using an artificial neural network: a 20-class experiment. International Journal of Remote Sensing **13**:917- 924.

Kansakar, V. S. 1980. Growth and distribution of the population: Population of Nepal. ESCAP, Bangkok.

Karan, P. P., and I. Shigeru. 1985. Environmental stress in the Himalaya. The Geographical Review **75**:71-92.

Kays, S., and J. L. Harper. 1974. The regulation of plant tiller density in a grass sward. Journal of Ecology **62**:97-105.

Kazmi, S. J. H., and E. L. Usery. 2001. Application of remote sensing and GIS for the monitoring of diseases: A unique research agenda for geographers. Remote Sensing Reviews **20**:45-70.

Kellman, M., and R. Tackaberry. 1999. Tropical environments: the functioning and management of tropical ecosystems. Routledge, Londen.

Kerr, J. T., and M. Ostrovsky. 2003. From space to species: ecological applications for remote sensing. Trends in Ecology & Evolution **18**:299-305.

Kharuk, V. I., A. A. Kozhukhovskaya, I. A. Pestunov, K. Ranson, and G. M. Tsibul' skiy. 2001. Use of NOAA/AVHRR imagery in monitoring Siberian moth outbreaks. Mapping Sciences and Remote Sensing **38**:272-281.

King, R. M., and H. Robinson. 1970. *Chromolaena odorata* (L) King and Robinson. Phytologia **20**:204.

Kleinschmidt, I., M. Bagayoko, G. P. Y. Clarke, M. Craig, and D. Le Sueur. 2000. A spatial statistical approach to malaria mapping. International Journal of Epidemiology **29**:355-361.

Kluge, R. L. 1991. Biological control of triffid weed, *Chromolaena odorata* (Asteraceae), in South Africa. Agriculture, Ecosystems and Environment **37**:193-197.

Kluge, R. L., and P. M. Caldwell. 1993. The biology and host specificity of Pareuchaetes aurata aurata (Lepidoptera: Arctiidae), a 'new association' biological control agent for *Chromolaena odorata* (Compositae). Bulletin of Entomological Research **83**:87-94.

Knops, J. M. H. 1995. Introduced and native plants of the Hastings reservation, central coastal California: a comparison. Biological Conservation **71**:115-123.

Kobayashi, M., and T. Kamitani. 2000. Effects of surface disturbance and light level on seedling emergence in a Japanese secondary deciduous forest. Journal of Vegetation Science **11**:93-100.

Kolar, C. S., and D. M. Lodge. 2001. Progress in invasion biology: Predicting invaders. Trends in Ecology and Evolution **16**:199-204.

Kriticos, D. I., T. Yonow, and R. E. Mcfadyen. 2005. The potential distribution of *Chromolaena odorata* (Siam weed) in relation to climate. Weed Research **45**:246–254.

Kriticos, D. J., and R. P. Randall. 2001. A comparison of systems to analyse potential weed distributions. Pages 61-79 *in* R. H. Groves, F. D. Panetta, and J. G. Virtue, editors. Weed Risk Assessment. CSIRO Publishing, Melbourne, Australia.

Kriticos, D. J., R. W. Sutherst, J. R. Brown, S. A. Adkins, and G. F. Maywald. 2003. Climate change and the potential distribution of an invasive alien plant: Acacia nilotica ssp. indica in Australia. Journal of Applied Ecology **40**:111-124.

Krueger, D. W., H. D. Coble, and G. G. Wilkerson. 1998. Software for mapping and analyzing weed distributions: gWeedMap. Agronomy Journal **90**:552-556.

Laliberte, A. S., A. Rango, K. M. Havstad, J. F. Paris, R. F. Beck, R. McNelly, and A. L. Gonzales. 2004. Object-oriented image analysis for mapping shrub encroachment from 1937 to 2003 in southern New Mexico. Remote Sensing of Environment **93**:198-210.

Lass, L. W., B. Shafii, W. J. Price, and D. C. Thill. 2000. Assessing agreement in multispectral images of yellow starthistle (Centaurea solstitialis) with ground truth data using a bayesian methodology. Weed Technology **14**:539-544.

Leslie, A. J., and J. R. Spotila. 2001. Alien plant threatens Nile crocodile (*Crocodylus niloticus*) breeding in Lake St. Lucia, South Africa. Biological Conservation **98**:347-355.

Levesque, J., and D. J. King. 2003. Spatial analysis of radiometric fractions from high-resolution multispectral imagery for modelling individual tree crown and forest canopy structure and health. Remote Sensing of Environment **84**:589–602.

Liggitt, B. 1983. The invasive alien plant C. odorata, with regard to its status and control in Natal. *in* Monograph. Institute of Natural Resources, University of Natal, Pietermaritzburg, South Africa.

Lillesand, T. M., and R. W. Kiefer. 1994. Remote sensing and image interpretation, 3 edition. John Wiley & Sons, New York.

Liu, X., A. K. Skidmore, and H. v. Oosten. 2003. An experimental study on spectral discrimination capability of a backpropagation neural network classifier. International Journal of Remote Sensing **4**:673-688.

Lodge, D. M. 1993. Biological invasions: Lessons for ecology. Trends in Ecology & Evolution **8**:133-137.

Lonsdale, W. M. 1999a. Global patterns of plant invasions and the concept of invasibility. Ecology. **80**:1522-1536.

Lonsdale, W. M. 1999b. Global patterns of plant invasions and the concept of invasibility. Ecology **80**:1522-1536.

Los, S. O., C. J. Tucker, A. Anyamba, M. Cherlet, G. J. Collatz, L. Giglio, F. G. hall, and J. A. Kendall. 2002. Environmental modelling with GIS and remote sensing. Taylor & Francis, London.

Luther, J. E., S. E. Franklin, J. Hudak, and J. P. Meades. 1997. Forecasting the susceptibility and vulnerability of balsam fir stands to insect defoliation with Landsat Thematic Mapper data. Remote Sensing of Environment **59**:77-91.

Macdonald, I. A. W., F. J. Kruger, and A. A. Ferrer. 1986. The ecology and management of biological invasions in Southern Africa. Oxford University Press, Cape Town.

Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout, and F. A. Bazzaz. 2000. Biotic invasions: Causes, epidemiology, global consequences, and control. Ecological Applications **10**:689-710.

Malcolm, J. R., and J. C. Ray. 2000. Influence of timber extraction routes on central African small mammal communities, forest structure, and tree diversity. Conservation Biology **14**:1623-1638.

Mast, J. N., T. T. Veblen, and M. E. Hodgson. 1997. Tree invasion within a pine/grassland ecotone: an approach with historic aerial photography and GIS modeling. Forest Ecology and Management **93**:181-194.

May, C. J., C. M. U. Neale, and N. Henderson. 2000. Mapping riparian resources in semiarid watersheds using airborne multispectral imagery. IAHS AISH Publication **267**:539-541.

McClelland, G. E., R. N. DeWitt, T. H. Hemmer, L. N. Matheson, and G. O. Moe. 1989. Multispectral image-processing with a three-layer back-propagation network. Pages 151-153 *in* Proceedings of the International Joint Conference on Neural Networks. I.E.E.E, New York.

McCormick, C. M. 1999. Mapping exotic vegetation in the Everglades from large-scale aerial photographs. Photogrammetric Engineering and Remote Sensing **65**:179-184.

McFadyen, C. R. E. 1989. Siam weed: a new threat to Australia's north. Plant Protection Quarterly **4**:3-7.

McFadyen, R. C., and B. Skarratt. 1996. Potential distribution of *Chromolaena odorata* (siam weed) in Australia, Africa and Oceania. Agriculture, Ecosystems and Environment **59**:1-2.

McFadyen, R. E., V. C. Moran, and J. H. Hoffmann. 1996. Biocontrol of *Chromolaena odorata*: divided we fail. Proceedings of the 9th international symposium on biological control of weeds, Stellenbosch, South Africa.

McWilliam, A. 2000. A Plague on Your House? Some Impacts of *Chromolaena odorata* on Timorese Livelihoods. Human ecology **28**:451-469.

Medlin, C. R., D. R. Shaw, P. D. Gerard, and F. E. LaMastus. 2000. Using remote sensing to detect weed infestations in Glycine max. Weed Science **48**:393-398.

Meekins, J. F., and R. C. McCarthy. 2000. Responses of the biennial forest herb Alliaria petiolata to variation in population density, nutrient addition and light availability. Journal of Ecology **88**:447-463.

Menges, R. M., P. R. Nixon, and A. J. Richardson. 1985. Light reflectance and remote sensing of weeds in agronomic and horticultural crops. Weed Science **33**:569-581.

Messenger, P. S. 1959. Bioclimatic studies with insects. Annual Rev. Entomol. **4**:183–206.

Metz, J. J. 1991. A reassessment of the causes and severity of Nepal's environmental crisis. World Development **19**:805-820.

Meyerson, L. A., and J. K. Reaser. 2002. A unified definition of biosecurity. Science **295**:44.

Meyerson, L. A., and J. K. Reaser. 2003. Bioinvasions, bioterrorism, and biosecurity. Front Ecol Environ **1**:307–314.

Molofsky, J., S. L. Morrison, and C. J. Goodnight. 1999. Genetic and environmental controls of the establishment of invasive grass, *Phalaris arundinacea*. Biol Invasions **1**:181–188.

Moody, M. E., and R. N. Mack. 1988. Controlling the spread of plant invasions: The importance of nascent foci. J. Applied Ecology **25**:1009-1021.

Mooney, H. A., and J. A. Drake. 1986. Ecology of biological invasions of North America and Hawaii. Springer Verlag, New York.

Mooney, H. A., and R. J. Hobbs. 2000. Invasive species in a changing world. Island Press, Washington, D.C.

Müller-Böker, U. 1991. Knowledge and evaluation of the environment in traditional societies of Nepal. Mountain Research and Development **11**:101-114.

Muniappan, R., M. Marutani. 1988. Ecology and distribution of C. odorata in Asia and the Pacific. Pages 22 *in* Proceedings of the First International Workshop on Biological Control of *Chromolaena odorata*, Bankok, Thailand.

Muriuki, G. W., T. J. Njoka, R. S. Reid, and D. M. Nyariki. 2005. Tsetse control and land-use change in Lambwe valley, south-western Kenya. Agriculture, Ecosystems and Environment **106**:99-107.

Murwira, A. 2003. Scale matters! A new approach to quantify spatial heretogeneity for predicting the distribution of wildlife. PhD thesis. PhD Thesis, Wageningen University, the Netherlands.

Myers, N. 1980. Conversion of tropical moist forests. National Academy of Sciences, Washington, DC.

Myers, N. 1986. Environmental repercussions of deforestation in the Himalayas. Journal of World Forest Resource Management **2**:63-72.

Naesset, E. 1996. Use of the weighted Kappa coefficient in classification error assessment of thematic maps. International Journal of Geographical Information systems **10**:591-604.

NASA. 2005. Landsat 7 science data users handbook. (access: Dec 12, 2005): http://ltpwww.gsfc.nasa.gov/IAS/handbook/handbook_toc.html.

Oetter, D. R., W. B. Cohen, M. Berterretche, T. K. Maiersperger, and R. E. Kennedy. 2000. Land cover mapping in an agricultural setting using multiseasonal Thematic Mapper data. Remote Sensing of Environment **76**:139-155.

Ojima, D. S., K. A. Galvin, and B. L. Turner II. 1994. The global impact of land-use change. BioScience **44**:300–304.

Pearson, R. L., L. D. Miller, and C. J. Tucker. 1976. Hand-held spectral radiometer to measure graminous biomass. Appl. Optics **15**:416-418.

Peddle, D. R., R. Johnson, J. Cihlar, and R. Latifovic. 2004. Large area forest classification and biophysical parameter estimation using the 5-Scale canopy reflectance model in Multiple-Forward-Mode. Remote Sensing of Environment BOREAS Special Issue **89**:139-162.

Peters, A. J., B. C. Reed, M. D. Eve, and K. C. McDaniel. 1992. Remote sensing of brome snakeweed (Gutierrezia sarothrae) with NOAA-10 spectral image processing. Weed Technology **6**:1015-1020.

Peters, W., S. C. Dewar, and T. L. Manandhar. 1955. A preliminary note on the anophelines of the Rapti Valley area of Nepal terai. Indian Journal of Malaria **9**:207-212.

Pierce, L. E., K. Sarabandi, and F. T. Ulaby. 1994. Application of an artificial neural network in canopy scattering inversion. International Journal of Remote Sensing **15**:3263-3270.

Pino, J., Xavier, F., Jaume, C., Marianna, J., Laura, P. 2005. Large-scale correlates of alien plant invasion in Catalonia (NE of Spain). Biological Conservation **122**:339-350.

Polley, H. W., H. B. Johnson, and H. S. Mayeux. 1994. Increasing CO "SUB 2" : comparative responses of the C "SUB 4" grass Schizachyrium and grassland invader Prosopis. Ecology **75**:976-988.

Polley, H. W., H. B. Johnson, and H. S. Mayeux. 1997. Leaf physiology, production, water use, and nitrogen dynamics of the grassland invader Acacia smallii at elevated CO2 concentrations. Tree-Physiology **17**:89-96.

Prain, D. 1906. Contributions to Indian Botany. West Newman & Co., London.

Pressey, R. L., and K. H. Taffs. 2001. Scheduling conservation action in production landscapes: priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss. Biological Conservation **100**:355-376.

Prince, S. D. 1987. Measurement of canopy interception of solar radiation by stands of trees in sparsely wooded savanna (Sudan). International Journal of Remote Sensing **8**:1747-1766.

Pulliam, H. R. 1988. Sources, sinks and population regulation. American Naturalist **132**:652-661.

Pys¡ek, P. 1995. On the terminology used in plant invasion studies. Pages 71–81 *in* P. Pys¡ek, K. Prach, M. Rejmánek, and M. Wade, editors. Plant Invasions: General Aspects and Special Problems. Academic Publishing, Amsterdam.

Pysek, P. 1997. Clonality and plant invasions: can a trait make a difference? Pages 405–427 *in* H. de Kroon and J. van Groenendael, editors. The ecology and evolution of clonal plants, Backhuys, Leiden.

Pysek, P., M. Kopecky, V. Jarosik, and P. Kotkova. 1998. The role of human density and climate in the spread of Heracleum mantegazzianum in the central European landscape. Diversity & distributions 4:9-16.

Pysek, P., and K. Prach. 1995. Invasion dynamics of Impatiens glandulifera - a century of spreading reconstructed. Biological Conservation 74:41-48.

Rajbhandari, K. R. 1976. History of botanical explorations in Nepal. Journal of the Bombay Natural History Society 73:468-481.

Ramsey, I. E. W., G. A. Nelson, S. K. Sapkota, E. B. Seeger, and K. D. Martella. 2002. Mapping Chinese tallow with color-infrared photography. Photogrammetric Engineering and Remote Sensing 68:251-255.

Raunkiaer, C. 1937. Plants life forms. The Clarendon Press, Oxford.

Reichard, S. H., and C. W. Hamilton. 1997. Predicting invasions of woody plants introduced into North America. Conservation Biology 11:193-203.

Reid, R. S., C. J. Wilson, R. L. Kruska, and W. Mulatu. 1997. Impacts of tsetse control and land-use on vegetative structure and tree species composition in south-western Ethiopia. Journal of Applied Ecology 34:731-747.

Rejmánek, M. 2000. Invasive plants: approaches and predictions. Australian Ecology 25:497-506.

Rejmanek, M., and D. M. Richardson. 1996. What attributes make some plant species more invasive? Ecology 77:1651–1661.

Rejmánek, M., D. M. Richardson, M. G. Barbour, M. J. Crawley, G. F. Hrusa, P. B. Moyle, J. M. Randall, D. Simberloff, and M. Williamson. 2002. Biological invasions: politics and the discontinuity of ecological terminology. ESA Bulletin 83:131–133.

Rennolls, K. 2002. Components of error of maps from remotely sensed images. Pages 223-230 *in* G. Hunter and K. Lowell, editors. Accuracy 2002, 5th Internal symposium on spatial accuracy assessment in natural resources and environmental sciences, Melbourne, Australia.

Rescia, A. J., M. F. Schmitz, P. Martin de Agar, C. L. d. Pablo, F. D. Pineda, C. L. De Pablo, and P. M. De Agar. 1997. A fragmented landscape in northern Spain analyzed at different spatial scales: implications for management. Journal of Vegetation Science **8**:343-352.

Reyers, B. 2004. Incorporating anthropogenic threats into evaluations of regional biodiversity and prioritisation of conservation areas in the Limpopo Province, South Africa. Biological Conservation **118**:521-531.

Rich, P. M. 1990. Characterizing plant canopies with hemispherical photographs. Remote Sensing Rev. **5**:13–29.

Richards, J. A. 1999. Remote sensing digital image analysis. Springer-Verlag, Berlin.

Richardson, D. M., and W. J. Bond. 1991. Determinants of plant distribution: evidence from pine invasions. American Naturalist **137**:639-668.

Richardson, D. M., P. Pysek, M. Rejmanek, M. G. Barbour, F. D. Panetta, and C. J. West. 2000. Naturalization and invasion of alien plants: Concepts and definitions. Diversity and Distributions **6**:93-107.

Ridley, H. N. 1930. The dispersal of plants throughout the world. L. Reeve & Co., Ashford, Kent.

Rikimaru, A. 1996. Landsat TM data processing guide for forest canopy density mapping and monitoring model. Pages 1-8 *in* ITTO workshop on utilization of remote sensing in site assessment and planning for rehabilitation of logged-over forest, Bangkok, Thailand.

Rikimaru, A., and S. Miyatake. 1997. Development of forest canopy density mapping and monitoring model using indices of vegetation, bare soil and shadow. (access: Jun 09, 2004), available at: http://www.gisdevlopment.net/aars/acrs/1997/ st5/ts5006pf.htm.

Rikimaru, A., P. S. Roy, and S. Miyatake. 2002. Tropical forest cover density mapping. Tropical Ecology **43**:39-47.

Ringrose, S., and W. Matheson. 1987. Monitoring desertification in Botswana using Landsat MSS data: with consideration as to the nature of the infra-red paradox. Geographical Papers, University of Reading, Department of Geography **96**:13-26.

Ripple, W. J. 1994. Determining coniferous forest cover and forest fragmentation with NOAA-9 Advanced Very High Resolution Radiometer data. Photogrammetric Engineering and Remote Sensing 60:533-540.

Robinson, J. V., and W. D. Valentine. 1979. The concepts of elasticity, invulnerability, and invadability. J. Theoret. Biol 81:91-104.

Roche, C. T., D. C. Thill, and B. Shafii. 1997. Reproductive phenology in yellow starthistle (*Centaurea solstitialis* L.). Weed Science 45:763-770.

Rouget, M., D. M. Richardson, J. L. Nel, M. Le, C. David, B. Egoh, and T. Mgidi. 2004. Mapping the potential ranges of major plant invaders in South Africa, Lesotho and Swaziland using climatic suitability. Diversity & distributions 10:475-484.

Rowlinson, L. C., M. Summerton, and A. Fethi. 1999. Comparison of remote sensing data sources and techniques for identifying and classifying alien invasive vegetation in riparian zones. Water SA 25:497-500.

Saatchi, S. S., J. V. Soares, and D. S. Alves. 1997. Mapping deforestation and land use in Amazon rainforest by using SIR-C imagery. Remote Sensing of Environment 59:191-202.

Sachs, T., and A. Novoplansky. 1997. What does a clonal organization suggest concerning clonal plants? Pages 55-77 *in* d. H. Kroon and J. van Groenendael, editors. The ecology and evolution of clonal plants. Backhuys, Leiden, The Netherlands.

Saich, P., W. G. Rees, and M. Borgeaud. 2001. Detecting pollution damage to forests in the Kola Peninsula using the ERS SAR. Remote Sensing of Environment 75:22-28.

Sajise, P. E., R. K. Palis, N. V. Norcio, and J. S. Lales. 1972. *Chromolaena odorata* imperils grassland. Pasture Newsletter 1:1-2.

Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18–32.

Scarth, P., and S. Phinn. 2000. Determining forest structural attributes using an inverted geometric-optical model in mixed eucalypt forests, Southeast Queensland, Australia. Remote Sensing of Environment 71:141-157.

Schullery, P. 2001. What is natural? philosophical analysis and yellow stone practice. Western North American Naturalist 61:255-256.

Shepherd, J. D., and J. R. Dymond. 2000. BRDF correction of vegetation in AVHRR imagery. Remote Sensing of Environment 74:397-408.

Shumway, S. W. 1995. Physiological integration among clonal ramets during invasion of disturbance patches in a New England salt marsh. Annals of Botany 76:225–233.

Siebert, S. F. 1987. Land use intensification in tropical uplands: effects on vegetation, soil fertility and erosion. Forest Ecology and Management 21:37–56.

Silander, J. A. J., and D. M. Klepeis. 1999. The invasion ecology of Japanese barberry (Berberis thunbergii) in the New England landscape. Biological Invasions 1:189-201.

Sindel, B. M. 2000. Australian Weed Management Systems. CSIRO publishing, Australia.

Singh, K. P., and C. P. Kushwaha. 2006. Diversity of flowering and fruiting phenology of trees in a tropical deciduous forest in India. Annals of Botany 97:265–276.

Skidmore, A. K., G. W. Forbes, and D. J. Carpenter. 1988. Non-parametric test of overlap in multispectral classification. International Journal of Remote Sensing 9:777-785.

Skidmore, A. K., B. J. Turner, W. Brinkhof, and E. Knowle. 1997. Performance of a neural network: mapping forests using remotely sensed data. Photogrammetric Engineering & Remote sensing 63:501-514.

Smallwood, K. S. 1994. Site invasibility by exotic birds and mammals. Biological Conservation 69:251-259.

Smirnov, S. I., and M. S. Kotova. 1994. System of ecological monitoring of the forest resources on territories contaminated by radionuclides. Lesnoe Khozyaistvo 3:40-42.

Sokal, R. R., and F. J. Rohlf. 1995. Biometry: The Principles & Practice of Statistics in Biological Research. Freeman, New York.

Soussan, J., B. K. Shresta, and L. P. Uprety. 1995. The social dynamics of deforestation: A case study from Nepal. Parthenon Publishing Group, New York.

Souza Jr., C., L. Firestone, L. M. Silva, and D. Roberts. 2003. Mapping forest degradation in the Eastern Amazon from SPOT 4 through spectral mixture models. Remote Sensing of Environment **87**:494-506.

Specter, C., and D. Gayle. 1990. Managing Technology Transfer for Coastal Zone Development: Caribbean Experts Identify Major Issues. Int. J. of Remote Sensing **11**:1729-1740.

Stainton, J. D. A. 1964. Notes on a Journey in East and Central Nepal 1964 with brief notes on journeys in East Nepal 1956 and in Central Nepal. Pages 51 pages *in*.

Stohlgren, T. J., M. W. Kaye, A. D. McCrumb, Y. Otsuki, B. Pfister, and C. A. Villa. 2000. Using new video mapping technology in landscape ecology. BioScience **50**:529-536.

Stone, K. D., G. A. Heidt, P. T. Caster, and M. L. Kennedy. 1997. Using geographic information systems to determine home range of the Southern Flying squirrel (Glaucomys volans). American Midland Naturalist **137**:106-111.

Stow, D., A. Hope, D. Richardson, D. Chen, C. Garrison, and D. Service. 2000. Potential of colour-infrared digital camera imagery for inventory and mapping of alien plant invasions in South African shrublands. International Journal of Remote Sensing **21**:2965-2970.

Sussman, R. W., M. G. Glen, and L. K. Sussman, editors. 1996. The use of satellite Imagery and anthropology to access the causes of deforestation in Madagascar. Columbia University Press, New York.

Sutherst, R. W. 2003. Prediction of species geographical ranges. J. Biogeography **30**:805-816.

Swain, P. H. 1978. Fundamental of pattern recognition in remote sensing. Pages 137–187 *in* P. H. Swain and S. M. Davis, editors. Remote sensing: The quantitative approach. McGraw-Hill, New York.

Systat. 2004. Syatat statistical software ver. 11. *in*. Syatat Software Inc.USA.

Tanner, E. V. J., V. Kapos, J. Adams, D. M. Newbery, H. H. T. Prins, and N. D. Brown. 1996. Tropical forests: spatial pattern and change with time, as assessed by remote sensing. Dynamics of tropical communities : the 37th symposium of the British Ecological Society, Cambridge University.

Terry, L. I., and G. J. Edwards. 1989. Efficacy of densitometric and multispectral techniques for monitoring infestations of citrus snow scale on citrus bark. Photogrammetric Engineering and Remote Sensing **55**:1471-1475.

Thompson, K., J. G. Hodgson, and T. C. G. Rich. 1995. Native and alien invasive plants: more of the same? Ecography **18**:390-402.

Tiwari, A. K., J. S. Mehta, O. P. Goel, and J. S. Singh. 1986. Geo-forestry of landslide-affected areas in a part of central Himalaya. Environmental Conservation **13**:299-309.

Tou, J. T., and R. C. Gonzalez. 1974. Pattern Recognition Principles. Addison-Wesley, Massachusetts.

Trani, M. K. 2002. The influence of spatial scale on landscape pattern description and wildlife habitat assessment. *in* J. M. S. e. al, editor. Predicting species occurances, issues of accuracy and scale. Island Press, Washington.

Tucker, C. J. 1979. Red and photographic infrared linear combinations for monitoring vegetation. Remote Sensing of Environment **8**:127– 150.

Tucker, C. J., B. N. Holben, and T. E. Goff. 1984. Intensive forest clearing in Rondonia, Brazil, as detected by satellite remote sensing. Remote Sensing Environ **15**:255-264.

Underwood, E., S. Ustin, and D. DiPietro. 2003. Mapping nonnative plants using hyperspectral imagery. Remote Sensing of Environment **86**:150-161.

Ungerer, M. J., M. P. Ayres, and M. J. Lombardero. 1999. Climate and the northern distribution limits of Dendroctonus frontalis Zimmermann Coleoptera: Scolytidae. J. Biogeography **26**:1133–1145.

Ustin, S. L., D. DiPietro, K. Olmstead, E. Underwood, and G. Scheer. 2002. Hyperspectral remote sensing for invasive species detection and mapping. International Geoscience and Remote Sensing Symposium 24th Canadian Symposium On Remote Sensing, Toronto, Canada.

Van Andel, J., and J. Aronson. 2005a. Concepts in restoration ecology. Pages 23-24 *in* J. Van Andel and J. Aronson, editors. Restoration Ecology: The New Frontier. Blackwell Scientific Publishing, Oxford.

Van Andel, J., and J. Aronson. 2005b. Restoration Ecology: The New Frontier. Blackwell Scientific Publishing, Oxford.

van der Valk, A. G. 1981. Succession in wetlands: A Gleasonian approach. Ecology **62**:688-696.

Veltman, C. J., Nee, S., Crawley, M.J. 1996. Correlates of introduction success in exotic New Zealand birds. American Naturalist **147**:542-557.

Venugopal, G. 1998. Monitoring the effects of biological control of water hyacinths using remotely sensed data: a case study of Bangalore, India. Singapore Journal of Tropical Geography **19**:92-105.

Vila`, M., and J. Pujadas. 2001. Land-use and socio-economic correlates of plant invasions in European and North African countries. Biological Conservation **100**:397-401.

Vitousek, P., L. Loope, C. D’Antonio, and S. J. Hassol. 1995. Biological invasions as global change. Pages 213-336 *in* S. J. Hassol and J. Katzenberger, editors. Elements of change 1994. Aspen Global Change Institute, Aspen, CO.

Vitousek, P. M., Mooney, H.A., Lubchenco, J., Melillo, J.M. 1997. Human domination of Earth's ecosystems. Science **277**:494-499.

Vrindts, E., J. De Baerdemaeker, and H. Ramon. 2002. Weed detection using canopy reflection. Precision Agriculture **3**:63-80.

Wallich, N. 1824-1826. Tentamen Florae Nepalensis Illustratae. Asiatic Lithographical Press, Calcutta.

Wallich, N. 1829-32. A Numerical List of dried specimens of Plants in the East India Company & Museum. British Museum, London.

Waltham, A., N. Stuart, D. Moss, and C. Place. 2002. Reducing uncertainties in natural resources mapping using remote sensing and GPS techniques. Pages 231-240 *in* G. Hunter and K. Lowell, editors. Accuracy 2002, 5th Internal symposium on spatial accuracy assessment in natural resources and environmental sciences, Melbourne, Australia.

Warren, M. S., J. K. Hill, J. A. Thomas, J. Asher, R. Fox, B. Huntley, D. B. Roy, M. G. Telfer, S. Jeffcoate, P. Harding, G. Jeffcoate, S. G. Willis, J. N. Greatorex-Davies, D. Moss, and C. D. Thomas. 2001. Rapid responses of British butterflies to opposing forces of climate and habitat change. Nature **414**:65-69.

Welch, R., M. Madden Remillard, and R. B. Slack. 1988. Remote sensing and geographic information system techniques for aquatic resource evaluation. Photogrammetric Engineering and Remote Sensing **54**:177-185.

Wells, M. J., R. J. Poynton, A. A. Balsinhas, C. F. Musil, H. Joffe, E. v. Hoepen, and S. K. Abbott. 1986. The history of introduction of invasive alien plants to southern Africa. Pages 21–35 *in* I. A. W. Macdonald, F. J. Kruger, and A. A. Ferrar, editors. The ecology and management of biological invasions in Southern Africa. Oxford University Press, Cape Town, South Africa.

Wilcove, D. S., and L. Y. Chen. 1998. Management costs for endangered species. Conservation Biology **12**:1405-1407.

Wilkinson, G. G. 1993. The generalization of satellite-derived thematic maps for GIS input. Geo-Informations-System **6**:24-29.

Williams, G. C. 1975. Sex and Evolution. Princeton University Press, Princeton.

Williamson, M. 1996. Biological Invasions. Chapman and Hall, London, UK.

Williamson, M. 1999. Invasions. Ecography **22**:5-12.

Wilson, E. O. 1997. Foreword. Pages ix-x in Simberloff, D., D. C. Schmitz, and T. C. Brown, editors. Strangers in paradise: impact and management of nonindigenous species in Florida. Island Press, Washington, DC.

Wilson, M. E. 1995. Travel and the emergence of infectious diseases. Emerg Infect Diseases 1:39–46.

Witkowski, E. T. F., and M. Wilson. 2001. Changes in density, biomass, seed production and soil seed banks of the non-native invasive plant, *Chromolaena odorata*, along a 15 year chronosequence. Plant Ecology 152:13-27.

Yonow, T., D. J. Kriticos, and R. W. Medd. 2004. The potential geographic range of *Pyrenophora semeniperda*. Phytopathology 94:805-812.

Young, J. A., R. A. Evans, and P. T. Tueller. 1976. Remote sensing for optimum herbicide application date for rabbitbrush. Journal of Range Management 29:342-344.

Zhang, M., X. Liu, and M. O'Neill. 2002a. Spectral discrimination of *Phytophthora infestants* infection on tomatoes based on principal component and cluster analyses. Int. Journal of Remote Sensing 23:1095-1107.

Zhang, M., X. Liu, and M. O'Neill. 2002b. Spectral discrimination of Phytophthora infestants infection on tomatoes based on principal component and cluster analyses. International Journal of Remote Sensing 23:1095-1107.

Zhu, Z., and D. L. Evans. 1994. US forest types and predicted percent forest cover from AVHRR data. Photogrammetric Engineering &Remote Sensing 60:525-531.

Zobel, M. 1997. The relative role of species pools in determining plant species richness: an alternative explanation of coexistence? Trends in Ecology and Evolution 12:266-269.

Epilogue*

Traveling downhill, “the big B” and my journey to PhD

Traveling downhill

“Do not chase monkeys, or they will attack you”. That was my mother’s advice when I left for school. However, my worries actually concerned a scary monitor lizard family who lived in a huge dead tree trunk along the trail on my way to school. Those were the days in 1972 when my family migrated from high mountains to the flat plains so called *Terai* (*Terai*: a narrow belt of alluvial and fertile flat plain of the southern boarder of Nepal extends from the westernmost part of the country to the eastern limit and covers about 17% of the total land) south of the Himalayas.

I was born in a remote village in the middle mountains in the west of Nepal. My family was fortunate since my father inherited sufficient land, which had not been divided among an increasing heir because my father, grandfather and great-grandfather were the only child in their family. Other families in our village, however, were less well off. Lack of land and low grain production mashed farmers with poverty and unemployment. As a result, many poor families from the hills migrated since the 1950’s towards the *Terai* region.

This migration in Nepal resulted from ‘push’ factors such as poverty and food insecurity and ‘pulls’ factors for instance availability of land and employment elsewhere. Plenty of fertile land was located in the lower foot slopes of the Himalayas. Until the 1950’s, this so called *Terai* was little populated. Indigenous malaria was preventing people from settling here. Only indigenous malaria resistant people such as the *Tharu*, occupied pocket settlements in dense jungles. For the others the *Terai* remained a forbidden frontier.

My brother, who later became a medical doctor, first told me about the malaria control programme. This USAID supported programme, aiming at malaria control first started in the central *Terai* in 1955, followed by others parts areas in 1963. Control activities included spraying of DDT and pyrethrum in addition to prophylactic treatment with paludrine tablets. By the mid 1960’s most of the *Terai* was declared save and freed of malaria.

The rumour quickly spread that fertile uninhabited lands freed of malaria were available in the *Terai*. It also reached my village. Many families abandoned their homes looking for brighter opportunities downhill. Soon our village was running empty. My parents, owning the better land in the village stayed until 1972. By that year, we were among the very few families remaining. Belonging to a so-called Brahman cast, my parents were not supposed to plough the field and we were always dependent on others to work for us. We decided to follow the others, not so much because of the opportunities, but of the dying social life in the village. My first brother (who is 20 years older than me) was the first migrant in my family. He was appointed as a school teacher in one of the *Tharu* villages. He claimed a small piece of land by cutting away some trees in the very dense jungle near Mahendranagar in 1968. I joined him with my parents in 1972.

---

* This epilogue is submitted to the WOTRO Newsletter

The big “B”

Initially, we were living amidst dense forest. Plenty of wildlife was still around. Out there, next to our big wooden house, I was fascinated by a flying fox family. They lived in a giant *Shorea robusta* tree and glided every evening for foraging. A vulture family worked hard to collect enormous amounts of heavy branches to built their nest on top of a *Terminalia alata* tree. Whenever I had some free time, I joined my playmate who never attended school. He was however, an expert “hunter”. We always looked for parrot nests and played “hide and seek” with the woodpeckers. They moved around tree trunks as we tried to glimpse them. A little further, a giant *Adina cordifolia* tree with a big hole in its trunk created a sort of “cave”- the perfect hiding place for us. Part of our game was to press the lateral side of *Chilanthus* (silver fern) leaves on our arms to create a beautiful silvery temporary “tattoo”. We were a disaster for the termite family since we constructed a “*Ghusudi*” or slide and with this, destructed the home of the termites.

Prior to 1961, Mahandranagar (my home town and the head quarter of the region which is named after late King Mahendra in 1961) was known as *Ghusudi* bazaar, where probably someone prior to us constructed a *Ghusudi* on some tall termite mountains.

My family cleared the forest to claim more and more land. Every Saturday, I was allowed to join them to set fire on dead trees and branches. Also, we collected latex of *Pterocarpus marsupium*, a valuable medicinal tree of which people believed to be a blood purifier or *Ophioglossum* plants for my grandmother, a wild vegetable for delicacy. We enjoyed the amazingly bright red and blue coloured fruits of *Rauvolfia serpentina*. Sometimes we raised our cattle in a near by part of the jungle, later gazetted as Royal Suklaphanta National Park and Wildlife Reserve.

Days passed and the forest disappeared quickly, as did my childhood. I had completed my secondary education in 1980 and went to India for higher education. Here, I graduated as a biologist in 1985 and immediately joined the National Herbarium and Plant Research Laboratories, under the Department of Plant Resources, Ministry of Forest and Soil Conservation as an assistant botanist. The forests, their ecology and Biodiversity -“the big B” became part of my professional life and drive. I explored my country in many botanical expeditions and started to appreciate its biodiversity when realizing there were more than 7000 flowering plant species out there.

Then I was appointed in the far west, the area where I spend my childhood. I found the forest all gone. Settlers had cleared it to plant their crops, and timber companies had seen to the rest. Yet I now realized that this was the very spot where in 1876 the first Rana Prime Minister Jang Bahadur had taken the Prince of Wales (later Edward VII) hunting for big game. The Royal party, transported through the woods and swamps by an army of 800 elephants, had on that occasion bagged a leopard, twenty Royal Bengal tigers and no less than a hundred deer.

In vain, I tried to find my lizard family, the *Terminalia* tree and the flying fox, the Adina “cave”, the vulture family, *Pterocarpus, Ophioglossum*, woodpeckers and termite mountains. Everything had changed, big concrete blocks of houses replaced all those species. Our family land was taken back by the government and two huge university campus complexes had been established there. However, I could see that the two mango trees planted by my brother and me were still standing just in front of the university canteen, the only witness of my childhood. The moist loving *Cheilanthus farinosa* (the juice of the plant is used as ear drops in the treatment of earache) had disappeared, replaced by *Ipomoea fistulosa* a useless exotic invasive species from tropical America.

My journey to PhD

Then, I looked at my neighbours. The land was dotted were thousands of houses, a pattern reminding me of the gone termitaria. The big pieces of land owned by their parents had been fragmented into small parcels. The house of the friends of my youth had been deprived. The forest were gone and where fragments remained there were holes in the canopy (Figure 9-6 left). I realized that the few remnant forest fragments were in poor condition. Even there, however, few of the original species survived. Instead the forests were degraded and disturbance loving species including invaders from the other side of the globe dominated the remnants of this part of our natural heritage.

Working in central Nepal I got particularly stuck by the widespread dominance of *Chromolaena odorata*, a major invasive weed of pasture, forest clearings, roadside and forest plantations, disturbed land and other similar land uses. No vegetation survives beneath its 2–3 m high dense thickets (Figure 9-6 right). Grasslands under disturbed forest environment are rapidly invaded, the grass shaded out and completely replaced by it. Bitter tasting pyrrolizidine alkaloids do not allowed animals to graze it and if cattle do eat it, the alkaloids progressively destroy their liver, and animals die. If forest degradation continues, soon *C. odorata* will replace native biota as well as valuable pasture by its dense growth, leaving the livestock to starve and the villagers further deprived of their source of milk, meat, cash income, non- timber forest products or medicinal plants.

Figure 9-6. Holes in the forest canopy: a hemispherical photograph of a heavily degraded *Shorea* forest near settlement (left) and *C. odorata* growing under *Shorea* forest (right).

Why was this? What were the root causes for this degradation? How and why do these forest canopy holes affect biodiversity. Did the degradation of the forest not affect human well-being as well? Should we blame the poor people for this and their own predicament? How about their government? Have they pursued policies that actually harm rather than sustain development?

Hundreds of these questions were with me when making the first move towards a WOTRO (the Netherlands Foundation for the Advancement of Tropical Research) scholarship. As I had a strong desire to learn about the spatial and temporal dynamics of those ecosystems, I applied for a NFP Fellowship (Netherlands Fellowship Programme) and started another MSc with the International Institute for Geoinformation Science and Earth Observation (ITC), the Netherlands, an international institute dealing with advanced GIS and remote sensing technologies.

Every time I turned on my computer, I experienced something new. Curiosity and fascination were generated by the pixels of satellite images of the country I knew so well. And, the more I experienced, my complex, scattered and multifaceted curiosities only raised my research hunger. My hunger for research was positively appreciated by WOTRO who in 2002 approved a PhD proposal I developed.

Since then as a researcher, I investigated the best way to locate holes in the forest canopy. Also I have answered the question why there are so many holes in those forests. What factors and government policies are responsible in creating such holes? What is the spatial extent of those holes? Why are forests having such holes in their canopy more susceptible to species invasion?

I am eventful to up-scale the spatial pattern of those holes to the national or continental extents. However, I still have to understand why floristic composition under those canopy holes changed. What happened to the *Cheilanthus* populations, flying fox or the wood peckers family who prefer dense forest canopies? During my PhD, I became familiar with techniques to address such questions and I feel confident I could support youngsters in Nepal to absorb this European scientific tradition.

Other questions still remain. There is the ambiguous feeling about the migration downhill which resulted in improved living conditions for people but also the vanishing of the friends of my youth. Although, no one will ask me why should the loss of flying fox should concern me? Why the species dynamics under such canopy holes is so important? However, my grand child will certainly ask me what the flying fox looked like or whether the delicacy of *Ophioglossum* was compensated by the benefits of the invasive *C. odorata*?

What next?

This is the personal account of my experiences preceding my journey to my doctoral research in Netherlands. However, it is the story of every person of my age who has migrated from the mountains to the *Terai*. This “going downhill” is an ongoing process. While I am writing this story, out there some families are on the way to the *Terai*. They continue to open up patches of forests (Figure 9-7). I doubt however whether the woodpeckers were still there to play “hide-and-seek”, because so many holes pierce the forest canopy these days.

Figure 9-7. A newly settled family house in the dense *Shorea* forest. I took this photo on 16 May 2005 near my home town Mahendranagar.

Only 40 years of age, I witnessed biodiversity of Nepalese ecosystems going downhill. It was affected as habitat decreased in size and health and the distances between forest patches greatly increased. Looking back, I realize that changes in habitat drove the woodpecker and *Ophioglossum* populations to local extinction. The disappearance of these species has certainly lowered “the big B”. However, I have no clue to questions for instance how many ecosystems, species and genetic diversity we have scratched. Or what is the impact of the invasive species we have introduced? And how do we keep those unwanted species under control so that they do not pose a further threat?

These questions look simple. However, I have not addressed them in this thesis. Nor has anyone found the answers yet. To me, it is crucial to mitigate the anthropogenic pressures put upon our native ecosystems. How people and their, institutions and organisations can be more effective in managing “the big B” still remains an apprehensive question at another level.

Returning to the *Terai* of my youth, *Ophioglossum* no longer grows there. No more safe drinking water left in the lake neighbouring my village due to an invasion of the water hyacinth (*Eichhornea crassipes*). One of the friends of my youth, *Rauvolfia serpentina* has been red listed in appendix II of the CITES endangered species list. Growing older I will act while re-introducing *Pterocarpus* to those ecosystems so that my grand children will be able to collect its latex to purify my aging blood.

Enschede, the Netherlands

ITC Dissertation List

1. Akinyede (1990), Highway cost modeling and route selection using a geotechnical information system
2. Pan He Ping (1990), 90-9003-757-8, Spatial structure theory in machine vision and applications to structural and textural analysis of remotely sensed images
3. Bocco Verdinelli, G. (1990), Gully erosion analysis using remote sensing and geographic information systems: a case study in Central Mexico
4. Sharif, M. (1991), Composite sampling optimization for DTM in the context of GIS
5. Drummond, J. (1991), Determining and processing quality parameters in geographic information systems
6. Groten, S. (1991), Satellite monitoring of agro-ecosystems in the Sahel
7. Sharifi, A. (1991), 90-6164-074-1, Development of an appropriate resource information system to support agricultural management at farm enterprise level
8. Zee, D. van der (1991), 90-6164-075-X, Recreation studied from above: Air photo interpretation as input into land evaluation for recreation
9. Mannaerts, C. (1991), 90-6164-085-7, Assessment of the transferability of laboratory rainfall-runoff and rainfall - soil loss relationships to field and catchment scales: a study in the Cape Verde Islands
10. Ze Shen Wang (1991), 90-393-0333-9, An expert system for cartographic symbol design
11. Zhou Yunxian (1991), 90-6164-081-4, Application of Radon transforms to the processing of airborne geophysical data
12. Zuviria, M. de (1992), 90-6164-077-6, Mapping agro-topoclimates by integrating topographic, meteorological and land ecological data in a geographic information system: a case study of the Lom Sak area, North Central Thailand
13. Westen, C. van (1993), 90-6164-078-4, Application of Geographic Information Systems to landslide hazard zonation
14. Shi Wenzhong (1994), 90-6164-099-7, Modeling positional and thematic uncertainties in integration of remote sensing and geographic information systems
15. Javelosa, R. (1994), 90-6164-086-5, Active Quaternary environments in the Philippine mobile belt
16. Lo King-Chang (1994), 90-9006526-1, High Quality Automatic DEM, Digital Elevation Model Generation from Multiple Imagery
17. Wokabi, S. (1994), 90-6164-102-0, Quantified land evaluation for maize yield gap analysis at three sites on the eastern slope of Mt. Kenya
18. Rodriguez, O. (1995), Land Use conflicts and planning strategies in urban fringes: a case study of Western Caracas, Venezuela
19. Meer, F. van der (1995), 90-5485-385-9, Imaging spectrometry & the Ronda peridotites
20. Kufoniyi, O. (1995), 90-6164-105-5, Spatial coincidence: automated database updating and data consistency in vector GIS

21. Zambezi, P. (1995), Geochemistry of the Nkombwa Hill carbonatite complex of Isoka District, north-east Zambia, with special emphasis on economic minerals

22. Woldai, T. (1995), The application of remote sensing to the study of the geology and structure of the Carboniferous in the Calañas area, pyrite belt, SW Spain

23. Verweij, P. (1995), 90-6164-109-8, Spatial and temporal modeling of vegetation patterns: burning and grazing in the Paramo of Los Nevados National Park, Colombia

24. Pohl, C. (1996), 90-6164-121-7, Geometric Aspects of Multisensor Image Fusion for Topographic Map Updating in the Humid Tropics

25. Jiang Bin (1996), 90-6266-128-9, Fuzzy overlay analysis and visualization in GIS

26. Metternicht, G. (1996), 90-6164-118-7, Detecting and monitoring land degradation features and processes in the Cochabamba Valleys, Bolivia. A synergistic approach

27. Hoanh Chu Thai (1996), 90-6164-120-9, Development of a Computerized Aid to Integrated Land Use Planning (CAILUP) at regional level in irrigated areas: a case study for the Quan Lo Phung Hiep region in the Mekong Delta, Vietnam

28. Roshannejad, A. (1996), 90-9009284-6, The management of spatio-temporal data in a national geographic information system

29. Terlien, M. (1996), 90-6164-115-2, Modeling Spatial and Temporal Variations in Rainfall-Triggered Landslides: the integration of hydrologic models, slope stability models and GIS for the hazard zonation of rainfall-triggered landslides with examples from Manizales, Colombia

30. Mahavir, J. (1996), 90-6164-117-9, Modeling settlement patterns for metropolitan regions: inputs from remote sensing

31. Al-Amir, S. (1996), 90-6164-116-0, Modern spatial planning practice as supported by the multi-applicable tools of remote sensing and GIS: the Syrian case

32. Pilouk, M. (1996), 90-6164-122-5, Integrated modeling for 3D GIS

33. Duan Zengshan (1996), 90-6164-123-3, Optimization modeling of a river-aquifer system with technical interventions: a case study for the Huangshui river and the coastal aquifer, Shandong, China

34. Man, W.H. de (1996), 90-9009-775-9, Surveys: informatie als norm: een verkenning van de institutionalisering van dorp - surveys in Thailand en op de Filippijnen

35. Vekerdy, Z. (1996), 90-6164-119-5, GIS-based hydrological modeling of alluvial regions: using the example of the Kisaföld, Hungary

36. Pereira, Luisa (1996), 90-407-1385-5, A Robust and Adaptive Matching Procedure for Automatic Modeling of Terrain Relief

37. Fandino Lozano, M. (1996), 90-6164-129-2, A Framework of Ecological Evaluation oriented at the Establishment and Management of Protected Areas: a case study of the Santuario de Iguaque, Colombia

38. Toxopeus, B. (1996), 90-6164-126-8, ISM: an Interactive Spatial and temporal Modeling system as a tool in ecosystem management: with two case studies: Cibodas biosphere reserve, West Java Indonesia: Amboseli biosphere reserve, Kajiado district, Central Southern Kenya

39. Wang Yiman (1997), 90-6164-131-4, Satellite SAR imagery for topographic mapping of tidal flat areas in the Dutch Wadden Sea

40. Saldana-Lopez, Asunción (1997), 90-6164-133-0, Complexity of soils and Soilscape patterns on the southern slopes of the Ayllon Range, central Spain: a GIS assisted modeling approach

41. Ceccarelli, T. (1997), 90-6164-135-7, Towards a planning support system for communal areas in the Zambezi valley, Zimbabwe; a multi-criteria evaluation linking farm household analysis, land evaluation and geographic information systems

42. Peng Wanning (1997), 90-6164-134-9, Automated generalization in GIS

43. Lawas, C. (1997), 90-6164-137-3, The Resource Users' Knowledge, the neglected input in Land resource management: the case of the Kankanaey farmers in Benguet, Philippines

44. Bijker, W. (1997), 90-6164-139-X, Radar for rain forest: A monitoring system for land cover Change in the Colombian Amazon

45. Farshad, A. (1997), 90-6164-142-X, Analysis of integrated land and water management practices within different agricultural systems under semi-arid conditions of Iran and evaluation of their sustainability

46. Orlic, B. (1997), 90-6164-140-3, Predicting subsurface conditions for geotechnical modeling

47. Bishr, Y. (1997), 90-6164-141-1, Semantic Aspects of Interoperable GIS

48. Zhang Xiangmin (1998), 90-6164-144-6, Coal fires in Northwest China: detection, monitoring and prediction using remote sensing data

49. Gens, R. (1998), 90-6164-155-1, Quality assessment of SAR interferometric data

50. Turkstra, J. (1998), 90-6164-147-0, Urban development and geographical information: spatial and temporal patterns of urban development and land values using integrated geo-data, Villaviciencia, Colombia

51. Cassells, C. (1998), 90-6164-234-5, Thermal modeling of underground coal fires in northern China

52. Naseri, M. (1998), 90-6164-195-0, Characterization of Salt-affected Soils for Modeling Sustainable Land Management in Semi-arid Environment: a case study in the Gorgan Region, Northeast, Iran

53. Gorte B.G.H. (1998), 90-6164-157-8, Probabilistic Segmentation of Remotely Sensed Images

54. Tegaye, Tenalem Ayenew (1998), 90-6164-158-6, The hydrological system of the lake district basin, central main Ethiopian rift

55. Wang Donggen (1998), 90-6864-551-7, Conjoint approaches to developing activity-based models

56. Bastidas de Calderon, M. (1998), 90-6164-193-4, Environmental fragility and vulnerability of Amazonian landscapes and ecosystems in the middle Orinoco river basin, Venezuela

57. Moameni, A. (1999), Soil quality changes under long-term wheat cultivation in the Marvdasht plain, South-Central Iran

58. Groenigen, J.W. van (1999), 90-6164-156-X, Constrained optimisation of spatial sampling: a geostatistical approach

59. Cheng Tao (1999), 90-6164-164-0, A process-oriented data model for fuzzy spatial objects

60. Wolski, Piotr (1999), 90-6164-165-9, Application of reservoir modeling to hydrotopes identified by remote sensing

61. Acharya, B. (1999), 90-6164-168-3, Forest biodiversity assessment: A spatial analysis of tree species diversity in Nepal

62. Akbar Abkar, Ali (1999), 90-6164-169-1, Likelihood-based segmentation and classification of remotely sensed images

63. Yanuariadi, T. (1999), 90-5808-082-X, Sustainable Land Allocation: GIS-based decision support for industrial forest plantation development in Indonesia

64. Abu Bakr, Mohamed (1999), 90-6164-170-5, An Integrated Agro-Economic and Agro-Ecological Framework for Land Use Planning and Policy Analysis

65. Eleveld, M. (1999), 90-6461-166-7, Exploring coastal morphodynamics of Ameland (The Netherlands) with remote sensing monitoring techniques and dynamic modeling in GIS

66. Yang Hong (1999), 90-6164-172-1, Imaging Spectrometry for Hydrocarbon Microseepage

67. Mainam, Félix (1999), 90-6164-179-9, Modeling soil erodibility in the semiarid zone of Cameroon

68. Bakr, Mahmoud (2000), 90-6164-176-4, A Stochastic Inverse-Management Approach to Groundwater Quality

69. Zlatanova, Z. (2000), 90-6164-178-0, 3D GIS for Urban Development

70. Ottichilo, Wilber K. (2000), 90-5808-197-4, Wildlife Dynamics: An Analysis of Change in the Masai Mara Ecosystem

71. Kaymakci, Nuri (2000), 90-6164-181-0, Tectono-stratigraphical Evolution of the Cankori Basin (Central Anatolia, Turkey)

72. Gonzalez, Rhodora (2000), 90-5808-246-6, Platforms and Terraces: Bridging participation and GIS in joint-learning for watershed management with the Ifugaos of the Philippines

73. Schetselaar, Ernst (2000), 90-6164-180-2, Integrated analyses of granite-gneiss terrain from field and multisource remotely sensed data. A case study from the Canadian Shield

74. Mesgari, Saadi (2000), 90-3651-511-4, Topological Cell-Tuple Structure for Three-Dimensional Spatial Data

75. Bie, Cees A.J.M. de (2000), 90-5808-253-9, Comparative Performance Analysis of Agro-Ecosystems

76. Khaemba, Wilson M. (2000), 90-5808-280-6, Spatial Statistics for Natural Resource Management

77. Shrestha, Dhruba (2000), 90-6164-189-6, Aspects of erosion and sedimentation in the Nepalese Himalaya: highland-lowland relations

78. Asadi Haroni, Hooshang (2000), 90-6164-185-3, The Zarshuran Gold Deposit Model Applied in a Mineral Exploration GIS in Iran

79. Raza, Ale (2001), 90-3651-540-8, Object-oriented Temporal GIS for Urban Applications

80. Farah, Hussein (2001), 90-5808-331-4, Estimation of regional evaporation under different weather conditions from satellite and meteorological data. A case study in the Naivasha Basin, Kenya

81. Zheng, Ding (2001), 90-6164-190-X, A Neural - Fuzzy Approach to Linguistic Knowledge Acquisition and Assessment in Spatial Decision Making

82. Sahu, B.K. (2001), Aeromagnetics of continental areas flanking the Indian Ocean; with implications for geological correlation and reassembly of Central Gondwana

83. Alfestawi, Y. (2001), 90-6164-198-5, The structural, paleogeographical and hydrocarbon systems analysis of the Ghadamis and Murzuq Basins, West Libya, with emphasis on their relation to the intervening Al Qarqaf Arch

84. Liu, Xuehua (2001), 90-5808-496-5, Mapping and Modeling the Habitat of Giant Pandas in Foping Nature Reserve, China

85. Oindo, Boniface Oluoch (2001), 90-5808-495-7, Spatial Patterns of Species Diversity in Kenya

86. Carranza, Emmanuel John (2002), 90-6164-203-5, Geologically-constrained Mineral Potential Mapping

87. Rugege, Denis (2002), 90-5808-584-8, Regional Analysis of Maize-Based Land Use Systems for Early Warning Applications

88. Liu, Yaolin (2002), 90-5808-648-8, Categorical Database Generalization in GIS

89. Ogao, Patrick (2002), 90-6164-206-X, Exploratory Visualization of Temporal Geospatial Data using Animation

90. Abadi, Abdulbaset M. (2002), 90-6164-205-1, Tectonics of the Sirt Basin – Inferences from tectonic subsidence analysis, stress inversion and gravity modeling

91. Geneletti, Davide (2002), 90-5383-831-7, Ecological Evaluation for Environmental Impact Assessment

92. Sedogo, Laurent G. (2002), 90-5808-751-4, Integration of Participatory Local and Regional Planning for Resources Management using Remote Sensing and GIS

93. Montoya, Lorena (2002), 90-6164-208-6, Urban Disaster Management: a case study of earthquake risk assessment in Carthago, Costa Rica

94. Ahmad, Mobin-ud-Din (2002), 90-5808-761-1, Estimation of Net Groundwater Use in Irrigated River Basins using Geo-information Techniques: A case study in Rechna Doab, Pakistan

95. Said, Mohammed Yahya (2003), 90-5808-794-8, Multiscale perspectives of species richness in East Africa

96. Schmidt, Karin (2003), 90-5808-830-8, Hyperspectral Remote Sensing of Vegetation Species Distribution in a Saltmarsh

97. Lopez Binnquist, Citlalli (2003), 90-3651-900-4, The Endurance of Mexican Amate Paper: Exploring Additional Dimensions to the Sustainable Development Concept

98. Huang, Zhengdong (2003), 90-6164-211-6, Data Integration for Urban Transport Planning

99. Cheng, Jianquan (2003), 90-6164-212-4, Modeling Spatial and Temporal Urban Growth
100. Campos dos Santos, Jose Laurindo (2003), 90-6164-214-0, A Biodiversity Information System in an Open Data/Metadatabase Architecture
101. Hengl, Tomislav (2003), 90-5808-896-0, PEDOMETRIC MAPPING, Bridging the gaps between conventional and pedometric approaches
102. Barrera Bassols, Narciso (2003), 90-6164-217-5, Symbolism, Knowledge and management of Soil and Land Resources in Indigenous Communities: Ethnopedology at Global, Regional and Local Scales
103. Zhan, Qingming (2003), 90-5808-917-7, A Hierarchical Object-Based Approach for Urban Land-Use Classification from Remote Sensing Data
104. Daag, Arturo S. (2003), 90-6164-218-3, Modeling the Erosion of Pyroclastic Flow Deposits and the Occurrences of Lahars at Mt. Pinatubo, Philippines
105. Bacic, Ivan (2003), 90-5808-902-9, Demand-driven Land Evaluation with case studies in Santa Catarina, Brazil
106. Murwira, Amon (2003), 90-5808-951-7, Scale matters! A new approach to quantify spatial heterogeneity for predicting the distribution of wildlife
107. Mazvimavi, Dominic (2003), 90-5808-950-9, Estimation of Flow Characteristics of Ungauged Catchments. A case study in Zimbabwe
108. Tang, Xinming (2004), 90-6164-220-5, Spatial Object Modeling in Fuzzy Topological Spaces with Applications to Land Cover Change
109. Kariuki, Patrick (2004), 90-6164-221-3, Spectroscopy and Swelling Soils; an integrated approach
110. Morales, Javier (2004), 90-6164-222-1, Model Driven Methodology for the Design of Geo-information Services
111. Mutanga, Onisimo (2004), 90-5808-981-9, Hyperspectral Remote Sensing of Tropical Grass Quality and Quantity
112. Šliužas, Ričardas V. (2004), 90-6164-223-X, Managing Informal Settlements: a study using geo-information in Dar es Salaam, Tanzania
113. Lucieer, Arko (2004), 90-6164-225-6, Uncertainties in Segmentation and their Visualisation
114. Corsi, Fabio (2004), 90-8504-090-6, Applications of existing biodiversity information: Capacity to support decision-making
115. Tuladhar, Arbind (2004), 90-6164-224-8, Parcel-based Geo-information System: Concepts and Guidelines
116. Elzakker, Corné van (2004), 90-6809-365-7, The use of maps in the exploration of geographic data
117. Nidumolu, Uday Bhaskar (2004), 90-8504-138-4, Integrating Geo-information models with participatory approaches: applications in land use analysis
118. Koua, Etien L. (2005), 90-6164-229-9, Computational and Visual Support for Exploratory Geovisualization and Knowledge Construction

119. Blok, Connie A. (2005), Dynamic visualization variables in animation to support monitoring of spatial phenomena

120. Meratnia, Nirvana (2005), 90-365-2152-1, Towards Database Support for Moving Object Data

121. Yemefack, Martin (2005), 90-6164-233-7, Modeling and monitoring Soil and Land Use Dynamics within Shifting Agricultural Landscape Mosaic Systems

122. Kheirkhah, Masoud (2005), 90-8504-256-9, Decision support system for floodwater spreading site selection in Iran

123. Nangendo, Grace (2005), 90-8504-200-3, Changing forest-woodland-savanna mosaics in Uganda: with implications for conservation

124. Mohamed, Yasir Abbas (2005), 04-15-38483-4, The Nile Hydroclimatology: impact of the Sudd wetland (Distinction)

125. Duker, Alfred, A. (2005), 90-8504-243-7, Spatial analysis of factors implicated in *mycobacterium ulcerans* infection in Ghana

126. Ferwerda, Jelle, G., (2005), 90-8504-209-7, Charting the Quality of Forage: Measuring and mapping the variation of chemical components in foliage with hyperspectral remote sensing

127. Martinez, Javier (2005), 90-6164-235-3, Monitoring intra-urban inequalities with GIS-based indicators. With a case study in Rosario, Argentina

128. Saavedra, Carlos (2005), 90-8504-289-5, Estimating spatial patterns of soil erosion and deposition in the Andean region using Geo-information techniques. A case study in Cochabamba, Bolivia

129. Vaiphasa, Chaichoke (2006), 90-8504-353-0, Remote Sensing Techniques for Mangrove Mapping

130. Porwal, Alok (2006), 90-6164-240-X, Mineral Potential Mapping with Mathematical Geological Models

131. Werff, Harald van der (2006), 90-6164-238-8, Knowledge-based remote sensing of complex objects: recognition of spectral and spatial patterns resulting from natural hydrocarbon seepages

132. Vlag, Daniël van de (2006), 90-8504-384-0, Modeling and visualizing dynamic landscape objects and their qualities

Curriculum vitae

Chudamani Joshi was born in 1964 in Jhigni, Devel Dibyapur, Dadeldhura district of Far West Nepal. He completed his primary education at Kedar Primary School, Kholi, Devel Dibyapur (1968-1972) and Baijnath High School, Mahendranagar, Kanchanpur (1972-1980) respectively. He graduate in 1985 with Bachelor of Science degree in Biology and Chemistry from Lucknow University, India. In 1989 he graduate with Bachelor of Legislative Law (specialization in Administrative law) from Lucknow University. In 1995 he joined Tribhuvan University, Kathmandu, Nepal and graduate with MSc in botany (specialization in plant taxonomy) in 1997.

Since 1986 he has been a permanent member of staff of the National Herbarium and Plant Research Laboratories under Department of Plant Resources, Ministry of Forest and Soil Conservation, Nepal and currently working as a research officer. As a manager and researcher he has long experience in botanical inventories, plant systematics, ethno-botanical survey and management of botanical gardens and Herbal farms. Between 1995-1997 he worked as a Botanist/Technical expert with HMGN/FINNIDA (Mitsahallitus, Finland) and involved in national forest inventory programme. He has visited almost all districts of Nepal on foot and collected a large number of plant specimen for National Herbarium, Kathmandu and many other international herbaria. He is author of 4 parts of flora of Nepal, some peer reviewed scientific papers, conference papers, project reports and other publications.

In 1999 he joined ITC for an another MSc course and graduated in Natural Resources Management (specialization in Rural Land Ecology) in 2001. In mid 2002 he received a WORTO fellowship and started his PhD research at ITC and Wageningen University the Netherlands that resulted in this thesis.

He married Geeta in 1989 and he is a father of daughter Richa and son Rishav.

Author's Publications

Books/chapters

Joshi, C. 1997. Flora of Nepal: Volume 5 Part 18, Apocynaceae. Department of Plant Resources HMG/Nepal

Joshi, C. and Chaudhary, R. P. 2000. Flora of Nepal: Malvaceae, Volume 4 part 32. Department of Plant Resources, HMG Nepal.

Joshi, C. 2000. Flora of Nepal: Bombacaceae, Volume 4 part 33. Department of Plant Resources, HMG Nepal

Joshi, C. 2000. Flora of Nepal: Stachyuraceae, Volume 4 Part 47. Department of Plant Resources, HMG Nepal

Theses:

Joshi, C. 1997. Revision of Malvaceae of Nepal, MSc. Thesis. Central Department of Botany, Tribhuvan University, Kathmandu, Nepal.

Joshi, C. 2001. Invasive Banmara (*Chromolaena odorata*), Spatial detection and prediction. MSc. Thesis. International Institute for aerospace Survey and Earth Science, Enschede, The Netherlands.

Peer review papers:

Joshi, C, J. De Leeuw, AK. Skidmore, I. C. Van Duren and H. van Oosten 2006. Remotely sensed estimation of forest canopy density: a comparison of the performance of four methods. International Journal of Geoinformatics and Earth Observation, (8) 2: 84-95.

Joshi, C., De Leeuw, J., van Andel, J., Skidmore, AK., Lekhak, HD., van Duren, IC and Norbu, N. 2006. Indirect remote sensing of a cryptic forest understorey invasive species. Journal of Forest Ecology and Management (225) 1-3: 245-256.

Peer review conference papers:

Joshi, C. 1998. New record of *Malachra capitata* L., a weed from Nepal. In: International Conference on Environment and Agriculture: Abs. Bio. 69. Ecological Society (ECOS), Kathmandu, Nepal, p 226.

Joshi, C. 1999. New record of Malvastrum coromandelianum L., for Nepal. Abstract BCS -062. In: III National Conference on Science and Technology: (March 8-11, 1999), Royal Nepal Academy of Science and Technology, Kathmandu, Nepal. 100 p.

Joshi, C., De Leeuw, J. and van Duren, IC. 2004. Application of remote sensing and geographical information system technologies in mapping biological invasion. ISPRS, 12-23 July 2004, Istanbul pp. 669-677.

Joshi, C., De Leeuw, J. van Andel , A. K. Skidmore, H. D. Lekhak, I. C. Van Duren 2005. Remote sensing and GIS for mapping and management of invasive shrub *C. odorata* in Nepal. AGILE 2005, 8th conference on Geographic Information Science, 26-28, May, 2005, Portugal pp. 71-80.

Joshi C., De Leeuw, J. and Skidmore AK. 2006. Upscaling species invasion patterns from local to regional for forest ecosystem management. ISPRS mid term symposeum: “Remote sensing: from pixel to processes”, Commission VI, WG VI/7; 8-11 May 2006, the Netherlands.

Project reports:

Joshi, C. 1997. General description of the bio-geography and forest vegetation of East Nepal. In: Forest Resources of the Eastern Development Region 1996, HMGN/FRIS-Project/Finland, Publication No. 70.

Joshi, C. 1998. General description of vegetation of the Western Development Region of Nepal. In: Forest Resources of the Western Development Region 1997. HMGN/FRIS-Project/Finland, Publication No. 71.

Joshi, C. 1998. Description and uses of the non-timber forest products recorded in the NFI of the hilly area of Western Development Region. In: Forest Resources of the Western Development Region 1997, HMGN/FRIS-Project/Finland, Publication No 71.

Joshi, C. 1999. General description of vegetation of Nepal. In: Forest Resources of Hilly Area of Nepal 1994-1999, HMGN/FRIS-Project/Finland, Publication No. 73.

Joshi, C. 1999. Description and uses of the non-timber forest products recorded in the National Forest Inventory of Nepal. In: Forest Resources of Hilly Area of Nepal 1994-1999. HMGN/FRIS-Project/Finland, Publication No. 73.

In review:

Joshi, C., De Leeuw, J., Skidmore, AK., van Duren, IC., Ferwerda, JG. 2006. Interrelated causes of invasibility of tropical forest ecosystems of Nepal.

Joshi, C., De Leeuw, J., and van Andel, J., 2006. Morphological plasticity in root structures and their role in invasion success of *Chromolaena odorata.*

Joshi, C., De Leeuw, J., Skidmore, AK, Vos, P. 2006. An integrated assessment of deforestation and forest degradation for biological conservation in the *Terai* of Nepal.

Joshi, C., De Leeuw, J., 2006. Modeling the spatial distribution of *Chromolaena odorata* in Nepal.

The work in this thesis was funded by :

the Netherlands Foundation for the Advancement of Tropical Research (WOTRO), residing under the Netherlands Organization for Scientific Research (NWO), the Netherlands.
Webpage: http://www.nwo.nl

and

International Institute for Geo-information Science & Earth Observation (ITC), Enschede, the Netherlands.
Webpage: http://www.itc.nl

This research work was carried out according to the PhD regulations of:

Wageningen University, the Netherlands.
Webpage: http://www.wageningenuniversiteit.nl/

Cover design by: Iris C. van Duren